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CHAPTER III

Carbon sequestration in Australian grasslands: policy and technical issues

Abstract

Although Australia belatedly ratified the Kyoto Protocol in December 2007, the diversity of political opinion about climate change has precluded Australia from reaching definitive national greenhouse gas (GHG) emission mitigation policies so far. However, mitigation options involving carbon (C) sequestration into the land is widely perceived as a potentially inexpensive option with environmental co-benefits. Australia has about 25 million ha of ley pasture and 460 million ha of permanent native pasture land that often includes shrubs and trees. Data on C stocks is scant, but there may be about 30 billion tonnes C below ground and 15 billion tonnes above ground (incl. trees) in the national grazed land. That is large compared with the formally reported 0.16 billion tonnes/year of Australian GHG emissions. Evaluation of the impact of grassland management on global climate requires full GHG accounting, including for methane (CH_4) and nitrous oxide (N_2O) fluxes from the soil, animals and wildfires, and surface energy budget analysis associated with changed albedo after tree removal. It is not yet possible to make a quantitative estimate, with stated uncertainty bounds, of the current area of grazed land in Australia that has soil or whole ecosystem C stocks that are lower than they would be without its history of pastoral use. There are no comprehensive quantitative surveys. Some forms of grazed land deterioration involve decreased C stocks (e.g. soil erosion), others involve increased C stocks (e.g. woody weed thickening). The database is so poor that three published estimates of the technical potential for increased C sequestration into Australian rangelands by reduced grazing intensity vary by a more than order of magnitude, namely 4.4, 11 and 78 Mt C/year. The data do not even preclude *decreased* C stocks in semiarid rangelands when grazing pressure is reduced. There are several factors that complicate the objective of better

managing grazing intensity by domesticated stock to sequester C into pasture lands. The high frequency of wildfire in Australia, especially where there is a high standing stock of above-ground vegetation, has repercussions for the emission of soot that warms the atmosphere by solar energy absorption. It also leaves very long-lived char in the soil. The high level of herbivory by native and feral animals renders managed reduction of grazing intensity problematic, with increased wild herbivory offsetting reduced domestic herbivory, especially as lower profits with reduced levels of commercial grazing means less funds for feral animal control. Confounding market-oriented C accounting on a project scale basis, is the vast quantity of organic matter that is frequently shifted around the continental landmass and out to sea by major windstorm and flood events. Most emphasis in C-trading via land management concerns remunerating a landholder for building up ecosystem C stocks annually. However, the issue of how the ongoing management regime to sustain those higher C stocks, often involving reduced income from animal production, is achieved and rewarded indefinitely also needs to be addressed. Costs are considerable for indefinitely measuring and verifying project C stocks and also for the opportunity cost of the of mineral nutrients tied up with C in organic matter. The considerable complexities of attempting to use C sequestration into grazing lands for GHG mitigation purposes will demand great transparency in the arrangements of any scheme and well-conceived and managed regulatory protocols.

POLICY ISSUES AND BACKGROUND

The technical possibilities that will be acted upon for sequestering new (i.e. net additional) carbon (C) into grasslands on a national basis are dictated by government policies developed in the context of international agreements. With a change of the national government in December 2007, Australia belatedly became a signatory to the Kyoto Protocol and has since then continued to be favourably disposed to setting up measures to address global climate change as part of a coherent international effort. Policies were developed to be presented to the Fifteenth Conference of the Parties to the 1992 UN Framework Convention on Climate Change (COP15) held in Copenhagen in December 2009.

