Section Three

Grazing Systems —Pastures and Rangelands

Papers and Presentations

Impacts of Potential Management Changes on Greenhouse Gas Emissions and Sequestration from Dairy Production Systems in Australia

Richard Eckard[1,2], Dawn Dalley[2], Michael Crawford[3]

[1]ILFR, The University of Melbourne, Victoria

[2]Agriculture Victoria, Ellinbank, Victoria,

[3]Agriculture Victoria, Rutherglen Victoria

Introduction

GLOBAL CONTEXT

Under the terms of the Framework Convention on Climate Change (FCCC) Australia is obliged to provide an annual inventory of greenhouse gas emissions since 1990, as well as introducing policies and measures to limit greenhouse gas emissions.

The role of vegetation sinks in meeting commitments to the Kyoto Protocol will be a major focus of future international climate change negotiations. There is a need to provide policy makers with the best available information on which to base targeted research and the development of national positions on sources, sinks, and impacts of potential management changes to reduce emissions and improve carbon sequestration in the agricultural industries.

AGRICULTURE AND THE DAIRY INDUSTRY

Agriculture is estimated to contribute 22% of total land-based emissions in Australia and is estimated to contribute 63% and 80% of national methane and nitrous oxide emissions respectively (NGGI, 1997). The livestock industries contribute around 70% of total agricultural emissions, being 15% of national emissions, with dairy farming estimated at around 11.6% of agricultural emissions (NGGI, 1997).

However, current estimates of land-based sources and sinks are thought to be out by as much as 70%, while uncertainty associated with emissions from the agricultural sector is thought to be between 20 and 80% (Australian Greenhouse Office, 1999). Thus dairy farming activities are estimated to contribute less than 2% of national emissions, or including the unreliability of estimates 3.2%.

However, this low figure should not allow complacency. Methane emissions attributable to dairy farming show an increase of around 15% since 1990 (Australian Greenhouse Office, 1999), while nitrous oxide emissions from dairy pasture will continue to increase, given recent exponentially increasing trends in nitrogen (N) fertiliser use (Eckard *et al.*, 1997).

At present there is a dearth of information specific to intensive dairy pasture production systems and dairy farming in general, while management change options do exist which can reduce greenhouse emissions on dairy farms without large losses in productivity.

This report aims to identify potential 'win-win' management change opportunities for emission reductions and sinks within current dairy farming systems, based on the review and interpretation of available and published information.

Methane

Methane accounts for an estimated 16-29% of Australia's greenhouse gas emissions (Interface, 1994). This contrasts with other developed countries where methane typically accounts for 8-13% of emissions, with the exception of New Zealand where methane accounts for 45% of emissions (Judd *et al.*, 1999). Ruminant livestock are the largest producers of methane in Australia and this source constitutes about 12% of the national net emissions (Howden and Reyenga, 1999). About 90% of methane emitted by ruminants comes from fermentation in the rumen and about 10% from fermentation in the hindgut (Joblin, 1998). In the rumen, methane-producing microbes (methanogens) convert CO_2 and H_2 to methane. The rumen-sourced methane is released through the mouth and nostrils by eructation, and approximately 90% of the methane from the intestine is routed through the bloodstream and lungs to be expired also through the mouth (Murray *et al.*, 1976). The other 10% is emitted from the anus.

Since 1990 there has been an overall decrease in methane emissions from livestock of 2%, with increases in beef (24%) and dairy cattle (15%), but a 21% decrease for sheep (NGGI, 1998a). Projections of emissions from the livestock sector suggest an increase of 7% from 1990 to 2010 (Anon, 1997) but this figure must be used cautiously given uncertainty in future livestock numbers and market directions.

Victorian dairy cattle statistics show a significant decline in the number of registered farms from 14920 in 1975 to 8084 in 1998 (ADC, 1998). However, in contrast, the numbers of dairy cows in milk and dry have increased from 1.16 million to 1.23 million and annual milk production per cow has increased from 3100 to 4765 l/cow over the same period. In 1997/98 whole milk production from Victoria accounted for approximately 5900 million litres of the 9440 million litres produced nationally (ADC, 1998).

SOURCES AND ESTIMATES

1. Eructation

Most direct methane emission measurements are made from cattle under controlled indoor feeding conditions (Gibbs and Johnson, 1994), but the majority of more than 109 cattle globally graze outdoors. Therefore applying estimates of methane emissions from indoor-fed studies to a grazed-based dairy industry could constitute a large source of error.

Eructation of methane begins approximately 4 weeks after birth when solid feeds are retained in the reticulo-rumen (Anderson *et al*., 1987). Fermentation and methane production rates rise rapidly during reticulo-rumen development.

Estimates of yearly methane production of dairy cows range from 109-126 kg (EPA, 1993). Measurements made from indirect respiration calorimetry show methane losses vary from approximately 2 to nearly 12% of gross energy intake (Johnson *et al*., 1993).

Crutzen *et al*. (1986) made the first detailed assessment of global methane emissions from livestock. For cattle he used an emission factor of 55kg/hd/yr from cattle of developed countries (plus Brazil and Argentina) and 35 kg/hd/yr from those of developing countries. These emission factors are based on "methane yields" (percent gross energy intake lost as methane) in the range 5.5-7.5%.

Lassey *et al*. (1992) utilised statistical and nutritional information by livestock class in New Zealand, assessing methane emission on the basis of a 7.25% loss of gross energy intake. Equivalent emission factors were 80.6 kg/hd/yr (mature dairy cows) and 69.5 kg/hd/yr for dairy cattle (including bulls and young stock), highlighting the need to distinguish between "dairy cows" and all "dairy cattle".

The IPCC (IPCC, 1995) have also developed guidelines to assist in the calculation of methane emissions from ruminants where local data are absent. For cattle the emissions depend on regional characteristics and in the case of dairy cattle also on the level of milk production. For the Oceania region, default emission factors are 53 kg/hd/yr for non-dairy cattle including "beef cows, bulls and young stock", and 68 kg/hd/yr for dairy cows. The estimate for dairy cattle is based on Australian data and assumes a milk production of 1700 kg/hd/yr, which is only one third of current average milk yield.

To determine the methane emissions from grazing dairy cattle Lassey *et al.,* (1997) conducted research on 10 lactating dairy cows over a 5 day period. A total of 40 cow-days of breath samples were collected. Mean emission rates from individual cows were in the range 229–313 g/day, averaging 6.2% of gross energy intake. This extrapolates to a herd mean emission of 96 (4 kg CH_4/hd/yr).

Extrapolation of daily yields to annualised emissions is not without problems. Firstly, pasture and environmental conditions vary seasonally as does the physiological state of the animals. Secondly, emission factors are averaged across a population with changing age structure and over a range of pasture qualities (Lassey *et al*., 1997). Consequently emission factors (i.e. annual emissions) should be critically compared with emission rates measured from a specific animal class on a specific pasture at a specific season.

Once the main determinants of methane emission are better understood, there will be more confidence in assessing national emission inventories on the basis of management regime and feeding properties.

SINKS

The largest source of methane is from animals and animal waste, while the largest exchange is the dryland sink. Galbally *et al*. (1992) estimated dryland uptake of methane to be 2.5 Tg/yr with an uncertainty of 1 Tg/yr. However, the extent to which the 2.8M ha of dairy pastures nationally contribute to this sink is not clear.

MANAGEMENT AND CHANGE OPTIONS

1. Reduction in animal numbers

Assuming no new technology becomes available for cost-effective emission reductions, Australia will require a decrease of 200,000 dairy cows numbers to reduce methane emissions by 10%. Given current industry trends showing a 15% increase in dairy cow numbers since 1990 (ADC, 1998), it is unlikely that dairy cattle populations in south eastern Australia will decline significantly in the near future. In fact the opposite is more likely as the pasture-based industry in south eastern Australia strives to increase pasture utilisation and profitability by increasing stocking rates.

The proposed deregulation of the Australian dairy industry on July 1 2000 is most likely to reduce farm numbers. However, the impact on total cow numbers may not be great as the demographics of cow numbers may be the only real net change. The dairy industry will probably see movement in dairy production from subtropical areas in Queensland and northern NSW to the more temperate environments with higher quality pastures in southern NSW, Victoria and Tasmania. To maintain profitability under lower milk prices more emphasis will need to be placed on high pasture utilisation, decreased grain inputs and a push to maximising per hectare production at the expense of high per cow production.

2.Reductions of emissions per animal

a) Intensification

Increasing livestock performance has been suggested as a means of reducing emissions through reductions in emissions per unit product (Leng, 1991). Feeding livestock high digestibility feed such as grain or high quality pasture may seem to be one management option for reducing methane emissions, as emissions per unit of production are likely to be lower (Kurihara *et al*., 1998). However, the grain used will have resulted in emission of about 0.5–2.0 kg of CO_2 equivalents per kg up to harvest (Howden and O'Leary, 1997) as well as subsequent emissions associated with processing and transport. For the dairy industry the higher productivity of the UK system results in lower methane emissions per litre of milk compared with a typical Queensland farm (Kerr, 1993). However, the higher nitrous oxide emissions due to greater levels of N, both in the pasture consumed by the animals and in the level of fertiliser application, mean that the total greenhouse gas emissions from the UK dairy system are higher than those from the Queensland system (Howden and Reyenga, 1999). The Victorian dairy system has a similar pasture base to that in the UK, however the level of N input is considerably lower, therefore it is likely that methane emissions will be lower than those observed in Queensland; however this requires further investigation.

Offering high quality diets favours higher intakes and therefore high milk production per cow. Consequently the cost of maintenance is spread over a larger output of milk so methane produced per unit of milk is reduced. Additionally methane emission as a percentage of dietary gross energy intake will decline as daily intake increases (Johnson, *et al*. 1993). Unfortunately in the low cost pasture based production system operating in Victoria emphasis is on high per hectare production at the expense of high per cow intake and production.

Other opportunities may exist, within this low cost system, to increase productivity without increasing intake. Provision of improved animal health and husbandry (Morris, 1987) and providing adequate shelter and shading (Daly, 1984) have been shown to increase animal performance. However, the impact of these options on methane emissions has not yet been addressed.

b) Rumen Modifiers

Monensin is one of the only products shown to be consistently effective in reducing rumen methane emissions to date (Van Nevel and Demeyer, 1995). Its inhibitory effects include suppressing feed intake and suppressing acetate production, therefore reducing the total amount of H_2 released. Decreases in methane production range from slight to approximately 25% (Wedegaertner and Johnson, 1983). However, investigations indicate that the decrease in methane production is short-lived (Rumpler *et al*., 1986). Methane production per unit of diet by cattle fed either grain or forage diets returned to initial levels within 2 weeks. It appears that the reduction seen in methane production by ionophore supplemented cattle is likely to be related to the reduction in feed intake and not a direct effect on methanogenesis (Johnson and Johnson, 1995).

The use of antibiotics in ruminant feeds has recently been reviewed (JETACAR, 1999). The JETACAR report concluded that there is evidence that bacterial resistance in livestock may result in resistance to antibiotics in human medicine. If changes are made to current registrations it is possible that some antibiotics will no longer be an option to modify methane emissions from ruminants.

c) Dietary Fats

Fat additions to ruminant diets impact on methane losses by several mechanisms, including biohydrogenation of unsaturated fatty acids, enhanced propionic acid production and protozoal inhibition (Johnson and Johnson, 1995). Czerkawski (1969) concluded that the inclusion of linseed oil at 7% of the ration would cause a 37% reduction in ruminal methane emission. This translated to the loss of dietary gross energy as methane declining from 6% to 4%. The suppressing effect of long chain fatty acids on methanogenesis indicates a direct toxic action on rumen microbes (Machmuller *et al.*, 1998)

It is unlikely that the inclusion of dietary fats to dairy cow diets in Australia will play a major role in decreasing national greenhouse gas emissions primarily due to the additional cost of these additives. In a deregulated environment there will be more pressure to keep the costs of production down, consequently it is likely that there will be a greater reliance on pasture and less on supplementary feeds, especially those with costly additives.

d) Carbohydrate type

The type of carbohydrate fermented in the rumen influences methane production most likely through impacts on ruminal pH and the microbial population. Fermentation of cell wall fibre results in higher acetic: propionic acid and higher methane losses (Moe and Tyrrell, 1979; Beever *et al*., 1989). Moe and Tyrrell (1979) found fermentation of soluble carbohydrate to be less methanogenic than cell wall carbohydrate. Johnson and Johnson (1995) stated that non-cell wall components should be further separated into soluble sugars, which are more methanogenic than starch. Additionally, as a greater amount of carbohydrate fraction is fermented per day, whether from fibre or starch, methane production is decreased. The fermentation of brewers grain and distillery products containing relatively available fibre results in methane production half to a third of that seen with common feedstuffs of comparable digestibility.

Concentrating dairy production in the temperate zones of Australia could potentially decrease methane emissions since temperate pasture under good grazing management will be of higher digestibility than tropical pastures. Temperate pastures are likely to be higher in soluble carbohydrates and easily digestible cell wall components; therefore one would expect lower methane production. Current estimates of methane emission from grazing dairy cows on a range of pasture qualities are insufficient to calculate potential reductions per cow; this subject requires further research.

e) Forage Processing

Grinding and pelleting of forages can markedly decrease methane production (Blaxter, 1989). At high intakes, methane loss/unit of diet can be reduced 20–40%. Increased rate of passage of the ground or pelleted forage is the likely cause of the reduced methane production.

Like many of the previous methods of reducing methane emission, grinding and pelleting is unlikely to be a major contributor in Victorian dairy systems. Firstly, it increases the cost of the feed and secondly supplements generally only constitute a small proportion of the total diet with grazed pasture making up the bulk of the diet.

f) Defaunation

In the absence of protozoa, rumen methane emissions are reduced by an average of 20% (Hegarty, 1998). Decreased methane emissions from the protozoa-free rumen may be a consequence of:

A decreased methanogen population

An altered pattern of volatile fatty acid production, or

Increased partial pressure of oxygen in the rumen.