The primary focus of the Australian Government greenhouse gas (GHG) mitigation policy, following ratification of the Kyoto Protocol, is the development of an emission "cap and trade" legislation, called the Carbon Pollution Reduction Scheme (CPRS), which was intended for introduction in July 2010 (Department of Climate Change, 2008). The CPRS, once passed by Parliament, would reduce, by 2020, the national annual emission rate of all GHGs by between 5 and 25 percent against a 2000 baseline. The actual percentage cap reduction adopted, within that range, depended on agreements at COP15. However, the lack of substantive international agreement at COP15 meant that the proposed target for Australia in 2020 was not established at that time. The CPRS scheme would auction emission permits to large "upstream" firms representing "points of compliance" for GHG emission reduction. This would involve approximately 1 000 (of the 7.6 million) registered businesses in the country that emit more than 25 kt of CO_2eq/year. Such firms account for 75 percent of Australian emissions. The scheme also includes provision for the use of afforestation offsets that can be used to "pay" for emissions in place of the auctioned permits, but it excludes, initially, agricultural sources and sinks (CH_4 and N_2O), which account for 10–15 percent of national net emissions. Although conceptually the Government is keen to include agriculture in the emissions trading scheme, because of the large number of small businesses and the complexity of quantifying agricultural emissions, agriculture will not be included in the scheme at the outset. However, it is proposed in the scheme to examine in 2013 the potential to include agriculture by 2015 at the earliest. The development of the CPRS was informed by a major review – the Garnaut Review (Garnaut, 2008) – which was the Australian equivalent of the earlier United Kingdom Stern Review (Stern, 2006). The proposed CPRS scheme as currently configured (January 2010) involves very large free allocations of tradable emission permits to energy-intensive trade-exposed industries as an initial transitional step.

The CPRS Bill was passed by the Lower House of the Australian Parliament but has (at the time of writing – January 2010) been twice rejected by the Upper House (the Senate) in which the governing Labour Party does not hold a majority. The major opposition Liberal-National Party Coalition has a variety of member-specific objections to the CPRS Bill and no Coalition-agreed position for an alternative. The Green Party's primary objections are that the caps are too low to avoid the risk of dangerous climate change, that the provisions are too favourable to large industries at the expense of the tax-paying community, and that they render personal and small business GHG emission reduction efforts (such as installing solar panels for hot water production, house insulation, using smaller cars, sequestering C in soil, etc.) ineffective because, with the national emission cap fixed, such voluntary savings would be offset by reduced large industrial efforts to decrease emissions to which the permits apply.

One of the reasons why the Government wishes to move to include agricultural businesses in the CPRS scheme is that it is felt that it provides inexpensive opportunities to reduce emissions that will reduce the burden on other sectors of the economy and potentially have environmental co-benefits.

THE NATURE AND CARBON STOCKS OF THE AUSTRALIAN PASTORAL ESTATE

Australian grazing lands span a huge range of ecosystems from a tiny proportion of highly intensive lush irrigated and fertilized pastures to the vast arid and semi-arid rangelands that are too dry, seasonally variable, have low output and are thinly populated for mineral fertilization and other capital improvements such as fencing – other than bores for stock water – to be cost- effective (Table 7).

The grazing areas involved are shown in the land-use map of Table 8. The permanent native grazing lands occupy about 56 percent (430 million ha) of the continent (Table 7). Additionally, there are 20–25 million ha of ley pasture in rotation with crops in areas classified as dryland agriculture and a small area of irrigated pasture. A large fraction of the native pasture rangelands contains trees as well as grazeable grasses and herbs, and is sometimes classified as "forest", such as when using the FAO definition of forest[1] for C accounting purposes. For much of the area, multidecadal management of the unpalatable woody trees and shrubs is a critical part of grazing land management as well as being a major part of the grazing land C stocks.

Published data on C stocks in Australian grazing lands are sparse. Gifford *et al*. (1992) made an estimate of above- and below-ground C in Australian ecosystems based on the global compilation of Olson *et al.* (1985). Bearing in mind the large uncertainties both in the areas that can be designated as grazed land, and in the C densities in grazed ecosystems, together with the year-to-year variation in grazed areas associated with rainfall variation, wildfire extent and prices for animal products, it is assumed (based on Gifford *et al.* 1992), for the purpose of this paper, that the below-ground C stock in grazed land approximates a rounded figure of 30 Gt C (which calculates to a mean density of approximately 60 tonnes C/ha). This average figure

[1] At least 10 percent crown cover of trees with a height at maturity of at least 2 m, in an area of at least 0.05 ha (FAO, 2006).

has a large but unknown uncertainty. The above-ground C in continental grazing land adopted here is 15 Gt C, including the C in trees and shrubs in the rangelands – also with high uncertainty. The huge size of these grazed ecosystem C stocks, relative to national annual GHG emissions of about 160 Mt Ceq/year, combined with a popular "received wisdom" that most rangelands are overgrazed/degraded (and, by tacit implication, have diminished C stocks), leads to a spirit of optimism, not least in some political and financial investment quarters, that there is a large inexpensive potential to accommodate national GHG emission reduction by improved management of grazing lands to increase C stocks at a low cost.