The success of defaunation on reducing methane production appears to be diet specific. Defaunation of the rumen of cattle fed a barley diet decreased methane production by approximately half (Whitelaw *et al.*, 1984). In contrast defaunation of animals on a high forage diet did not reduce methane losses (Itabashi *et al.*, 1984). Moate (1989) defaunated dairy cows grazing ad libitum white clover dominant pasture and measured a 2.7 l/cow/day increase in milk production and a 1.5 g/l increase in milk protein concentration. Rumen methane production was not measured but the milk yield and composition responses were consistent with the known effects of defaunation on volatile fatty acid production. Hegarty (1998) reviewed the current practices of defaunation and concluded that none of the currently available techniques are considered practical for commercial application.

One of the challenges of using defaunating agents to reduce methane gas emissions is the rate at which the animals refaunate. As a consequence regular dosing of the defaunating agent is required. Because dairy cattle are brought to the shed at least once per day for milking, and most cattle are offered supplements at milking, there exists a convenient mechanism for the administration of a defaunating agent. Additionally if compounds are effective in small concentrations in the rumen it may be possible to deliver them over reasonably long periods of time using an intra-ruminal, slow-release or controlled release device. Further investigation is required into suitable defaunating agents and mechanisms for delivering them.

g) Acetogens

In microbial ecosystems, such as those in the hind gut of rodents and humans, acetogenic bacteria (acetogens) rather than methanogens often function to remove H_2. Acetogens convert CO_2 and H_2 by reductive acetogenesis to acetate, a nutrient which the host animal can use. Acetogens are now known to be present in the rumen (Joblin, 1998) but their metabolic capabilities and their ecological significance in the rumen in the presence of methanogens is poorly understood. In cattle acetogen populations appear to be highly variable. Cattle in the USA contain high rumen population densities (about 108 per gram digesta) compared to grazing dairy cattle, where concentrations range from barely detectable to up to 106 per gram digesta (Joblin, 1998).

The jury is still out on the potential for reductive methanogens to lower ruminant methane emissions because little is known about the natural populations of acetogens in the gut of herbivores. Information on acetogens comes mainly from overseas studies on animals fed non-forage diets and there is a shortage of information on ruminal acetogens in grazing ruminants. Before the potential for reductive acetogenesis to lower ruminant methane emissions can be assessed we need to understand the factors affecting their population levels and activities in the rumen.

h) Vaccination

The methanogens are antigenetically distinct from other organisms in the rumen allowing a vaccination approach to the reduction of methane production by rumen methanogens (Baker, 1998). Using forage consumption constraint (FCC) the effectiveness of vaccination on rumen methanogens was investigated. In non-vaccinated sheep the FCC was not different from that predicted based on the energy required to shear the material. By comparison, FCC was reduced in vaccinated sheep and it was associated with a significant decrease in methane production *in vitro*. This technology offers the opportunity to reduce methane production in free-ranging ruminants if it can be successfully adapted from its current level of development.

ENTERIC METHANE—CONCLUSIONS

Until recently, national estimates of methane gas emissions from dairy cattle were calculated using emission rates measured in cows feed indoors on a forage/concentrate diet. How representative these emission rates are of grazing dairy cattle is in question and requires investigation. With the development of new techniques that enable methane emissions to be estimated in grazing dairy cows we are now in a better position to predict losses under Australian conditions. Methods available to measure methane emissions on both an individual cow, and a herd basis while grazing, should be utilised to provide more accurate estimates of methane emissions by the Australian dairy industry. Confidence in current estimates is low due to the substantial level of uncertainty associated with the estimates i.e. 3.3 (1.7 Tg CH_4/yr (Galbally, 1992)).

In addition emission rates have traditionally been determined in short term experiments and the daily results multiplied up to generate annual emissions. Extrapolation of daily yields to annualised emissions is not without problems. Firstly, pasture conditions and environmental conditions vary seasonally as does the physiological state of the animals. The second is that emission factors are averaged across a population with sustainable age structure and over a range of pasture qualities (Lassey *et al.*, 1997). Consequently emission factors (i.e. annual emissions) should be critically compared with emission rates measured from a specific animal class on a specific pasture in a specific season.

Many of the opportunities to reduce methane emissions e.g. fat supplementation, increased grain feeding, high per cow production etc. are not complementary to low cost production systems. If the current industry focus on per hectare production at the expense of high per cow production continues, new technologies to reduce methane emission per cow will be a necessity if Kyoto Targets are to be met in the dairy industry. The two current technologies which offer the most potential are defaunating agents and promoting natural populations of acetogenic bacteria in the rumen.

Carbon dioxide

SOURCES, SINKS AND ESTIMATES

1. Pastures and Soils

Pastures and soils utilised in dairying in Australia are not a major source of CO_2. Most pastures used in the dairy industry in Australia are located in higher rainfall areas or are under irrigation and consequently, are based on permanent perennial pasture species. On average, pastures are resown every 10–20 years and when resown are typically direct drilled or oversown with improved species. Approximately 10 to 15% of dairy pastures are sown to annual pasture species, the majority of these being in WA (80% or 6% of total dairy pasture), Queensland (20% or 3% of total dairy pasture) and NSW (20% or 4% of total dairy pasture), with majority of pastures based on perennial species.

Organic carbon stocks in soils under dairy pastures vary enormously with soil type and locality, but due to higher rainfall and cooler temperatures in areas where the majority of dairying occurs and the permanent nature of the pastures, carbon levels are higher than under other agricultural enterprises in Australia. Data from Victoria shows that the amount of soil carbon in the 0–60 cm zone can vary from 140 to 280 t C/ha (Crawford *et al.*, 1999).

Practices that would lead to the rundown of these levels (i.e. fallowing, cultivation, burning, conversion to annual vegetation) do not frequently occur in these farming systems. It has been suggested that one minor source of CO_2 release may be the practice of growing annual fodder crops. This amounts to around 400,000 ha nationally (c. 14% of total dairying area) and generally involves a number of cultivations to produce a fine seedbed for sowing. Data for soil carbon changes under fodder cropping are hard to find. However, research shows that the reduction in soil organic carbon is usually rapid after the initial cultivation event with high organic matter soils losing a greater proportion of carbon compared with soils with low levels (Mann, 1986). Extrapolating from field crops, the loss of carbon upon cultivation may be as high as 10%. However, the growing of fodder crops in the dairy industry is usually done as part of a pasture renovation operation, and after the fodder crop is harvested, the land is sown down again to permanent pasture, where soil carbon levels will soon return to pre fodder crop levels. Thus the net carbon effect of fodder cropping over time would be zero.

2. Enteric

While there is a dearth of information on methane emissions from ruminants, carbon dioxide emissions do not rate a mention. Moate *et al.* (1997) reported that, before grazing the rumen headspace of dairy cattle was composed of carbon dioxide 65%, methane 31% and N 4% whereas one hour after grazing the headspace was composed of carbon dioxide 76%, methane 22% and N 2%. Assuming the headspace composition represents that eructated then dairy cattle belch approximately 2.7 times more carbon dioxide than methane. However, if we assume annual methane emission per cow is approximately 100 kg CH_4 (2100 kg CO_2 eq), the corresponding carbon dioxide emission would be approximately 346 kg CO_2/cow/year.

Because of the lack of information on CO_2 emissions from ruminants it is impossible to predict management changes which might reduce emissions.

3. Energy and Fuel

Power consumption on dairy farms was estimated using data from ABARE (1998), NGGI (1997) and a limited survey of dairy farmers on-farm costs (Table 1). It must be emphasised that these figures are indicative only, and are not reliable, given the limited scope of our phone survey, but may be useful for comparative purposes.

The data in Table 1 highlight the higher consumption of diesel in states with greater reliance on annual pastures i.e. WA. Higher electricity consumption is also noted in states where irrigation water is pumped i.e. SA and Tas, rather that flood irrigated.

	NSW	VIC	QLD	WA	SA	TAS	Australia Total
Diesel							
Litres/year	6004	4423	4427	7072	5704	4502	4822
Tonnes CO_2 /year	17	12	12	20	16	12	13
Electricity							
KWh/year	75436	51453	53637	67029	83264	77265	59070
Tonnes CO_2 /year	68	46	48	60	75	70	53
Total CO_2 (tonnes/year)	84	59	60	80	91	82	66

Data taken from a limited survey of local farmers, ABARE (1998) and NGGI (1997)

Table 1. Average power consumption on dairy farms in Australia

MANAGEMENT CHANGE OPTIONS

1. Pastures and Soils

Some opportunities exist to increase the sequestration of carbon in soils under dairy pastures. This could be done through increasing fertiliser inputs (especially phosphorus) and stocking rates. Increasing fertiliser inputs increases primary productivity, above and below ground, thereby increasing the inputs of carbon. Increasing stocking rates can lead to greater conversion of above ground dry matter to manure, which can be more readily incorporated in the upper layers of the soil. Experimental data to support this hypothesis are difficult to find but a survey of low, medium and high input dairy pastures in Gippsland, Victoria showed that soil carbon levels (to 60 cm) were 239, 259 and 273 t C/ha respectively (Crawford *et al.,* 1999). Whilst most dairy pastures would at least be receiving moderate levels of phosphorus input, there appears to be scope for sequestering another 10 t C/ha by increasing inputs on a greater number of farms.

Whilst N fertiliser will often increase pasture yield, there can be a decrease in soil carbon levels at high levels of N. On a Rhodes grass dairy pasture in south-eastern Queensland, organic carbon levels in the top 10 cm decreased from 2.3% at 150 kg N/yr to 1.4% at 600 kg N/yr (Cowan *et al*., 1995). Presumably, this is because of the stimulating effect of mineral N on organic matter decomposition and reduced allocation of resources to root growth where N availability is not restricted (Whitehead, 1995).

Other opportunities for increasing carbon sequestration in dairy pasture soils are minimal. The predominant grass species in dairy pastures in temperate Australia is perennial ryegrass, which is relatively shallow rooted, compared with phalaris (Pook and Costin, 1971). It has been suggested by Oades (1988), that turnover of organic matter in soil can be reduced if it is placed deeper in the soil where it is less accessible to decomposer organisms. Following this logic, carbon sequestration may be improved through the replacement of perennial ryegrass pastures with phalaris or tall fescue. However, in most situations, perennial ryegrass gives better milk production than phalaris and fescue pastures and the adoption of phalaris and/or fescue as a replacement for perennial ryegrass on a greater scale would be uneconomic and unacceptable to the dairy industry.

A need exists to quantify the impact of changes in fertiliser and grazing management on soil carbon levels in the dairy industry. Because soil organic carbon has not been seen as a limiting factor or as a driver of productivity in the past, it has been largely neglected in dairy pasture research. A first step would be a detailed analysis of soil carbon levels in all the grazing and fertiliser management trials in the dairy industry in Australia. Subsequent research should focus more on the input and decomposition processes under dairy pastures—work that has largely been done in the lower rainfall crop-livestock zone to date.

2. Trees

The potential impact of revegetation with trees on carbon sequestration has been discussed elsewhere (Australian Greenhouse Office, 2000). Given the relative high value of dairy enterprises per hectare, substantial economic benefits would need to be derived from tree planting on dairy farms to make them competitive. Under a best-case scenario, it may be possible that the area of trees on farms could be increased to up to 5% of the farm area without significant detrimental impacts on the dairy enterprise. This would need to occur as a part of a whole farm plan, where consideration is given to shelter, aesthetics, biodiversity, and potential income from wood products, as well as carbon sequestration potential. However, given the current high value of dairying in many parts of Australia, relative to growing trees, it is unlikely that there will be a significant increase in tree planting.

However, if Emissions Trading recognises small areas of tree planting on dairy properties (i.e. allow the aggregation of tree planting on farms into an allowable 'Kyoto forest'), this could result in a win-win situation for the farmer and the environment. Not only would there be some financial gain from the sale of timber and carbon credits, but there would be the added benefit of salinity reduction, through a lower water table, nutrient buffer strips on drainage lines and creek banks, shade and shelter reducing cold and heat stress in cows and improved aesthetics resulting in an improved public perception of clean and green agriculture. Under these conditions, where there would be an economic and environmental benefit in addition to timber sales, it is estimated that more than 10% (280,000 ha) of current dairy land could be re-forested.

3. Energy and fuel

Management change options presented here include those suggested by the surveyed farmers and relate entirely to reducing power consumption and not improving carbon sequestration.

a. Electricity

As dairy sheds usually have a large roof area, the use of solar energy for water heating and running the plate cooler/heat exchanger will reduce electricity consumption considerably. Data to quantify these savings were not found.

Most dairy farms run on 2-phase power, while start up power use of a 2 phase motor is substantially higher than that of 3-phase. At the same time, 3-phase motors are more efficient, last longer and would use less electricity over their lifetime. However, the capital cost of installing 3-phase power, where farmers have to pay for long distances of cabling, make this an unattractive option without government incentive.

b. Diesel

Diesel consumption on dairy farms, while higher than other grazing industries, is much lower than all cropping industries. However, the use of more perennial pastures, particularly in states like WA and, to a lesser extent in NSW and SA, would result in lower emissions from diesel fuel. This management change is not likely until research provides more perennial forages for these regions.

Automotive Diesel Oil (ADO) has a higher CO_2 intensity than most 'lighter' fuels. Conversion of diesel motors to compressed natural gas could save up to 50% of current emissions (NGGI, 1997). Regular servicing of diesel motors, particularly timely replacement of diesel injectors will improve engine efficiency and reduce emissions by up to 25%.

Nitrous Oxide

SOURCES

Historically the dairy industry has used N fertiliser sparingly, with most of the N being sourced from pasture legumes in the temperate regions of Australia. However, N fertiliser use on dairy pasture has increased exponentially over the past 15 years, with over 60% of dairy farms in south eastern Australia applying up to 200 kg N/ha annually (Eckard *et al.*, 1997, Eckard, 1998). Nitrogen fertiliser applications on dairy pastures in New South Wales and Queensland are higher with around 300 kg N/ha being applied on temperate pastures and around 100 to 150 kg N/ha applied to Kikuyu or other sub-tropical pastures (Cowan *et al.*, 1995; Lowe *et al.*, 1996; Fulkerson per comm.).

White clover and sub-clover are the main sources of legume N_2 in dairy pastures, fixing anywhere between 20 and 280 kg N/ha annually (Eckard, 1996, Rifkin *et al.*, 1997). However, given that temperate pastures require between 450 and 600 kg N/ha annually for 90% of maximum production (McKenzie and Jacobs, 1997; Eckard, 1998), even with the current strategic management of N fertiliser growth rates of dairy pastures are N-limited for most of the active growing season.

Denitrification losses from dairy pastures are expected to be higher than most other grazing industries due to higher N fertiliser use, higher stocking rates and concentrate feeding. Feed supplements alone add between 30 and 50 kg N/ha annually through the excreta.

Nitrogen in the waste of livestock is the second largest anthropogenic source of nitrous oxide in Australia, although actual emissions are much lower than losses direct from soil (NGGI, 1998b). Nitrous oxide emissions from manure management systems can occur via combined nitrification-denitrification of ammoniacal N contained in the wastes. The amount released depends on the system and duration of waste management. The emissions of nitrous oxide from livestock themselves are very small and are not estimated in line with the IPCC (1995) guidelines.

ANNUAL LOSS ESTIMATES

1. Soils and Pastures

Recent reviews and Australian studies of denitrification rates in pasture soils have provided denitrification estimates ranging between 0 and 5 kg N/ha where no N fertiliser was applied and 17 to 25 kg N/ha with the application of N fertiliser (Denmead *et al.*, 1979; Chen *et al.*, 1996; Barton *et al.*, 1999). Based on daily denitrification rates reported by Denmead *et al.* (1979) and Galbally *et al.*, (1980), Ellington (1986) estimated annual denitrification losses from legume-based pasture between 6 and 60 kg N/ha per year. However, the extrapolation of daily denitrification estimates, from short-term studies to annual losses, is highly questionable due to the high climatic, edaphic and temporal variability of losses.

Current research on grazed dryland grass/clover dairy pastures in Victoria has measured annual denitrification losses of 8 and 14 kg N/ha, with N_2O-N losses of 6.5 and 11.3 kg N_2O-N /ha, from annual N applications of 0 and 200 kg N/ha, respectively (Eckard *et al.*, 2001; Eckard *unpublished data*). These losses agree with New Zealand data where Ledgard *et al.* (1996) reported denitrification losses of 6–15 kg N/ha/yr, the range largely a function of increasing N fertiliser use. Fortunately, for most of the dryland winter rainfall regions, denitrification is restricted by low soil moisture in summer and low soil temperature in winter. Thus peak losses of N would only occur after the autumn break and in early spring, where denitrification losses as high as 14% of N applied have been recorded (Eckard *unpublished data*).

Losses from irrigated pasture in South Australia were reported at 12% of applied N fertiliser (Packrou *et al.*, 1997). Mundy and Mason (1989) reported apparent denitrification losses of 14 and 35% of N fertiliser applied as urea and ammonium nitrate on flood irrigated pasture in northern Victoria, respectively. These losses equate to annual losses of up to 75 kg N/ha given the high rate of summer N fertiliser application in flood irrigated pasture. However, these data were estimated from N balance studies and not measured directly. Barton *et al.* (1999) reviewed denitrification losses as high as 239 kg N/ha or 37% of applied N from N fertilised and irrigated grassland in the USA, indicating that large denitrification losses are likely from N fertilised and flood irrigated pastures in Australia. Given that flood irrigated pasture makes up around 25% of dairy pasture in Australia (ABARE, 1998), these losses may be significant.

Denitrification losses from dairy pastures in the tropics and sub-tropics of Australia are also likely to be high, if losses of 2 to 25 kg N/ha from sugar cane soils are any indication (Weier *et al.*, 1996). These losses may be nationally significant given that dairy pastures in northern New South Wales and Queensland comprise approximately 21% of national dairy pasture land.

From the above review it is clear that the greatest loss of nitrous oxide from dairy pastures is likely from the 1 to 1.5 M ha areas of irrigated pasture in Australia, particularly flood irrigated areas where no studies have clearly quantified annual denitrification losses under summer N fertilisation.

2. Livestock waste

Nitrous oxide losses from the application of dairy waste to pasture would depend largely on the N content of waste being applied, with N_2O-N losses being similar to an equivalent N fertiliser rate, assuming the volume of effluent water applied did not cause water-logging. Dairy effluent may contain between 100 to 320 kg N/ML (Hopkins and Waters, 1999), with current recommendations applying up to 25 mm of effluent per hectare. This rate of water application may not result in waterlogging, but may apply as much as 80 kg N/ha. Given that these applications are usually made to a grass/clover pasture in the dry summer, when grass growth is limited by moisture stress, this additional N would be adding to an already under-utilised pool of clover-derived N_2 and mineralised N. Under such conditions, denitrification rates may be high, depending on the availability of the N in the effluent and waterlogging of the soil.

Losses of N_2O from animal waste management systems (AWMS) appear extremely low in the national context, although not negligible. In the 1997 National Greenhouse Gas Inventory, N excretion into anaerobic animal waste management systems was only 3% of the loss from agricultural soils and cattle waste contributed only 2.3% of total agricultural emissions.

SINKS

Although it is physically and biologically possible, no Australian studies have shown soils to act as a nitrous oxide sink (Galbally, 1991).

MANAGEMENT CHANGE OPTIONS

1. Nitrogen fertiliser use

In global terms N fertiliser use on Australian dairy pasture is reasonably low, but increasing exponentially (Eckard *et al.*, 1997). However, opportunities for reducing N fertiliser applications are limited, given that N fertiliser is mainly used as a strategic management intervention, with dairy pastures being N limited. Denitrification loss from the summer application of N fertiliser in flood irrigated pasture systems require further investigation, before management change options can be formulated further.

Where N fertiliser is applied to pasture, the use of urea rather than ammonium nitrate should reduce the free nitrate in the soil available for denitrification and leaching during the colder months, but will not retard pasture responses. However, most dairy farmers already use urea or diammonium phosphate (DAP) as the main source of N, as ammonium nitrate is almost double the elemental cost. These effects still require further investigation.

The extension of current Best Management Practices for N fertiliser use on dairy pastures is viewed as the most effective means of ensuring efficient N use and thus minimal loss through denitrification. Data from current research indicates that judicious management of N fertiliser on dryland dairy pasture in south eastern Australia can maintain denitrification losses below 5% of N inputs, with total N losses being less than 30% of total N inputs (Eckard *et al.*, 2001; Eckard *unpublished data*). This contrasts with many European countries where total N losses may be between 60 and 80% of total N inputs.

Guidelines for farmers in south eastern Australia have been published and are available at www.nitrogen.landfood.unimelb.edu.au. A software decision support system has also been developed to assist dairy farmers in the management of N fertiliser.

2. Legumes

The use of legumes in intensive pasture systems is considered the key to the global competitive advantage of the pasture-based Australian Dairy industry. Estimates of the annual contribution of legumes to the N status of temperate pasture range between 20 and 280 kg N/ha annually (Eckard, 1996; Rifkin *et al*., 1997). Given that this level of N supply is now considered insufficient to maintain required levels of pasture production for higher stocked farms, particularly through the cooler months, there would be both social and economic resistance to reducing legume N inputs.

3. Waterlogged soils

Denitrification rates appear to be enhanced for longer periods in poorly drained soils (Barton *et al*., 1999). A high proportion of dairy pastures in south eastern Australia are poorly drained, being seasonally water logged for long periods in winter and early spring. Drainage of these soils may reduce denitrification loss, but local estimates of reductions in denitrification are not available. A recent review of forest soils (Barton *et al*., 1999) implied a reduction in denitrification from 40 to <1 kg N/ha/year with improved drainage. In such a review, however, it is difficult to separate soil structural impacts (water-filled porosity) from drainage *per se*. The drainage of seasonally waterlogged soils will also result in some reduction in methane emission, although the magnitude of these reductions are not known and are likely to be small.

Improved drainage will also reduce pugging damage to soil and pasture thereby improve N extraction from the soil through more efficient plant growth. However, improved drainage may increase the leaching of soil nitrate. At present there is a DRDC funded project investigating management options for seasonally water-logged dairy pasture soils (G. Ward, pers.comm.)).

Apart from the greater volume of water required for flood irrigation, waterlogging cannot be managed as well as under spray irrigation. The conversion of flood irrigation systems to spray irrigation, apart from reducing water use substantially, has potential to reduce denitrification loss. Reductions achieved may be significant given that approximately 25% of dairy pastures are currently flood irrigated. However, there are enormous social and economic constraints to this management change option, in addition to the carbon implications of increase electricity consumption from pumping.

4. Livestock waste

Currently dairy farmers may apply up to 25 mm of effluent per hectare, which may apply more than 80 kg N/ha in a single application. A nutrient analysis of effluent prior to application to pastures is essential to ensure that N application rates do not exceed current best management recommendations of a maximum of 50–60 kg N/ha for any single regrowth period. In most cases this will reduce the effluent volume spread to recommended volumes of between 20 and 25 mm/ha.

Nitrous oxide emissions from manure management systems can occur via combined nitrification-denitrification of ammoniacal N contained in the wastes. The amount released depends on the system and duration of waste management. Although no specific data were available and estimates of N_20 loss from anaerobic waste appear very low (NGGI, 1997), reducing the duration of digestion of waste before land application will reduce both N_20 and methane emissions. On most dairy properties in south eastern Australia, the land application of waste occurs annually in the drier summer or early autumn. This practice is designed to minimise the leaching of nutrients and maximising the water benefits at a time when soil moisture is usually limiting pasture growth.

Summary of Management Change Options and Research Requirements

CARBON DIOXIDE: SOILS AND PASTURE

1. Most dairy pastures are based on perennial species, with direct drilling used for renovation offering limited scope for increasing C sequestration.
2. Most dairy pastures are based on perennial ryegrass, while other pasture species, like phalaris and fescue are deeper rooted placing organic carbon deeper in the soil. However, dairy farmers would resist adopting these species because of their historical lower milk production potential. Modern cultivars of these species need to be assessed for milk production potential and these data extended to dairy farmers.
3. Increasing P fertiliser rates may sequester another 10 t C/ha on farms currently receiving low to medium rates of P. This requires further investigation.
4. High rates of N fertiliser may increase the loss (up to 0.9%) of soil organic carbon in the long-term, while regular lower

rates of N fertiliser may increase soil carbon marginally. This requires further investigation.

5. Soil carbon levels under all comparative fertiliser and grazing management trials in dairy areas could be analysed in order to get a more accurate understanding of the carbon sequestration potential of various management practices.
6. The rates of carbon input and decomposition in dairy pasture soils requires further study in order to understand the longer term dynamics of carbon under dairy pasture.

CARBON DIOXIDE: TREES

1. There is an opportunity to increase tree planting on dairy farms, occupying around 5% of existing land on areas considered marginal for dairying i.e. creek banks, drainage lines. However, farmers would need to be able to derive a carbon credit from these trees, in addition to timber sales, for there to be a profit incentive. Potential additional benefits would include salinity reduction, nutrient buffer strips, shade and shelter and improved public perception of clean and green agriculture.

CARBON DIOXIDE: POWER CONSUMPTION

1. Given the large roof area of most dairy sheds, the use of solar power for water heating will reduce electricity consumption.
2. The installation of 3-phase power on dairy farms, while a prohibitive capital cost in many cases, will result in considerable savings in electricity and efficiency of equipment.
3. The conversion of diesel motors to compressed natural gas could save up to 50% of current emissions.
4. The regular servicing of diesel motors, particularly timely replacement of injectors will improve engine efficiency and reduce emissions by up to 25%.

NITROUS OXIDE

1. Current N fertiliser applications on dairy pastures are low and usually well managed. The extension of current best management practices to dairy farmers should ensure that N losses through denitrification remain below 5% of total N inputs in the dryland regions of south eastern Australia.
2. Denitrification loss from the summer application of N fertiliser in flood irrigated pasture systems may be significant and require further investigation.
3. Drainage of wet soils may result in significant reductions in denitrification, as well as possible reductions in methane production. However, no data are available to quantify these reductions.
4. The conversion of flood irrigated pasture to spray irrigation should result in reduced denitrification, as well as more efficient water use. However, this would lead to greater electricity consumption, in addition to the prohibitive capital cost of irrigation installation. Further research is required on denitrification rates of summer N fertiliser applications to flood irrigated pastures.
5. Nutrient analysis of effluent water prior to land application will ensure that N applications to not exceed current best management recommendations of 50–60 kg N/ha or 20 to 25 mm effluent/ha for any single regrowth period.
6. Where possible, reducing the duration of effluent storage and digestion may reduce both methane and N_20 loss.

METHANE

1. Deregulation in the dairy industry may result in a reduction in cow numbers in tropical and sub-tropical latitudes, and a decline in farm numbers nationally, but a continued net increase in dairy cows in the temperate high-rainfall regions. As this will result in less tropical forages in cow diets, there may by a net reduction of not more than 5% in national methane output from dairy cows due to deregulation.
2. The relevance of methane emission data from indoor-fed cattle to pasture-based grazing systems, as applied in Australia needs to be investigated further.
3. Reducing methane emissions per unit of production is not highly feasible in the pasture-based dairy industry as maximum efficiency results from maximum production per unit area, at the expense of production per cow. However, the effect of strategic use of supplements on methane production per cow and per unit of production requires further investigation. Similarly, the methane production of cows on a range of pasture qualities requires further study.
4. The use of antibiotics as a rumen modifier is currently under review, which is likely to result in the removal of this option for methane reduction in the long-term.
5. Fat additions to ruminant diets may decrease methane emissions by as much as 37%. Likewise, the grinding and pelleting of forages may reduce methane emissions by up to 40%. However, the high carbon cost of processing and feeding these fats and pellets make them an unlikely feed additive in a deregulated and low production cost industry.
6. Defaunation of the rumen protozoa may result in a 20% reduction in methane emission. Research is still required to

identify feed additives that can be economically fed to cows on a regular basis to prevent refaunation.

7. Higher populations of acetogens in the rumen may result in lower methane production. However, research is still required to understand the factors affecting acetogen population levels and activities in the rumen.
8. Vaccination of ruminants to eliminate methanogenic microbes offers the opportunity to reduce methane production in free-ranging ruminants if the technique can be successfully adapted from its current level of development.
9. More regular land application of effluent onto pasture will reduce methane lost from effluent ponds. The potential magnitude of this reduction is unclear.