SCOPE OF GREENHOUSE GAS EMISSIONS FROM PASTURES

For meaningful national or global climate change mitigation and evaluation of the potential to reduce net GHG emissions to the atmosphere from the land *full* GHG accounting above and below ground is required, as is consideration of wider C cycle and climate change issues of surface energy balance, owing to interactive effects of management options. Not only carbon dioxide (CO_2), but also CH_4 and N_2O emissions to, and/or removals from, the atmosphere occur in agricultural land, including grazed grassland soils. Methane is emitted by grazing ruminants and by wildfire. Ruminant enteric fermentation produced about 16 Mt Ceq in Australia in 2007 (Department of Climate Change, 2009), this amounting to approximately 10 percent of the nation's official GHG inventory. Nitrous oxide emissions are relatively minor but can be substantial in locations where nitrogen (N) fertilization is practised. The amount of CH_4 emitted per kg of animal products decreases with increasing quality of the feed. Thus, concentrating agricultural inputs, including fertilizer and irrigation, in high-quality pastureland can have the effect of maintaining the meat and dairy output for less CH_4 production. However, where intensive animal production involves the use of artificial N fertilizer, N_2O emissions may increase, counteracting the greenhouse impact of reduced CH_4 emissions. In addition, with the present decade-long period of rainfall deficit in Southeast Australia, which may or may not be an expression of global climate change, opportunity for irrigation is currently declining, rather than increasing.

Above-ground management of rangelands can have a substantial impact on the total ecosystem C stocks and hence on CO_2 emissions. As indicated previously, the above-ground C, including woody components, occurs at about half the density per unit land area as below-ground C as an overall continental average. Management by grazing, and by tree clearing and reclearing after woody regrowth (Gifford and Howden, 2001), has big impacts on the total ecosystem C stock mainly via the amount of woody biomass. These need to be taken into account.

In terms of the impact on climate, the effect of the type of vegetation cover on surface energy balance, and hence temperature, also needs consideration. Woody vegetation is generally darker than the dry grassy vegetation of the rangelands. The darker surface has a lower albedo and hence may warm the adjacent atmosphere by day (Bounoua *et al.,* 2002).

Thus, although this paper is primarily about biological C sequestration, it is important to recognize that, when attempting to use biological C sequestration as a GHG mitigation strategy, the implications for the climate stretch beyond the CO_2 removed from the air by the ecosystem under management. The climate change implications of additional repercussions should be quantitatively accounted for in any approach to financial remuneration.

WHAT IS THE POTENTIAL FOR SOIL CARBON SEQUESTRATION INTO AUSTRALIAN GRASSLANDS?

The Garnaut assessment of the potential for soil carbon sequestration in Australian pastures

According to the Chicago Climate Exchange rules for C accounting, which were adopted by Garnaut (2008) to calculate the C sequestration potential by Australian grasslands, soil C stocks in degraded rangelands may be increased for C credit purposes by certain changes in grazing management practices – "that include use of *all* the following tools through the adoption of a formal grazing plan:

- light or moderate stocking rates;
- sustainable livestock distribution which includes:
  - rotational grazing
  - seasonal use" (Chicago Climate Exchange, 2006).

Thus, it is assumed that, if a grazier undertakes to adopt all of the above grazing management practices on a degraded rangeland, certain amounts of C sequestration will be assumed. The Garnaut Review estimated that the technical potential for C sequestration rate into Australian pasture soils is 78 Mt C/year (286 Mt CO_2eq/year) over a period of 20–40 years. Over the 358 million ha of land that Garnaut considered as grazing land, this amounts to an annual sequestration rate of 270 kg C/ha/year. Although details were not given, this calculation was said to be based on the Chicago Climate Exchange rules for when degraded pastures are managed by the above-specified practices. Gifford and McIvor (2009) subsequently attempted an analysis of the potential of Australian pastures to sequester additional C and were unable to find evidence to support the large Garnaut assessment. The evaluation asked whether all Australian grazing lands are degraded and hence potentially amenable to increased C stocks by the above grazing plan, and by how much reduced grazing of degraded pastures increases C stocks.

How degraded are Australian pastures?

The terms "degradation" and "deterioration" are applied both to the condition of the vegetation and the condition of the soil. Although the two may be related, they are not synonymous. "Desertification" is another term used to refer to degradation (Dregne, 2002). The notion of "degradation" varies with author. No explicit agreed definition has emerged and distinctions are not always specified or their existence acknowledged. The word "overgrazed" is also used and is not synonymous with either "degradation" or "deterioration". The extent of soil or pasture degradation through overgrazing, anywhere in the world, has relied on local or regional expert subjective opinion of the state of deterioration, rather than systematic quantitative criteria. Globally, such local expert opinion on degradation was compiled by a GLASOD (Global Assessment of Human-induced Soil Degradation – International Soil Reference and Information Centre) survey (Oldeman, Hakkeling and Sombroek, 1990; Oldeman, 1994; and ftp://ftp.fao.org/agl/agll/docs/landdegradationassessment.doc/).The tropical north of Australia has also been subject to more specific evaluation. A compilation of local expert opinions was prepared by Tothill and Gillies (1992) throughout Queensland and the tropical north of Australia. These two compilations give divergent perspectives of the proportion of grazed land that is thought by local experts to be degraded in Australia. Conant and Paustian (2002) calculated from the GLASOD survey of the 1990s that 11 percent (49 million ha) of 437 million ha of grassland in the Australia/Pacific (predominantly Australia) region was overgrazed. Ash, Howden and McIvor (1995) summarized the opinion survey conducted by Tothill and Gillies (1992) for 143 million ha of grazing lands in northern Australia covering Queensland, the Northern Territory and western Australia. The survey found that 30 percent of these lands had deteriorated somewhat and 9 percent were severely degraded.

The difference of impression is not only because different areas of territory are involved, but also because they may not be clearly distinguishing soil degradation from pasture degradation and not explicitly defining what the local experts meant by "degraded". Perhaps each local expert did not know explicitly either.

For Queensland alone, the Tothill and Gillies (1992) compilation is summarized in Table 8. It indicates that 41 percent of Queensland rangeland pastures were considered deteriorated around 1990 but could be recoverable with improved management and "normal" rainfall, while 17 percent were considered degraded beyond recovery without high expenditure and complete land-use change. There are many forms of degradation, such as soil erosion of various types, soil compaction, soil acidification, salinization, undesirable change in herbaceous species composition (e.g. annual grasses replacing perennials), loss of plant cover, woody plant thickening, weed invasion and loss of biodiversity, each with different implications for soil C stocks. Notes alongside the individual entries of the Tothill and Gillies (1992) compilation that are summed up in Table 8 indicated woody species thickening was a dominant form of deterioration in Queensland. But the fraction of the area that is designated in Class B or C (see Table 8) that is experiencing increased woody plant cover, as opposed to replacement of forage plant cover by bare ground, is not indicated. This distinction is critical in terms of whether the C stocks of the rangeland have increased or decreased as a result of the deterioration and degradation. For 60 million ha of grazed woodlands in Queensland, Burrows *et al*. (2002) showed that the mean rate of increase of above-ground biomass by woody thickening was 530 kg C/ha/year from which they estimated that the total above- and below-ground increase in all grazed woodlands of Queensland could be about 35 Mt C/year.

An earlier assessment for Australia as a whole in 1975 (Australia, 1978) was summarized by Woods (1983). This study indicated that of 336 million ha of grazed arid rangeland in Australia, 55 percent was affected to some degree by vegetation or soil deterioration. The fraction in the substantial degradation category was 13 percent (43.2 million ha) of the pastoral land in the arid zone (8 percent of the total arid zone).