References

ABARE (1998). Australian Dairy Industry Survey: Regional Results 1995–96 to 1997–98. Report to client. Australian Bureau of Agricultural and Resource Economics, Camberra, Project no: 1137.

ADC (1998). Australian Dairy Corporation Dairy compendium 98. pp 11–13.

Anderson, K.L., Nagaraja T.G., Morrill J.L., Avery T.B., Glaitzer S.J. and Boyer J.E. (1987). Ruminal microbial development in conventionally or early-weaned calves. *Journal of Animal Science* 64, 1215–26.

Anon (1997). Climate Change: Australia's Second National Report under the United National Framework Convention on Climate Change. Commonwealth of Australia: Canberra.

Australian Greenhouse Office (1999) 1997 National Greenhouse Gas Inventory, Fact sheet 4. Agriculture. http://www.greenhouse.gov.au/ inventory/pdfs/ fs_agriculture.pdf.

Australian Greenhouse Office (2000). Greenhouse Sinks and the Kyoto Protocol— An Issues Paper. Australian Greenhouse Office, Canberra.

Baker, S.K. (1998). Biology of rumen methanogens, and stimulation of animal immunity. *Meeting the Kyoto Target. Implications for the Australian Livestock Industries.* P.J Reyenga. and S.M. Howden (Eds). Bureau of Rural Sciences, 29–37.

Barton, L., McLay, C.D.A., Schipper, L.A. and Smith, C.T. (1999). Annual denitrification rates in agricultural and forest soils: a review. *Australian Journal of Soil Research* 37, 1073–93.

Beever, D.E., Cammell, S.B., Sutton, J.D., Spooner, M.C., Haines, M.J. and Harland, J.I. (1989). The effect of concentrate type on energy utilisation in lactating cows. Proc 11th Symp. on Energy Metabolism EAAP Publication No. 43, 33.

Blaxter, K.L. (1989). Energy Metabolism in Animals and Man. Cambridge University Press, New York.

Chen, D., White, R.E., Chalk ,P.M., Heng, L.K., Helyar, K.R. and Fisher R. (1996). Measurement of gaseous N losses from grazed pastures. *Proceedings of the ASSI and NZSSS National Soils Conference, July 1996,* 41–2.

Cowan, R.T., Lowe, K.F., Ehrlich, W., Upton, P.C. and Bowdler, T.M. (1995). Nitrogen–fertilised grass in a subtropical dairy system 1. Effect of level of nitrogen fertiliser on pasture yield and soil chemical characteristics. *Australian Journal of Experimental Agriculture* 35, 125–35.

Crawford, A., Hirth, J. and Gourley, C. (1999). Maintaining Soil Quality of Grazed Pastures. Final Report SAS Project 02249/12134. Department of Natural Resources and Environment, Victoria.

Crutzen, P.J., Aselmann, I. and Seiler, W. (1986). Methane production by domestic animals, wild ruminants, other herbivorous fauna and humans. *Tellus* 38B, 271–84.

Czerkawski, J.W. (1969). Methane production in ruminants and its significance. *World Review of Nutrition and Dietetics* 11, 240–82.

Daly, J.J. (1984). Cattle need shade trees. Queensland Agricultural Journal Jan–Feb., 21–2.

Denmead, O.T., Freney, J.R. and Simpson, J.R. (1979). Studies of nitrous oxide emission from a grass sward. *Soil Science Society of America Journal* 43, 726–28.

Eckard, R.J., (1996). The Window of Opportunity for Nitrogen Fertiliser use in on Dairy Farms. Proceedings of the Large Herds Australia conference, Launceston, February 1996.

Eckard, R.J., (1998). A critical review of research on the nitrogen nutrition of dairy pastures in Victoria. ILFR, The University of Melbourne and Agriculture Victoria, Ellinbank, DNRE. October 1998. ISBN 0 7311 4270 5. pp 1–31.

Eckard, R.J., Chapman, D.F., White, R.E., Smith, A., Chen, D. and Durling, P.A. (2001). Nitrogen balances in high rainfall, temperate dairy pastures of south eastern Australia. Proceedings of the XIX International Grasslands Congress, São Pedro, Brazil, February 2001 (in press).

Eckard, R.J., Mckenzie, F.R. and Lane, P.R., (1997). The 'Pendulum Paradigm'—Trends in nitrogen fertiliser use on temperate grass/ clover pastures. Proceedings of the XVIII International Grasslands Congress, Canada June 1997.

Ellington, A., (1986). Nitrogen inputs and utilisation in leguminous pasture: a review of recent Australian literature. Department of Agriculture & Rural Affairs, Technical Report no. 128. AgDex 137/531. pp 19.

EPA (1993). Anthropogenic methane emissions in the United States: Estimates for 1990. Report to Congress. EPA 430–R–93–003.

Galbally, I.E. (1991). Land surface sinks for methane, nitrous oxides (NOx) and tropospheric zone over Australia. Ed: J.A. Taylor. Proceeding of a workshop on Greenhouse Gas Sinks in Australia. Centre for Resource and Environmental Studies, ANU. 35–39.

Galbally, I.E. (1992). Australian research requirements regarding exchange of greenhouse gases between the atmosphere and agricultural and natural systems. A report prepared for the National Greenhouse Advisory Committee.

Galbally, I.E., Fraser, P.J., Meyer, C.P. and Griffith, D.W.T. (1992). Biosphere-atmosphere exchange of trace gases over Australia. In: *Australia's renewable resources: sustainability and global change*. R.M. Gifford and M.M. Barson (Eds). International Geosphere Biosphere Programme (Australia). Bureau of Rural Resources Proceeding No. 12. pp117–50.

Galbally, I.E., Freney, J.R., Denmead, O.T. and Roy, C.R. (1980). Processes controlling the nitrogen cycle in the atmosphere over Australia. In: *Bio-geochemistry of ancient and modern environments*. P.M. Trudinger and M.A. Walter (Eds) Aust. Acad. of Sci. Canberra. pp. 319 –25.

Gibbs, M.J. and Johnson, D.E. (1994). Methane emissions from the digestive process of livestock. In: *International Anthropogenic Methane Emissions: Estimates for 1990*. US Environmental Protection Agency, Rep. EPA 230–R–93–010. Washington DC, U.S.A.

Hegarty, R.S. (1998). Reducing methane emissions through elimination of rumen protozoa. *Meeting the Kyoto Target. Implications for the Australian Livestock Industries.* P.J Reyenga. and S.M. Howden (Eds). Bureau of Rural Sciences, 55–61.

Hopkins, D. and Waters, C. (1999). Dairy waste: Application to pastures. Agnote no: AG0419, Agriculture Victoria. ISSN 1329–8062.

Howden, S.M. and O'Leary, G.J. (1997). Evaluating options to reduce greenhouse gas emissions from an Australian temperate wheat cropping system. *Environmental Software and Modelling* 12, 169–76.

Howden, S.M. and Reyenga, P.J. (1999). Methane emissions from Australian livestock. In: *Meeting the Kyoto Target. Implications for the Australian Livestock Industries.* P.J Reyenga. and S.M. Howden (Eds). Bureau of Rural Sciences, 55–61.

Interface (1994). Methane from Australian livestock: implications for the greenhouse effect. In: *Resource Sciences Interface*. Bureau of Resource Sciences. Commonwealth of Australia.

IPCC (1995). Greenhouse Gas Inventory Reference Manual. Vol. 3. Jointly published by UNEP, OECD, IEA, U.K. Meteorological Office, Bracknell, U.K.

Itabashi, H., Kobayashi, T. and Matsumoto, M. (1984). The effects of rumen ciliate protozoa on energy metabolism in some constituents in rumen fluid and blood plasma of goats. *Japanese Journal of Zootechnical Science* 55, 248–56.

JETACAR (1999). The use of antibiotics in food-producing animals: antibiotic-resistant bacteria in animals and humans. Report of the Joint Expert Advisory Committee on Antibiotic Resistance (JETACAR). Commonwealth Department of Health and Aged Care Commonwealth Department of Agriculture, Fisheries and Forestry—Australia. ISBN 1 86496 061 2

Joblin, K.N. (1998). Acetogens—opportunities and constraints to use in the rumen. In *Meeting the Kyoto Target. Implications for the Australian Livestock Industries.* P.J Reyenga. and S.M. Howden (Eds). Bureau of Rural Sciences, 39–44.

Johnson, K.A. and Johnson, D.E. (1995). Methane emissions from cattle. *Journal of Animal Science* 8, 2483–92.

Johnson, D.E., Hill, T.M., Ward, G.M., Johnson, K.A., Branine, M.E., Carmean, B.R. and Lodman, D.W. (1993). Principle factors varying methane emissions from ruminants and other animals. In: *Atmospheric Methane: Sources, Sinks and Role in Global Change.* M.A.K. Khalil (Ed.). NATO ADI Series Vol 113, Springer-Verlag, Berlin, Germany.

Judd, M.J., Kellier, F.M., Ulyatt, M.J., Lassey, K.R., Tate, K.R., Shelton, D.I., Harveys, M.J. and Walker, C.F. (1999). Net methane emissions from grazing sheep. *Global Change Biology* 5, 647–57.

Kerr D. (1993). Queensland Dairy Farm Study 1990–91. Summary Report-April 1993. Information Series Q193015. Queensland Department of Primary Industries.

Kurihara, M., Terada, F., Hunter, R.A., Nishida, T., and McCrabb, G.J. (1998). The effect of diet and liveweight gain on methane production in temperate and tropical beef cattle.

In: *Proceedings of the 8th World Conference on Animal Production.* Seoul National University, Seoul, Korea, Vol 1. pp364–5.

Lassey, K.R., Lowe, D.C., Manning, M.R. and Waghorn, G.C. (1992). A source inventory for atmospheric methane in New Zealand and its global perspective. *Journal of Geophysical Research* 97, 3751–3765.

Lassey, K.R., Ulyatt, M.J., Martin, R.J., Walker, C.F. and Shelton, D.I. (1997). Methane emissions measured directly from grazing livestock in New Zealand. *Atmospheric Environment* 31, 2905–2914.

Ledgard, S.F., Sprosen, M.S., Brier, G.J., Nemaia, E.K.K., and Clark, D.A. (1996). Nitrogen inputs and losses from New Zealand dairy farmlets, as affected by nitrogen fertiliser application: year one. *Plant and Soil.* 181, 65–9.

Leng, R.A. (1991). Improving Ruminant Production and Reducing Methane Emissions from Ruminants by Strategic Supplementation. USEPA report 400/1–91/004. Office of Air and Radiation: Washington.

Lowe, K. F., Bowdler, T.M., Casey, N., Fulkerson, W. J., Slack, K., Bryant, R., Robinson, B., Walker, R.G., Knight, R. I., Moody, P.W., Armour, J. D., and Best, E.K. (1996). Comparative productivity of annual irrigated ryegrass pasture receiving nitrogen fertiliser or grown with legumes. Final Report DAQ 092, Department of Primary Industries, Ipswich, Queensland. 73p.

Machmuller, A., Ossowski, D.A., Wanner, W. and Kreuzer, M. (1998). Potential of various fatty feeds to reduce methane release from rumen fermentation *in vitro* (rusitec). *Animal Feed Science and Technology* 71, 117–30.

Mann, L.K. (1986). Changes in soil carbon after cultivation. *Soil Science* 142, 279–288.

McKenzie, F. and Jacobs, J. (1997). Strategic use of nitrogen on grazed pastures. Proceedings of the Large Herds Australia Conference, Warrnambool, April 1997. pp. 98–111.

Moate, P.J. (1989). Defaunation increases milk yield of dairy cows. In: *Recent Advances in Animal Nutrition in Australia.* University of New England, Armidale, NSW.

Moate, P.J., Clarke, T., Davis, L.H. and Laby, R.H. (1997). Rumen gases and bloat in grazing dairy cows. *Journal of Agricultural Science, Cambridge* 129, 459–69.

Moe, P.W. and Tyrrell, H.F. (1979). Methane production in dairy cows. *Journal of Dairy Science* 62, 1583–86.

Morris, R.S. (1987). Health and management programs for individual livestock enterprises. In: *Proceedings of the 4th AAAP Animal Science Congress February 1–6, Hamilton, New Zealand.* T.F. Reardon. (Ed.) pp101–3.

Mundy, G.N. and Mason, W.K. (1989). The effect of flood irrigation and nitrogen source on the fate of nitrogen fertiliser applied to pasture. *Australian Journal Agricultural Research* 40, 107–16.

Murray, R.M., Bryant, A.M. and Leng, R.A. (1976) Rates of production of methane in the rumen and large intestine of sheep. *British Journal of Nutrition* 36, 1–14.

NGGI (1997). National Greenhouse Gas Inventory 1995. National Greenhouse Gas Inventory Committee, Canberra.

NGGI (1998a). National Greenhouse Gas Inventory 1996. National Greenhouse Gas Inventory Committee, Canberra.

NGGI (1998b). Agriculture, Workbook for Livestock. Workbook 6.1 with Supplements. National Greenhouse Gas Inventory Committee. 1998.

Oades, J.M. (1988). The retention of organic matter in soils. *Biogeochemistry* 5, 35–70.

Packrou, N., Dillon, P. & Stanger, G. (1997). Impact of pastoral land use on groundwater quality. Final Report LWRRDC Project No. CCW9, Water Industry Research Award. Centre for Groundwater Studies Report no 64, SA.

Pook, E.W. and Costin A.B. (1971). Root distribution and soil moisture studies in some perennial ryegrass and phalaris pastures on the southern tablelands, south-eastern Australia. CSIRO Division of Plant Industry, *Field Station Records* 10, 59–72.

Riffkin P., Quigley P. & Cameron F. (1997). Improving the White Clover Feedbase by Optimising Nitrogen Fixation. Final Report DAV 313, Dairy Research & Development Corporation.

Rumpler, W.V., Johnson D.E. and Bates D.B. (1986). The effect of high dietary cation concentration on methanogenesis by steers fed diets with and without ionophores. *Journal of Animal Science* 62, 1737–41.

Van Nevel, C.J. and Demeyer, D.I. (1995). Feed additives and other interventions for decreasing methane emissions. In: *Biotechnology and animal feeds and feeding.* R.J. Wallace and A. Chesson (Eds). VCH Publishers Inc. New York. pp 329–49.

Wedegaertner, T.C. and Johnson, D.E. (1983). Monensin effects on digestibility, methanogenesis and heat increment of a cracked corn-silage diet fed to steers. *Journal of Animal Science* 57,168–77.

Weier, K.L., McEwan, C.W., Vallis, I., Catchpoole, V.R. and Myers, R.J. (1996). Potential for biological denitrification of fertiliser nitrogen in sugarcane soils. *Australian Journal of Agricultural Research* 47, 67–79.

Whitehead, D.C. (1995). Legumes: Biological nitrogen fixation and interaction with grasses. In: *Grassland Nitrogen*. pp 48–51. CAB International, Wallingford, UK.

Whitelaw, F.G., Eadie, J.M., Bruce, L.A. and Shand, W.J. (1984). Methane formation in faunated and ciliate-free cattle and its relationship with rumen volatile fatty acid proportions. *British Journal of Nutrition* 52, 261–75.

Carbon Sequestration in Australia's Rangelands

Brendon Baker, Guy Barnett and Mark Howden,

CSIRO Wildlife and Ecology, Canberra

Australia it is a signatory to the Framework Convention on Climate Change and has signed the Kyoto Protocol. Article 3.4 of the Protocol opens the way for discussions of 'additional activities' to be included in commitments to limit emissions in the commitment period from 2008 to 2012. This area of 'additional activities' is expected to be a key part of negotiations later in this year at the Sixth Conference of the Parties to the United Nations Framework Convention on Climate Change (COP 6). The potential size of sinks and sources associated with 'additional activities' in Australia's rangelands is a key reason why the CRC for Greenhouse Accounting is involved in assessing this component.

The CRC for Greenhouse Accounting consider that in terms of meeting Kyoto Protocol commitments to reduce greenhouse gas emissions, the role of carbon sinks will become a major focus of climate change negotiations. According to Ash *et al.* (1996):

> *"Emission reduction measures taken under the National Greenhouse Response Strategy (Australia 1992), whilst reducing emissions, are estimated to leave emissions at 7 % above 1990 levels by the year 2000. Hence there is a need to identify approaches to further reduce emissions or to increase biospheric sinks for carbon dioxide." p19*

As such, the purpose of this paper is to examine the carbon sequestration potential of the ecosystems that comprise Australia's rangelands and to inform policy makers on the potential effect of rangeland management activities on carbon sinks and sources. Four broad management activities are considered: stocking rates, fire, woody vegetation and landscape rehabilitation.

Under Article 3.4 of the Kyoto Protocol, the effects of activities related to land use, land use change and forestry may be able to be included as part of a country's carbon accounting system. The potential global impact of 'additional activities' to sequester carbon is estimated to be up to 0.52 Gt C per year for developed countries (Loveland and Belward, 1997).

Including major land use activities as adjustments to the assigned emissions under the first commitment period of the Kyoto Protocol could potentially reduce the degree to which many countries may need to alter energy use and energy production technology. However, in the Australian case, the inclusion of woody vegetation in the 1990 account of emissions may decrease the 1990 net account, and hence increase the 2008–2012 abatement task.

What is carbon sequestration?

Carbon sequestration refers to any activity that removes CO_2 from the atmosphere by 'locking up' carbon in a solid state (Anon, 1998). Trees and other vegetation sequester carbon through the process of photosynthesis, which involves the conversion of atmospheric CO_2 into complex carbon molecules used for plant growth, releasing oxygen and water in the process. Soils and water bodies are also able to sequester carbon.

Much of the technical research into carbon sequestration has been conducted in the densely forested ecosystems of the northern hemisphere. But unfortunately, extrapolation of results is not appropriate for most Australian forests, let alone the functionally different ecosystems of Australia's rangelands. Of the work that has been undertaken in Australia, most has been concentrated on the open forest ecosystems rather than the grass and shrub dominated rangeland ecosystems.

The rate at which ecosystems can accumulate carbon, the ultimate size of the newly-stored carbon pools, and the rate at which the carbon can be lost again under altered circumstances, are all dependant on the form of the newly-formed carbon, the magnitude of the land use or management change, the inherent biological productivity of a site, and the type and depth of soil. The capacity to store carbon through land use change is finite, both because the land area available for this purpose has competing uses and is limited by upper carbon pool limits.

It is important to note the distinction between a carbon 'store' and a carbon 'sink'. For vegetation or soil to be a carbon 'sink' it must be actively sequestering atmospheric CO_2. Yet rates of carbon sequestration vary depending on several key factors, including climate, topography, soil characteristics, vegetation species and age composition, and different stages in the vegetative growth cycle (FTS, AACM International and CCI, 1998). In assessing changes for carbon accounting under the Kyoto Protocol, changes in the rate of storage over a given period is important.

Given the above, it is critical to remember that even if vegetation is not harvested, it will eventually reach a steady or declining state of carbon storage as plants mature, die, and decompose, potentially returning CO_2 back into the atmosphere (Shea, 1998). Thus woody vegetation may be no longer storing carbon when carbon sequestration is equal to carbon emissions through plant decay or soil carbon release. Woody ecosystems may also be a source of carbon when emissions are greater than rates of sequestration, as happens during disturbances such as harvesting, vegetation clearing, fire, storms or pests. For instance, it has been estimated that 30% of the total of global CO_2 emissions are a result of vegetation loss and degradation (Anon 1998).

What are the stores, sources and sinks of carbon in the rangelands?

Carbon storage rates per hectare in arid and semi-arid systems are low compared to other biomes, but rangeland systems cover very large areas. As noted by Ash *et al.* (1996), rangelands occupy 70% of Australia's land mass and through better management could provide a potentially important carbon sink.

There are two main ways that rangeland activities can contribute to reducing the carbon emissions of Australia:

1. Increasing carbon sequestration: Carbon dioxide is stored in soil organic matter. Soil scientists have suggested that this store could contribute significantly in reducing net CO_2 emissions. More importantly, the role of soil biomass and above ground biomass is gaining increasing attention as an effective carbon store. The production of grass, woody vegetation and litter in semiarid lands increases the storage of carbon, however this vegetative method has variable stabilities associated with persistence and monitoring.
2. Decrease associated emissions: Methane (CH_4) is released by livestock and also native fauna. The role of native fauna, such as kangaroos and termites, is not discussed, because only anthropogenic issues are dealt with in the framework convention on climate change and the Kyoto Protocol for inventory issues. However the role of natural processes in contributing to net carbon flows should not be forgotten. To a lesser extent, nitrous oxide (N_2O) is released by livestock.

The main ways to implement carbon sequestration is by increasing carbon inputs into the system and decreasing carbon outputs.

CARBON STORES IN THE RANGELANDS

The Australian rangelands occupy around 6,000,000,000 hectares. Calculations based on Squires and Glenn (1996) suggest that this land could have a present carbon pool of about 48 Gt of carbon to a depth of one meter. As Squires and Glenn (1996) note:

> *"The soil store of [carbon] in these rangeland ecosystems is a very important pool since it is stabilized for hundreds to thousands of years, and forms the bulk of the rangeland [carbon] pool." p532*

CARBON SOURCES IN THE RANGELANDS

Ash *et al.* (1996) investigated the correlation between poor range condition and reduced soil organic carbon in northern Australia. Degraded or dysfunctional lands fail to maintain the carbon pool due to overutilisation of plant biomass and soil erosion. This leads to the release of CO_2 and associated emissions. Thus, while a majority of rangeland landscapes are in a degraded state, the rangelands will be a net carbon source.

Landscape dysfunction tends to selectively displace the organic matter of surface soils and thus can significantly reduce carbon stocks at a particular location (Freudenberger *et al.*, 1997).

Poor grazing strategies and frequent burning reduce plant cover significantly and may lead to increased soil erosion (Baker *et al.*, 1999), and consequent carbon loss (Ash *et al.*, 1996). Burning represents the rapid transfer of carbon from vegetation, litter and to a lesser extent soil to the atmosphere. Fire transfers a large portion of the above ground carbon into the atmosphere as well as smaller proportions of below ground carbon. Associated emissions also occur. Increasing fire frequency over time tends to reduce soil carbon stores (Jones *et al.*, 1990; Knapp *et al.*, 1998).

A relatively recent assessment of the condition of northern Australia's grazing lands (Tothill and Gillies, 1992) showed that significant areas have either deteriorated or been degraded through overgrazing and/or woody weed invasion.

The dominant trend in condition, and therefore the potential for carbon sequestration, is linked to management.

Most management activities have benefits in terms of carbon sequestration, however some options for sustainable rangeland management may in fact be negative for carbon sequestration. For example, fire regimes to control woody 'weeds'; or relatively high stocking rates that are ecologically sustainable, yet contribute high associated emissions.

Additionally, many activities that on the surface seem to increase carbon sequestration have hidden carbon "costs" in terms of greater emissions of CO_2 into the atmosphere (Schlesinger, 1999).

CARBON SINKS IN THE RANGELANDS

Net carbon uptake can be achieved via reversion of degrading processes (overgrazing and erosion), which also assists agricultural productivity. Increased carbon storage can also occur though enhanced woody plant productivity. Grazing and fire offer rangeland graziers the most effective tools for sustainable (economic and ecological) management (Abel and Ryan, 1996). Changes in stocking rates and fire regimes can be used to manipulate soil and biomass carbon pools. However, the 'woody weed' problem associated with infrequent fire regimes has adverse impacts on grazing productivity (Hodgkinson, 1983; Woody Weeds Task Force 1990) and also biodiversity (Landsberg *et al.*, 1997).

The soil carbon store in Australia's rangelands is presently estimated at about 48 Gt of carbon to a depth of one meter. More importantly, Australia's rangelands could be a sequestering 0.07 Gt of carbon per year (calculated from Squires and Glenn, 1996).

Landscape rehabilitation and erosion control enhances the sustainable use of rangelands, and hence is positive for carbon sequestration.

Restoring vegetation results in increased biomass, litter and soil carbon pools. Vegetation also prevents soil erosion due to wind and water, which removes carbon from an ecosystem and may be oxidised to CO_2.

Soil conservation procedures that restore landscape function (or patchiness), such as the use of shrub branches lain across a slope, can increase soil carbon (Tongway and Ludwig, 1996). Activities that improve landscape function are likely to substantially increase the carbon sequestration ability of soils and parcels of land.

Lands that are put aside for environmental reasons, including National Parks, can act as a carbon sink. However, the rate of sequestration will decline over a period of about 50 years (H. Shugart pers. comm).

The protection of land through Government programs or private mechanisms (see Binning, 1997) with the aim of restoration can increase the above and below ground biomass and soil carbon.

Store	Potential Size	Source
Soil (top 1 metre)	48 Gt for entire rangeland	Ash *et al.*, 1996
Plant Biomass: grasses	0.88 t per ha in the semi-arid tropics	Cameron and Ross 1996
Source	**Potential Size**	**Source**
Soil degradation	Unknown—Net loss	Ash *et al.*, 1996; Squires and Glenn 1996
Clearing and conversion to pasture	4–6 t per ha (over 15 years) (0.3–0.4 t per ha per year)	Howden *et al.*, 1994
Fire	10 to 50 percent of the carbon pool	Gifford and Howden (unpublished)
Overgrazing	7 t per ha (over 15 years)	(0.5 t per ha per year) Howden *et al.*, 1994
Sink	**Potential Size**	**Source**
Soil (top 1 metre)	0.07 Gt for entire rangeland per year	Ash *et al.*, 1996
Thickening of woody mulga stands (not maintained for productive grazing)	1000 kg per ha per year (1 t per ha per year)	Moore *et al.*, 1997
Charcoal as a result of fire	Marginal (eg, 8 grams per kg soil) but stable	Skjemstad *et al.*, 1996

Table 1: The potential size (carbon equivalents) of stores, sources and sinks for selected examples in Australia's rangelands.

What is the potential size of carbon stores, sources and sinks?

The following examples are taken from various sources and refer to Australia's rangelands. These figures may be low compared to similar figures from forested and other agricultural biomes, however the extensive nature of Australia's rangelands and the adaptability and willingness of rangeland enterprises to initiate carbon sequestration activities, makes them more impressive.

What effect will management activities have on stores, sources and sinks?

It is important to recognise that most activities for increasing carbon storage in the rangelands are not independent of each other, but rather are interrelated through the dominant biophysical processes that operate in rangeland ecosystems.

The potential of various rangeland activities, such as stocking rates, fire regimes, woody vegetation management and rehabilitation, in storing carbon and reducing associated emissions is linked to the sustainable habitation of the rangelands (Abel and Ryan, 1996). Various management activities have important impacts on the condition of the rangeland.

There is a constant threat of detrimental rangeland change due to poor grazing strategies, misuse and inappropriate management. Such change would have negative consequences for carbon sequestration.

MANAGING STOCK AND TOTAL GRAZING PRESSURE

Degradation of vegetation and soil in the rangelands is usually ascribed to grazing by domestic sheep, although in many rangeland areas feral and native herbivores contribute significantly to total grazing pressure. Grazing pressure fluctuates due to climate, vegetation fluxes, the impact of pests, grazier management and rangeland type.

The effects of heavy grazing by sheep, kangaroos, goats and rabbits are compounded by drought. Changes generally include the replacement of palatable perennial species by annual and/or less desirable plants, reductions in plant cover, accelerated erosion, and an increase in effective aridity (loss of productivity and decline in soil fertility).

One of the most significant impacts of total grazing pressure on vegetation is the loss of perennial grass species. Perennial vegetation provides long-term protective soil cover and is important for the capture of ecosystem resources.

To some extent, the judgment of vegetation degradation or change is subjective, depending on each user's land management objectives. The objective of graziers is to maintain pastures that will sustain the growth and reproduction of their stock. Overgrazing (where stocking rates exceed the available biomass (is an inherent risk of pastoralism. Even when total grazing pressure is low, stress during dry periods and drought can induce plant death and soil degradation.

Current rangeland vegetation communities can be managed adequately to avoid vegetation and soil degradation. Under appropriate management regimes, degradation can be halted and controlled, although the conservation of biodiversity is less assured (Baker *et al.*, 1999).

The management activities employed by graziers could significantly affect carbon storage through reducing carbon loss during degradation processes or increasing carbon inputs.