From the above, it is not possible to make an unambiguous quantitative estimate, with stated uncertainty bounds, of the current area of grazed land in Australia that has soil, or whole ecosystem, C stocks that are lower than they would be without its history of pastoral use. However, from all the above efforts, the areas that are deemed by local experts to be deteriorated or degraded seem to be much less than the 100 percent implicitly assumed in the Garnaut (2008) estimate.

By how much does reduced grazing intensity increase soil or ecosystem carbon stocks?

It seems simple. As a first line of consideration, removal of herbage by grazing animals, the products of which are exported off the land, must reduce the amount of both organic C and minerals that an ecosystem recycles into its litter and organic matter stocks via tissue death, decomposition and turnover, compared with the same ecosystem if it were not so grazed. Therefore, decreasing the grazing pressure should increase C storage by the ecosystem, thereby removing CO_2 from the air. *Unfortunately, ecosystems are much more complex than the above simple logic suggests.* One of the complexities is that ecosystems are *dynamic* – they are in a continuous state of change, both naturally (Walker and Abel, 2002) and under different management regimes.

One of the dynamic changes in "native" pastures is the fraction of trees and shrubs in the grazed ecosystem. A major form of degradation of Australian grazed tropical rangelands is woody species thickening and encroachment (Gifford and Howden, 2001). This is in fact a big problem for graziers in tropical Australia. The thickening woody vegetation competes with the herbaceous forage and reduces stock carrying capacity and profitability. The reason for woody thickening is not unequivocally established but the most well-received hypotheses are that: (i) the woody species that proliferate are unpalatable to the domesticated stock and therefore, once established, become predominant over the grazed species; and (ii) the grazing of the dead standing grassy biomass reduces wildfire frequency and intensity, thereby increasing the amount of woody plant establishment and survival that are otherwise suppressed by fire. Thus, since grassy ecosystems have higher C stocks with thicker density of trees and shrubs than without, where woody "weed" thickening occurs there can be a switch from high grazing intensity fostering whole ecosystem C accumulation (i.e. a positive correlation between grazing and ecosystem C accumulation) to negative correlation between grazing intensity and ecosystem C stock accumulation because the form of high C stocks (woody weeds) reduces stocking capacity. In Australia, there now exist laws and regulations that inhibit graziers from clearing the trees from the land. Where this reaches the point at which a grazier is forced out financially and the stock is removed, it is an open question as to what happens to the ecosystem dynamics and C stocks thereafter. One course of events could be that the trees, once well established before abandoning of grazing, would continue growing and thickening until a major intense wildfire event occurs, removing the woody cover, opening up the landscape to grass re-establishment and the frequent-fire controlled grassy landscape. In that case, the increased C stocks associated with the (tree-forced) reduced grazing would go back to the atmosphere as CO_2. We do not know the answer, but the key point is that for climate change mitigation purposes, the tree-encroached tropical rangeland is not necessarily a stable or reliable repository for atmospheric C.

It is assumed in the Chicago Climate Exchange rules, which were used by Garnaut (2008), that by reducing grazing intensity a grazier could increase soil C stocks. What is the evidence for that assumption? There has been surprisingly little study of the effects of grazing intensity in Australia (or, indeed, elsewhere) on soil C stocks. The combination of paucity of relevant measurement and experimental data combined with the complexity of confounding factors in the complex adaptive system of rangeland ecosystems (Walker and Abel, 2002) makes that question difficult to answer with confidence and is partly dependent on the timescale to which one is referring. Annual farm-level accounting of ecosystem C for mitigation monitoring via measurement is neither financially viable nor conceptually appropriate. As with climate change itself, in the C cycle of terrestrial ecosystems we are dealing with phenomena that have relaxation times of decades to centuries. These difficulties notwithstanding, Ash, Howden and McIvor (1995) used the results of grazing exclusion experiments and paired sites to estimate that if all deteriorated (43 million ha) and degraded (13 million ha) northern Australian rangelands could be returned to a desirably sustained condition by reduced stocking, 459 Mt C could be sequestered in the top 10 cm of soil. If achieved to saturation of the potential over 40 years, this would represent an annual average sink of about 11 Mt C/year, amounting to a mean 205 kg C/ha/year over the half century.