Total grazing pressure influences carbon and nutrient cycling, as well as vegetation characteristics of rangeland ecosystems. Grazing results in some of the plant carbon being utilised by animals—some is converted to weight gain, some is emitted as CO_2 and CH_4, and 25–50% is returned in wastes to the environment.

In general, where grazing is managed to maintain plant productivity, soil carbon is also maintained or increased (Conant *et al.*, 2000). However, overgrazing can significantly decrease primary production and cause landscape degradation. The consequences for soil carbon are then negative.

Overgrazing is the single greatest cause of degradation in rangelands (Oldeman *et al.*,1991, Baker *et al.*, 1999). This human-induced factor influences carbon storage by removal of biomass and nutrients, much of which is lost to the soil carbon pool. Overgrazing also leads to soil and vegetation degradation. Often, extensive heavy grazing practices result in decreases in carbon pools stored in biomass and soil (Ash *et al.*, 1996; McIntosh *et al.*, 1997). Consequently, improved grazing management, such as optimising stock numbers and tactical grazing (Hodgkinson, 1993), will result in increases to the carbon pool. There are also productivity and environmental benefits of adopting suitable grazing strategies (Freudenberger *et al.*, 1997).

Adoption of more appropriate grazing practices that track climate variability and change is important for the sustainable habitation of Australia's rangelands, reducing the risk of degradation (Abel and Ryan, 1996; Baker *et al.*, 1999) and hence increasing the potential of carbon sequestration.

Ash *et al.* (1996) suggest that the adoption of reduced stocking rates to increase perennial grass cover could sequester 315 million t of carbon per year in the top 10cm of soil over 30 years. The economic benefits of such a management change are also positive (Stafford-Smith *et al.*, 1999; Freudenberger *et al.,* 1997).

The introduction of grass and legume species is possible in some rangeland ecosystems. This activity can significantly increase production where previously nutrient-limited vegetation dominated (Conant *et al.,* 2000). Also, in some grazing operations in the semi-arid tropics, fertiliser application can increase the average above ground grass biomass of 0.88 t C per ha to 3 t C per ha (Cameron and Ross, 1996).

Overall, total grazing management strategies will determine if particular grazing enterprises could be considered a carbon sink. Perhaps in some rangelands biomes, such as belah/bluebush communities and mulga lands there is potential to have a net store of carbon in the woody vegetation under low grazing regimes.

Wilson (1979) suggests that sheep production in belah/bluebush communities is favoured by continuous grazing over the whole property at about 0.25 sheep/hectare (1 sheep per 4 hectares). This relatively low rate is due to the low overall protein content of the vegetation. The few important perennial species are unpalatable and are not grazed until forage is sparse and so provide a key carbon store within a continuous grazed system.

FIRE MANAGEMENT

The most practical tools available to rangeland managers are grazing and fire. Fire is often an essential tool for controlling woody vegetation, removing dead biomass, clearing, stimulating grass growth and palatability, hunting and controlling wildfires and pests (Harrington *et al.*, 1984). Fire is also important for some species to propagate seed. Prior to European settlement of Australia, humans were the main causes of frequent fire. Natural fires occur due to lightning strike, particularly during drought or under poor fire management regimes. The importance of fire makes its carbon sequestration potential difficult to assess.

Changes in fire regimes, in combination with grazing, are often associated with an increase in woody vegetation, resulting in large increases to the carbon store (Archer *et al.*, 1995; Burrows *et al.*, 1999). However, the usefulness of areas with increased woody biomass for traditional purposes, including grazing, often declines. Fire will continue to play a major role in sustainable grazing operations. Also, in many rangeland ecosystems, specific fauna and flora are fire-dependant and removing fires will result in a loss of biodiversity (Noble, 1998).

Charcoal generated by fires can constitute 8g C per kilogram of soil, creating a very stable store (Skjemstad *et al.*, 1996). A small fraction of the biomass consumed by fire is also converted to black carbon, proving another stable store that is virtually imperious to decay (Stallard, 1998).

Reducing fire frequency does retain carbon stocks, however these stocks are very vulnerable to future unplanned fires.

The management role of fire is likely to make it a poor option for activities that act as a carbon sink. However, optimising fire timing may increase biomass in some systems while also increasing productivity (Cox and Morton, 1986).

MANAGEMENT AND CONTROL OF WOODY VEGETATION

Unpalatable native shrubs that have increased in number in the arid and semi-arid rangelands are often referred to as ‘woody weeds'. They have encroached on large areas of formerly open lands and are now considered by many to be the major environmental problem facing landholders in semi-arid environments.

Since European settlement, management of the rangelands has induced high grazing pressure and fire exclusion, thereby allowing shrubs to survive to become adults. Once established, dense areas of shrubs take up moisture and nutrients, preventing growth of vigorous perennial grasses. Consequently, shrub encroachment becomes self-perpetuating (Noble, 1998).

It is clear that unpalatable woody shrubs are a production problem, but it is less clear that they are an ecological problem. The effects of woody weed encroachment on biodiversity values have not been studied. Effects are likely to be positive for some taxa and negative for others (Baker *et al.*,1999).

In some rangeland landscapes, changes in species composition under grazing encourage woody vegetation, thus increasing carbon levels in the surface soil. However, these species are associated with reduced grazing productivity. Moore *et al.* (1997) have suggested that thickening of woody mulga stands, which could consequently not be maintained for productive grazing, could sequester one ton of carbon per year.

Where woody species are dominant due to natural patterns or changed management conditions, fire, physical removal, or sometimes heavy grazing will reduce the previously sequestered carbon.

LANDSCAPE REHABILITATION

Restoring the productivity of over-utilised rangelands is difficult, particularly from an economic perspective (Ludwig and Tongway, 1998). Rehabilitation at local scales has been well researched with excellent results and many rangeland landscapes have been successfully rehabilitated.

Perhaps the most successful form of rehabilitation has promoted the rebuilding of 'patches' to restore landscape function. Tongway and Ludwig (1996) described this rehabilitation technique and reported that it successfully restores landscape function in degraded woodlands. Tongway and Ludwig (1998) suggested that in some vegetation communities landscape function can be restored by rebuilding patches (or log mounds) that serve to capture and retain soil water and nutrients in runoff, and organic matter in wind-borne litter, rather than have these vital ecosystem resources lost from the system. During a ten-year experiment, they showed that the soil productive potential remains even after considerable degradation has occurred, and that this productive potential can be restored using simple and inexpensive techniques.

The recovery of perennial grasses can also be encouraged using Tongway and Ludwig's (1996) approach. Patches capture ecosystem resources, including seeds and nutrients, and provide shelter for the recruitment of perennial grasses (Freudenberger *et al.*, 1997).

How might management activities affect the net carbon store?

Traditional management of grazing operations are unlikely to provide a net CO_2 benefit, particularly under difficult economic conditions. Even under effective grazing strategies that maintain perennial grass species and a suite of woody vegetation, the role of vegetation as a carbon store will fluctuate greatly over time as climatic conditions vary and plants are consumed by stock.

Some rangelands biomes, such as belah/bluebush communities and mulga lands have potential to provide a net store of carbon held in the woody vegetation under low grazing regimes. However, the nature of rangeland vegetation makes the total carbon store difficult to estimate, particularly over time. Landscapes may appear stable for moderate periods of time (10 years), but may undergo rapid degradation, and hence carbon loss, under drought or poor management conditions.

Long-lived species dominate the rangelands, providing stability to the system by surviving for long periods at low density; for this reason, a lack of regeneration due to management changes may go unnoticed until aged plants begin to die. Short-lived plants are important as the basis of plant and animal productivity. Populations of short-lived plants responding to rainfall are resilient to pastoral management.

Currently, carbon sequestration is a low priority for graziers because carbon as a commodity has no value. Graziers are interested in the role of carbon as a commodity, and making carbon sequestration an additional and specific activity of rangeland enterprises. The main limitations on this activity relate to accurate monitoring and the transaction costs associated with monitoring and trading (M. Howden, pers. comm.).

The concept of carbon credits carries less weight in the rangelands because the estimated value of carbon held in woody and grass biomass is likely to be low per hectare, less certain and less stable compared to the value of carbon stored in timber grown in other biomes.

However, land managers in the rangelands may be convinced to adopt a 'no regrets' approach, which would benefits land owners ecologically and potentially economically.

A change in management on selected parcels of land to increase woody vegetation would cause accumulating production losses, however may offer ecological benefit, including land restoration, biodiversity opportunities and carbon sequestration. These ecological benefits have the potential to attract future economic gains through carbon credits, recreation value or bioprospecting potential. Additionally, long-term landscape function may be restored, perhaps leading to increased farm productivity (Freudenberger *et al.*, 1997).

Monitoring sinks and sources

In most rangeland environments, surface soil condition is correlated with pasture condition (Ash *et al.*, 1996), which could be assessed using field measurements such as Landscape

Function Analysis (LFA) (Tongway and Hindley, 1995) or remote sensing (Bastin *et al.,* 1998).

Rapid assessments of pasture and woody biomass at the site scale are well established (Tothill *et al.,* 1992). LFA may be a useful link for this data in assessing soil carbon stores and sinks.

Ground-based monitoring systems have the potential to be linked with remotely sensed data. Pickup (1989) successfully used remotely sensed data to interpret range condition, but remote sensing (from satellites or aerial photography) has yet to prove itself as an effective rangeland monitoring tool (Arzani *et al*., 1996; Gardiner *et al.,* 1998).

While the science is well advanced, uptake by State agencies has been low. Future monitoring programs should focus on the practical application of remote sensing in an appropriate spatial framework.

There is some capacity to model carbon changes provided that accurate information on factors such as climate, soils and stocking rates are known. There are many computer-based models, such as GRASP (McKeon *et al*., 1992) and the Century model (Parton *et al*.,1988).

The application of carbon accounting in agricultural systems has been suggested as part of the Greenhouse Challenge Program (FTS, AACM International and CCI 1998), however the application of quantitative methods for measuring carbon sequestration in rangeland operations has not been adequately studied.

Complexities and scientific uncertainty

There are many issues that have complex consequences for carbon sequestration in the rangelands. The processes and time scales of carbon storage are the most difficult to assess. Ecosystem carbon stocks are determined by balancing inputs, via photosynthesis and organic matter imports, with losses through plant, animal and decomposer respiration, fire, harvests and other exports. These processes operate at several spatial and temporal scales.

Additionally, the rate of accumulation of carbon from a change in land use or management activity cannot be sustained indefinitely. Eventually, input and loss rates come into balance and carbon stocks will approach a steady state (Odum, 1969; Johnson, 1995). Rates of carbon sequestration are highest soon after adoption of a new practice, but subside over time.

Importantly for Australia's rangeland, the continued accumulation of carbon is dependent on appropriate management practices and climatic regimes. The termination of carbon sequestration activities due to economic feasibility can lead to the partial or complete loss of previous gains. Also, the long term pattern of carbon accumulation is responsive to changes in climatic conditions. For example, stored carbon may be susceptible to loss during drought periods due to accelerated decomposition from heat. Under higher temperatures associated with the enhanced greenhouse effect itself, carbon stores may be lost (Jenkinson, 1991; Trumbore *et al*., 1996). As stocks increase so does the risk of potential future losses due to poor management or unfavourable climatic conditions.

The response of soil carbon stocks to grazing intensity varies for different rangeland ecosystems. The basic processes that change carbon stocks under grazing activities are well understood (Howden *et al*., 1994). However, the magnitude of carbon fluxes as a function of grazing intensity has received little attention and is less certain.

The size of inert charcoal pools and pool dynamics is poorly known.

Much of the potential of carbon sequestration through improving landscape function depends upon an increase in biomass productivity (but not livestock productivity), which means a change in current grazing or harvesting practices. It is clear that improved grazing management systems which reduce erosion also lead to increases in the soil carbon stock (Tongway and Ludwig, 1996).

The use of fossil fuels to establish and maintain carbon sequestration activities must also be considered. For example, a management activity may emit the equivalent or more CO_2 from energy production than it sequesters. Initial energy requirements for erosion control techniques would be offset by long term benefits of erosion control.

Conclusions

Australia is a signatory to the framework convention on climate change and has signed the Kyoto Protocol. Article 3.4 of the Protocol opens the way for discussions of 'additional activities' to be included in commitments to limit emissions in the commitment period from 2008 to 2012. This area of 'additional activities' is expected to be a key part of negotiations later in this year at COP 6.

The use of rangelands for carbon sequestration stored within woody shrubs and organic matter in soils has significant potential. However, at present on degraded rangelands, release of carbon due to over-utilisation of plant production ensures that rangelands are a continuing source for carbon emissions (Squires and Glenn, 1996).

Carbon storage rates per hectare in arid and semi-arid systems are low compared to other biomes, but rangeland systems cover very large areas. But, degraded or dysfunctional lands fail to maintain the carbon pool due to overutilisation of plant biomass and soil erosion. This leads to the release of CO_2 and associated emissions. Thus, while a majority of the landscapes are in a degraded state, the rangelands will be a net carbon source.

Therefore, the dominant trend in condition, and therefore the potential for carbon sequestration, is linked to management. Most management activities have benefits in terms of carbon sequestration.

There are many ancillary benefits associated with erosion control and landscape rehabilitation. These include increased productivity, reduced methane emissions, biodiversity benefits, improvements to water quality, and increased economic returns from land. Opportunities for protection and landscape restoration have environmental and recreational benefits.

Traditional management of grazing operations are unlikely to provide a net CO_2 benefit, particularly under difficult economic conditions. Even under effective grazing strategies that maintain perennial grass species and a suite of woody vegetation, the role of vegetation as a carbon store will fluctuate greatly over time as climatic conditions vary and plants are consumed by stock.

The potential of various rangeland activities, such as stocking rates, fire regimes, woody vegetation management and rehabilitation, in storing carbon and reducing associated emissions is linked to the sustainable habitation of the rangelands (Abel and Ryan, 1996). Various management activities have important impacts on the condition of the rangeland.

There is a constant threat of detrimental rangeland change due to poor grazing strategies, misuse and inappropriate management. Such change would have negative consequences for carbon sequestration.

The role of Australia's rangelands in assisting the global human-induced climate change issue through carbon sequestration is linked to regional management. Many variables, uncertainty and larger issues complicate scenarios of climate change and the ability of additional activities to sequester carbon and account for that carbon. However, despite uncertainties, there is a recognised need for positive actions to reduce human pressures on the environment, including our contribution to greenhouse emissions. Rangeland enterprises could be in a strong position to contribute to greenhouse solutions though carbon sequestration schemes.

It is clear that the carbon sequestration potential of the rangelands is high. It is also clear that this potential is linked to management and the ability to adapt to ecological and economic change. Ultimately, this is why we have to use our best judgement, guided by the current state of science, to determine what the most appropriate response to global warming should be.

References

Abel, N. and Ryan, S. (Eds) 1996. Sustainable Habitation in the Rangelands. *Proceedings of a Fenner Conference on the Environment,* Canberra, 29–30 Oct 1996.

Anon. (1998). Can't see the Carbon for the Trees. Australian Energy News. 18 April 1998, http://www.ecosecurities.com/article1.htm, accessed May 2000.

Archer, S., Schimel, D.S. and Holland, E.A. (1995). Mechanisms of shrubland expansion: land use, climate or CO_2? *Climate change* 29, pp91–99.

Arzani H., King G.W., West N.E. (1996). An examination of techniques for estimating rangeland production in semiarid and arid NSW. In: *Rangelands in a Sustainable Biosphere. Proceedings of the Fifth International Rangeland Congress, Salt Lake City, Utah, USA, 23–28 July, 1995. Volume 1: Contributed Presentations.*(Society for Range Management, Denver, USA).

Ash, A.J., Howden, S.M., McIvor, J.G. and West, N.E. (1996). Improved rangeland management and its implications for carbon sequestration. In: *Proceedings of the Fifth International Rangeland Congress.* Salt Lake City, Utah, 23–28 July, 1995. Volume 1, pp19–20.

Baker, B. L., Barnett, G., Doherty, M. D., Kearns, A. J., Sarre, A., Gordon, A., Mazzer, and T., Tatnell, B. (1999). "Study D: Biophysical—Status and trends of the land & water systems of the Western Division", *Western Lands Review Consultant Studies,* (Western Lands Review, NSW).

Bastin, G.N., Tynan, R.W. and Chewings, V.H. (1998). Implementing satellite-based grazing gradient methods for rangeland assessment in South Australia. *Rangeland Journal,* 20, 61–76.

Binning, C. (1997). 'Beyond reserves: Options for achieving nature conservation objectives in rural landscapes'. In: Klomp, N. and Lunt, I (eds) *Frontiers in Ecology: Building the links.* Elsevier Science: Oxford.

Bridge, B.J., Mott, J.J. and Hartigan, R.J. (1983). The formation of degraded areas in the dry savanna woodlands of northern Australia. *Australian Journal of Soil Research* 21, 91–104.

Burrows, W.H., Compton, J.F., Hoffmann, M.B., Back, P.V. and Tait, L.J. (1999). Allometric relationships and community biomass estimates for some dominant eucalypts and associated species in central Queensland woodlands. *Australian Journal of Botany* (submitted).

Cameron, A.G. and Ross, B.J. (1996). Fertiliser responses of established grass pastures in the Northern Territory. *Tropical Grasslands* 30, 426–29.

Conant, R.T., Paustian, K. and Elliot, E.T. (2000). Grassland management and conversion into grassland. *Ecological Applications* (accepted).

Cox, J.R. and Morton, H.L. (1986). The effects of seasonal fire on live biomass and standing crop in a big sacaton grassland. In: Rangelands: A resource under siege. Proceedings of the 2nd International Rangeland Congress, Adelaide. Australia Academy of Science, Canberra, pp 596–97.

Dregne, H., Kassa, M. and Rzanov, B. (1991). A new assessment of the world status of desertification. *Desertification Control Bulletin* 20, pp6–18.

Freudenberger, D., J. Noble, K. Hodgkinson 1997. Management for Production and Conservation Goals in Rangelands. In: Ludwig JA, Tongway DJ, D. Freudenberger, Noble J, Hodgkinson KC. *Landscape Ecology. Function and Management—Principles From Australia's Rangelands.* (CSIRO Publishing, Melbourne, pp. 93–106).

Forestry Technical Services, AACM International and Clean Commodities Inc. (1998). Greenhouse Challenge Vegetation Sinks Workbook—Quantifying Carbon Sequestration in Vegetation Management Projects. Australian Greenhouse Office, Canberra

Gardiner DB, Tupper GJ, Dundeon GS 1998. A quantitative appraisal of woody shrub encroachment in western New South Wales. *Rangeland Journal* 20, 26–40.

Gifford, R.M. and Howden, M. (unpublished). 'Vegetation thickening in an ecological perspective: Significane to national greenhouse gas inventories and mitigation policies'.

Harrington G.N., D. Mills, A. Pressland and Hodgkinson K.C. (1984). Semi-arid woodlands. In: Harrington, A., A. Pressland, Hodgkinson KC (Ed). *Management of Australia's Rangelands.* (CSIRO, Melbourne, pp. 189–207).

Hodgkinson, K. (1983). Prescribed Burning for Woody Weed Control in Semi-arid Woodlands. In: *Proceedings of Workshop on Woody Weeds of Northern Summer Rainfall Areas. Rockhampton, October, 1983*, Haseler, W. (Convenor) (Qld Department of Lands, Rockhampton, Qld).

Hodgkinson, K. (1993). Identification of critical thresholds for opportunistic management of rangeland vegetation In: *Proceedings of the Fourth International Rangeland Congress,* Gaston, A., M. Kernick, H. Le Houerou (Ed.), Montpellier, France, 22 April 1993–26 April 1993, (Service Central d'Information Scientifique et Technique, Montpellier).

Howden, S.M., White, D.H., McKeon, G.M., Scanlan, J.C. and Carter, J.O. (1994). Methods for exploring management options to reduce greenhouse gas emissions from tropical grazing systems. *Climatic Change* 27, 49–70.

Jenkinson, D.S. (1991). The Rothamsted long-term experiments: are they still of use? *Agronomy Journal* 83, 2–12.

Johnson, M.G. (1995). The role of soil management in sequestering soil carbon. In: Lal, R., Limble, J., Levine, E. and Stewart, B.A. (Eds) *Soil Management and Greenhouse Effect.* Lewis: Boca Raton. pp 351–363.

Jones, C.L., Smithers, N.L., Scholes, M.C. and Scholes, R.C. (1990). The effect of fire frequency on the organic components of a basaltic soil in the Kruger National Park. *South African Journal of Plant and Soil,* 7, 236–38.

Kirschbaum, M. U. F. (1996). The Carbon Sequestration Potential of Tree Plantations in Australia. In: K.G. Eldrige; M.P. Crowe, and K.M. Old, (Eds). Environmental Management: The role of Eucalypts and Other Fast Growing Species. Proceedings of the Joint Australian/Japanese Workshop held in Australia. p77–89. CSIRO Forestry & Forest Products, Canberra.

Knapp, A.K., Conrad, S.L. and Blair, J.M. (1998). Determinants of soil CO2 flux from a sub-humid grassland: effect of fire and fire history. *Ecological Applications* 8(3), pp760–770.

Landsberg, J., James, C., Morton, S., Hobbs, T., Stol, J., Drew, A. and Tongway, H. 1997. *The Effects of Artificial Sources of*

Water on Rangeland Biodiversity. (Commonwealth of Australia, Canberra).

Loveland, T.R. and Belward, A.S. (1997). The IGBP-DIS global 1 km land cover data set, DISCover: first results. *International Journal of Remote Sensing* 18, 3289–95.

Ludwig, J. and Tongway, D. (1998). Ten years on, created landscape patches are still functioning. *Range Management Newsletter*. November, pp1–7.

McIntosh, P.D., Allen, R.B. and Scott, N. (1997). Effects of exclosure and mangement on biomass and soil nutrient pools in seasonally dry high country, New Zealand. *Journal of Environmental Management* 51 pp169–86.

McKeon, G.M., Howden, S.M. and Stafford-Smith, M.D. (1992). The management of extensive agriculture: greenhouse gas emissions and climate change.In: *Assessing Technologies and Management Systems for Agriculture and Forestry in Relation to Global Climate Change.* Proceedings of IPCC Working Group III Workshop, Canberra.

Moore, J.L., Howden, S.M., McKeon, G.M., Carter, J.O. and Scanlan, J.C. (1997). A method to evaluate greenhouse gas emissions from sheep grazed rangelands in south west Queensland.

Noble, J.C. (1998). *The Delicate and Noxious Scrub*. (CSIRO, Canberra).

Odum, E.P. (1969). The strategy of ecosystem development. *Science* 164, 262–270.

Oldeman, L.R., van Engelen, V.W.P. and Pulles, J.H.M. (1991). The extent of human-induced soils degradation. In: *World Map of the status of human-induced soil degradation: an explanatory note*.

Parton, W.J., Stewart, J.W.B. and Cole, C.V. (1988). Dynamics of C, N, P and S in grassland soils: a model. *Biogeochemistry* 5, 109–131.

Pickup, G. (1989). New land degradation survey techniques for arid Australia—problems and prospects. *Australian Rangeland Journal* 11, 74–82.

Schlesinger, W.H. (1999). Carbon sequestration in soils. *Science* 284, 2095.

Shea, S. (1998). Farming Carbon. *Landscope* 14, 17–22.

Skjemstad, J.O., Clarke, P., Taylor, J.A., Oades, J.M. and McClure, S.G. (1996). The chemistry and nature of protected carbon in soil. *Australian Journal of Soil Research* 34, 251–71.

Stafford-Smith, M., McKeon, G.M., Buxton, R. and Breen, J. (1999). The integrated impacts of price, policy and productivity changes on land use in northern Australia. In: *Proceedings of the 6th International Rangelands Conference*, Townsville, Australia, pp864–866.

Stallard, R.F. (1998). Terrestrial sedimentation and the carbon cycle: Coupling weather and erosion to carbon burial. *Global Biogeochemical Cycles* 12, 231–257.

Squires, V. and Glenn, E. (1996). Creating an economic linkage between fossil fuel burning, climate change and rangeland restoration. In: *Rangelands in a Sustainable Biosphere. Proceedings of the Fifth International Rangeland Congress, Salt Lake City, Utah, USA, 23–28 July, 1995. Volume 1: Contributed Presentations.*(Society for Range Management, Denver, USA).

Tongway, D. and Hindley, N. (1995). *Manual for the Assessment of Soil Condition of Tropical Grasslands.* (CSIRO Wildlife and Ecology, Canberra).

Tongway, D. and Ludwig, J. (1996). Rehabilitation of Semiarid Landscapes in Australia. I. Restoring Productive Soil Patches. *Restoration Ecology* 4, 388–397.

Tothill, J.C. and Gillies, C.G. (1992). The pasture lands of northern Australia. *Occasional Publication No. 5*. Tropical Grasslands Society, Brisbane.

Tothill, J.C., Hargreaves, J.N.G., Jones, R.M. and McDonald, C.K. (1992). BOTANAL—a comprehensive sampling and computing procedure for estimating pasture yield and composition, I: Field sampling. Tropical Agronomy Technical Memorandum No. 78. CSIRO Tropical Crops and Pastures, Brisbane.

Trumbore, S.E., Chadwick, O.A. and Amundson, R. (1996). Rapid exchange between soil carbon and atmospheric carbon dioxide driven by temperature change. *Science* 272, 393–96.

Wilson, A.D. (1979). Management of the natural pastures of south-western New South Wales. *Australian Rangeland Journal* 1, 351–362.

Woody Weeds Task Force (1990). *Woody Weeds in Western NSW: Background Statement.* (NSW Government Printing Service, Dubbo NSW).

Carbon in Woodlands—CRCGA Project 2.2

Bill Burrows

Queensland Department of Primary Industries

This project is utilising ground based measurements to estimate C stocks & fluxes in grazed woodlands of (initially) north-east Australia. It primarily draws upon a network of permanent woodland transects, established since 1982, to monitor structural and growth parameters on all woody plants present in the transect belts. A QDPI field team, led by Laurie Tait, maintains the sites and progressively updates site records. A CRC appointee, Madonna Hoffmann, is helping to synthesise the enormous data files already in hand. Appropriate tree allometrics have been developed with support from the AGO (via BRS) to augment relationships previously formulated by QDPI and other agencies.

The project has submitted a paper "Allometric relationships and community biomass estimates for some dominant eucalypts in Central Queensland woodlands" to Aust. J. Bot. Another detailing "Growth increment, carbon stocks and fluxes in eucalypt trees in grazed woodlands of north-east Australia" is in preparation. Results are given for 59 eucalypt sites (representative of c.40M ha eucalypt woodland in Qld) in which stock change in standing live and dead woody biomass has been monitored for a mean 8.2 yrs (range 2–18 yrs) – all values (95% C.I.: (see table below)

There were major differences in the mean stock flux in eucalypt woodlands of similar initial live tree biomass (c. 60 t/ha) on 31 sites monitored for an average 14 yrs cf. 28 sites monitored for an average <3 yrs (0.63 cf. 1.55 t dm/ha/yr respectively).

Live tree basal area increment averages c.2%/yr which is in the range reported by Bob Scholes for southern African savannas. The stock change estimates are also in accord with mean values reported for all Indian 'forests' – 1.13 t/ha/yr – by Lal and Singh (Environ. Mon. & Assess. 60:315). The latter study does not take into account any decay in standing dead material.

We have now extended the individual tree allometrics to woodland stands. This has produced reliable predictor equations between mean stand basal area (m2/ha) and above-ground biomass (t/ha). A stand 'height' predictor for stand biomass looks equally promising. Stand growth rates (t/ha/yr) for eucalypts are derived from the basal area increment measured on the permanent monitoring plots and stand biomass equations. These suggest maximum biomass accretion in woodland eucalypt trees (c. 1.6t/ha/yr) occurs at a stand basal area of c. $19m^2$/ha. Mean total basal area of eucalypts contributing to the 59 stands in the above table is presently c. $10.5m^2$/ha.