Conant and Paustian (2002) attempted an analysis of peer-reviewed world literature on soil C in relation to overgrazing. They found only 22 studies globally meeting their selection criteria of deteriorated soil C stocks' response to grazing pressure. Only one of these was in Australia. That was in the environmentally special alpine meadows (of relatively minute extent) high in the Snowy Mountains in temperate Southeast Australia, grazing of which is no longer permitted. Making the most of the limited data, Conant and Paustian tentatively estimated that the technical potential to sequester soil

C by reduced grazing in Australian permanent pastures was 4.4 Mt C/year, corresponding to 90 kg C/ha/year. However, in the actual data set found by Conant and Paustian, seven of the 22 points indicated *decreased* soil C stocks after grazing pressure was relaxed. The decreases occurred in the drier environments. As a consequence, the error bars around the estimate are very large indeed. It is possible, therefore, that for drier areas like most of the Australian rangelands, reduced grazing intensity could reduce soil C stocks.

These two estimates of the technical potential for C sequestration in Australia (11 and 4.4 Mt C/year) are an order of magnitude lower than the Garnaut (2008) estimate of 78 Mt C/year based on the Chicago Climate Exchange methodology in the hands of those advocating market-based C trading. Of course, realizable sequestration would be much less than the national technical potential owing to various problems of implementation and documentation. Given the fact that there were several examples at the dry end of the data range in the Conant and Paustian data set for which removal of grazing decreased C soil stocks, even the low estimates of technical potential could be too high or even of wrong sign. Accordingly, it is imperative to understand the circumstances in which soil C stocks decrease when grazing pressure is relaxed. If it is true that there are circumstances in which relaxation of grazing intensity leads to decreased soil C stocks, then an added layer of uncertainty and complexity is introduced to the objective of improving grassland soil C stocks and sequestering C into soil by grazing management, especially through a cost-effective market mechanism.

Do soil carbon stocks really decrease under reduced grazing pressure in some sites?

There is a risk in meta analyses of data from disparate literature sources, such as that of the Conant and Paustian (2002) study, that contrasting results may be attributable to unidentified differences in methodology between studies. Thus, one might fear that some observations indicating the opposite trend to expectation are not correct. However, with regard to the decrease in soil C when grazing is relaxed or removed, a recent experimental study of grazing exclusion effects on soil C in grasslands of the Rio del la Plata region of Uruguay and Argentina (Pineiro *et al*., 2009) has confirmed, using a single methodology, the observation of variable effects of grazing on soil C stocks in the top metre of soil for 15 paired sites (grazed versus ungrazed non-shrubby grasslands) over 70 million ha of the region. In this study the soil C stocks increased upon grazing removal in the upland sites, but *decreased* in lowland sites and in shallow soils. As a hypothesis, these contrasting responses of soil C stocks to grazing pressure may be a reflection of: (i) root mass response to grazing; and (ii) N cycle responses to grazing (Pineiro *et al*., 2009), soil C dynamics being known to be tightly linked to root turnover and N dynamics. Literature evidence suggests that grazing reduces root biomass in mid-range rainfall sites (400–850 mm/year), but *increases* root biomass in wetter and in drier locations. Thus, in rangelands (dry environments) an increased root biomass under grazing pressure could increase soil C stocks, particularly if the methodology adopted includes root C as part of "soil" C, as it often does. With regard to the N-cycle link, grazing can have two opposing effects: (i) the grazed ecosystem can lose a lot of N via volatilization of ammonia and nitrate leaching from animal urine and dung patches, the amount depending on many factors; and (ii) in increasing root growth in wet and dry locations, grazing also increases N retention in roots that will increase soil organic N content as the root dies and decomposes. The balance between these opposing effects will vary according to a range of site-specific factors leading to increased soil C under grazing pressure in some sites and decreases in other sites.

COMPLICATING FACTORS

There are additional complicating factors that need to be addressed in order to implement a successful and equitable use of biosequestration of C as a tradable offset to fossil fuel emissions of CO_2.