In addition to the above summary attention was drawn to the "Woody Plant Encroachment Bibliography" detailing over 300 references to the worldwide proliferation of woody plants in grazed grasslands and savannas. This is accessible through the active Web site maintained by Dr Steve Archer, Texas A&M University. The Web address is: http://cnrit.tamu.edu/rlem/faculty/archer/bibliography.html

	Initial stock	Final stock	Stock change	Stock flux
	(t dm/ha)	(t dm/ha)	(t dm/ha)	(t dm/ha/yr)
Eucalypts	65.3 (8.47)	70.62 (9.81)	5.31 (2.39)	0.76 (0.34)
All species	71.5 (9.21)	78.34 (10.47)	6.84 (2.67)	1.06 (0.48)

Non-Forestry Vegetation Fluxes (Vegetation thickening, thinning and clearing)

Rod Fensham

Queensland Herbarium, Brisbane

It is important that the components of vegetation fluxes are clearly separated on the basis of the extent to which they can be assigned an "anthropogenic" origin. The following framework may be useful in this regard and hopefully identifies the areas for which there is clear agreement and the areas over which there is argument.

a) Changes in woody biomass, cover or basal area that result from mechanical vegetation clearing

These include decreases as a result of bulldozing, tree herbicide, ringbarking, tree felling and regrowth of vegetation after these impacts. These processes are clearly understood, are a direct and exclusive result of human activity and can be easily accounted. This category of flux should be accepted as accountable under the Kyoto agreement. This category can also be relatively readily manipulated with an easily measured and guaranteed emissions outcome. It is unclear that this category of vegetation flux is accepted by these terms in the "Greenhouse Sinks and Kyoto Protocol: An Issues Paper" because regrowth seems to be included in the definition of vegetation thickening which it is argued should not be included.

b) Changes in woody biomass, cover or basal area that result from fire and/or grazing

The essential difference between this form of vegetation flux and a) is their ambiguous origin i.e. they can be i) directly human induced, ii) indirectly human-induced or iii) non-human. For example in the case of fire there are non-anthropogenic fires (lightning fires), manipulation of fires indirectly by humans (i.e. in grazing systems herbivory can reduce the potential fire frequency regardless of ignition source), and direct human induced fires (i.e. where fire is used deliberately as a means of vegetation clearing or where a deliberate policy of fire suppression allows for forest development). This is the category that is currently contentious. The direct component should be included, the indirect could be included and the non-human clearly shouldn't be included. Burrows and others would argue that the indirect component of this flux should be included and that it may be of sufficient magnitude to result in the LUCF sector being a net sink in 1990 (i.e. Article 3.7 would not be triggered). However, there is clearly insufficient evidence on this point to make a quantified judgement. Furthermore this component is not really relevant as a sink (under Article 3.4) unless the rate at which CO_2 is being sequestered by this process is increased or decreased. Clearly the manipulation of these processes would be difficult and could not conceivably be performed with sufficient certainty to qualify for GGAP funding.

c) Changes in woody biomass, cover or basal area that result from climatic variation

These fluxes are essentially not human induced. However, it should be recognised human activity may be influencing climate, it is certainly not something that can easily be manipulated at national spatial scales. This category of trends is not contentious in relation to the Kyoto agreement and clearly should not be included. Fensham's recent research has suggested that this component is substantial and would make the separation of b) from c) for carbon accounting purposes extremely difficult.

Note: This categorisation is not completely perfect because all of the factors in b) and category c) can influence fluxes in that part of a) represented by regrowth after clearing. Hence these components would be difficult to tease out. However, it is the least ambiguous of the categorisations I can come up with. For practical accounting purposes it may be possible to arrive at an average equilibrium of regrowth (x% of original biomass for some broad land type categories). Regrowth is accepted as a direct anthropogenic phenomena because clearing is only accounted for down to that equilibrium, but the fate of that regrowth can be otherwise ignored for accounting purposes.

Woody weeds

The invasion of woody weeds may fall into category b). That is they are an indirect effect of changes to grazing and fire regimes. Some woody weeds are exotic and noxious. Thus if the indirect-human component of category b) were accepted in an inventory, it may be necessary to distinguish between relatively benign increases in vegetation structure (such as thickening in native vegetation) and undesirable increases (such as the spread of prickly acacia).

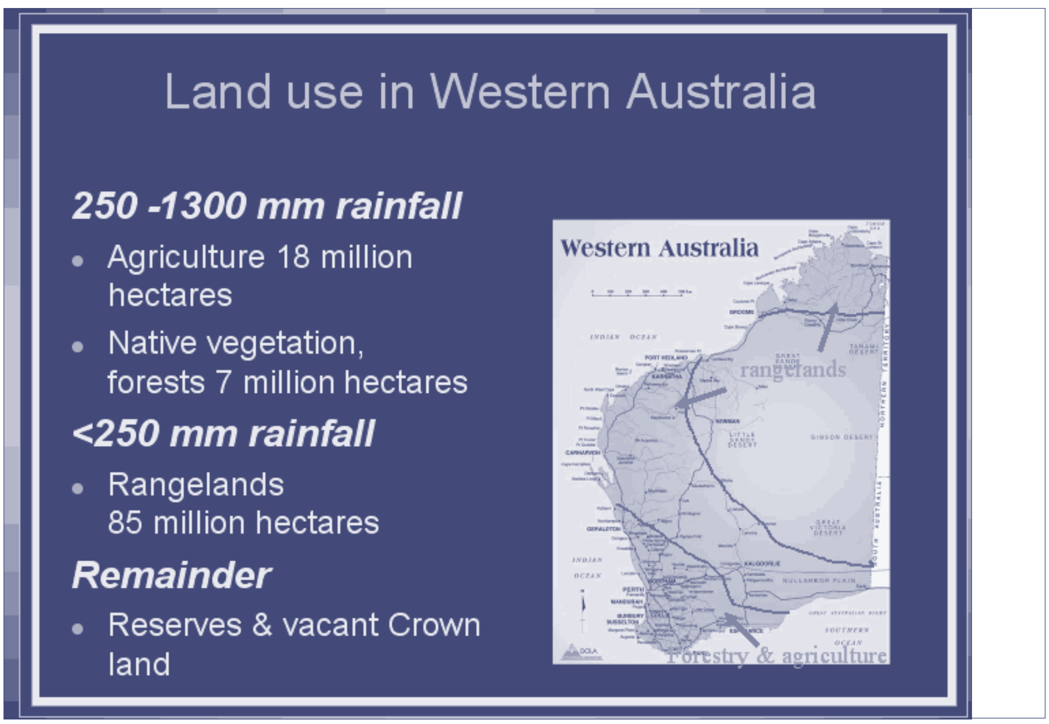
Conclusion

There is an essential difference between changes in vegetation structure because of a) clearing, b) fire and grazing, and c) climate. Category a) is unambiguously an effect of direct human activity and category c) is widely accepted as a non-human activity. Category b) is ambiguous because it includes direct and indirect human components as well as a non-human component.

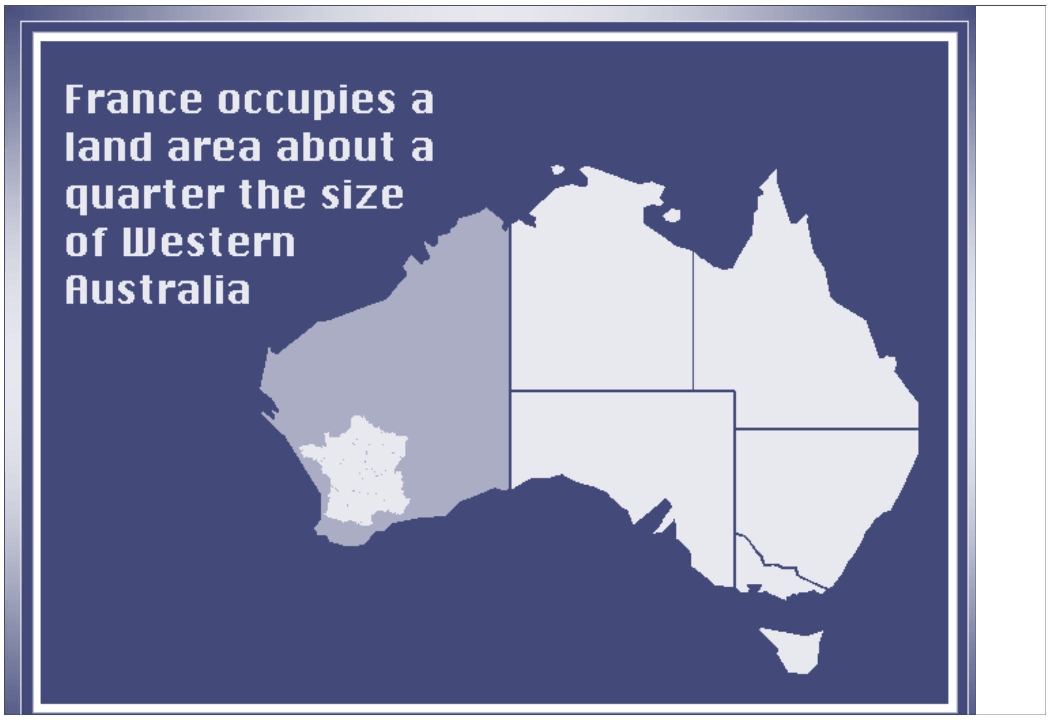
Paul Biggs

Department of Conservation and Land Management CRC

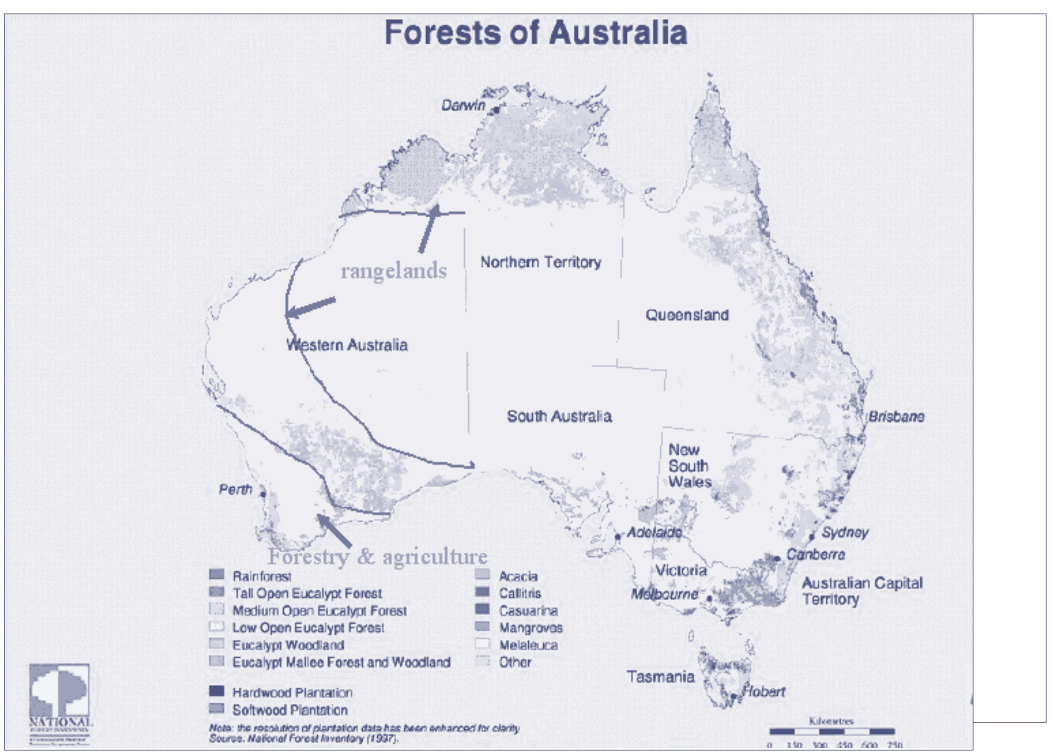
Slide 1



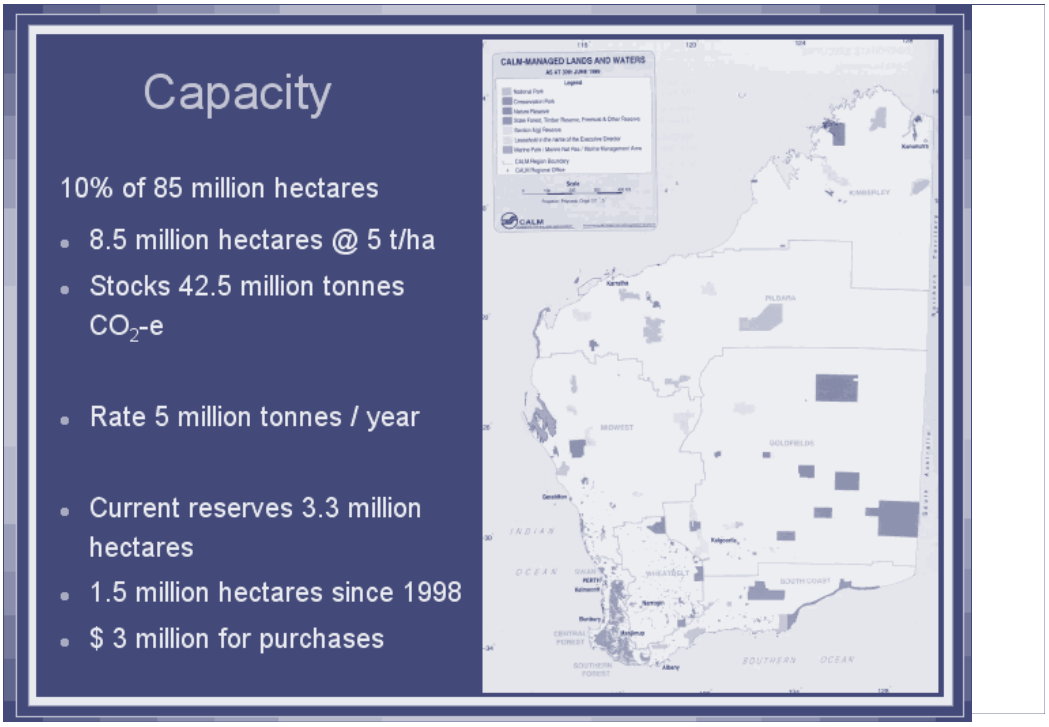
Slide 2



Slide 3



Slide 4



Slide 5