Wildfire

Australia is a wildfire-prone nation as the tropical savannahs are the most fire-prone ecosystems. They burn as frequently as annually in the late dry season. The burning not only converts above-ground biomass to CO_2 but also gives off CH_4, N_2O and black C (soot) in the smoke. Grazing intensity influences both fire amount and fire intensity and the latter influences the amount of CH_4, N_2O and soot emitted per unit biomass burned. The effects on soil C are far from clear, but burned grass and litter are organic matter that cannot become incorporated into soil organic matter (SOM). However, the small fraction of burned biomass C that becomes char on the soil, which is a long-lived form of soil C, has a residence time said to be in the order of 2 000 years (Lehmann *et al.,* 2008). The fraction of soil C that is black C ranges as high as 82 percent in Australia, although mostly much lower than that (Lehman *et al.,* 2008). While the black C that goes into the soil is a long-lived C stock, the black C that goes into the air as soot is another source of atmospheric warming. Atmospheric black C from fire absorbs incoming solar radiation, thereby warming the atmosphere. Global emissions of black C are claimed now to be the second highest cause of global warming after CO_2 (Ramanathan and Carmichael, 2008). However, unlike incremental CO_2, which remains airborne for at least 100 years, black C has an atmospheric lifetime of about a week (Rodhe, Persson and Akesson, 1972). Thus, reduction of black C emission is a powerful mechanism for quick reduction of global warming. Policies to increase the standing stock of pasture grasses in Australia would have led to increased organic matter consumed in wildfires and increased black C emission to the atmosphere, thereby offsetting, in the short term, the longer-term advantage of increased net standing stock of C in the nation's rangeland grass and soil C inventory. In short, the implications for global warming of building up Australian savannah biomass are complex and difficult to analyse, given the variety of effects of organic C stored, black C produced in the soil, black C emitted to the atmosphere, and CH_4 and N_2O production.

Non-commercial herbivory

Competing with production from the approximately 25–30 million cattle and 70–90 million sheep on the Australian rangelands are native herbivores – kangaroos and wallabies, and grasshoppers and locusts – and several feral herbivores. If grazing land is allowed to recover C in herbage, and possibly in soil, by reducing stocking intensity with ruminants, there is a tendency for the non-commercial herbivore numbers to increase, especially if the watering-points are not closed off.

When an income is being derived from grazing, stock graziers can afford the routine culling of kangaroos that is necessary to have enough herbage for the ruminants to graze profitably. Despite the culling, the national red and grey kangaroo population varies between 15 million and over 40 million depending on rainfall, which determines forage available to domesticated stock (Pople, 2004). A small fraction of the kangaroo population is harvested commercially for meat and leather under a well-controlled government management scheme but the economic return is minimal compared with that from ruminants (Ampt and Baumpter, 2007). Whether or not kangaroo production could eventually substitute for cattle and sheep production to a significant extent is a much and emotionally debated question. An advantage of kangaroos is that they do not regurgitate CH_4 (Klieve and Ouwerkerk, 2007). There are, however, several major practical disadvantages.

The numbers of feral herbivores also vary widely with conditions and so available data on numbers are approximate. They include the worst feral herbivore – rabbits (high numbers, highly variable); camels (0.5 to 1 million and rapidly increasing, Australia DEH, 2004a); horses (about 0.3 million, Australia DEH, 2004b); donkeys (5 million, Australia DEH, 2004b); and goats (2.6 million, Australia DEH, 2004c) and six species of deer (unknown numbers). These high numbers are despite major control measures. The collective impact of all these non-commercial herbivores is considerable and, given the low success of expensive control measures, greatly reduces the capacity to decrease overall herbivory in order to build up ecosystem C stocks.

Lateral transport of carbon

As in many parts of the world, movement of topsoil by water and wind erosion is a significant confounding factor in determining the amount of C stored *in situ* by any management action in Australia. Arid and semi-arid regions are particularly prone to normal lateral transfer of soil owing to the extremes in weather in which prolonged drought, causing low vegetated cover of the soil, is punctuated by extremes in wind or rainfall intensity. While some surface soil is being moved around the landscape locally at low levels all the time in rural areas, the rate and space scale of impact varies hugely depending on whether a major episodic erosion event has occurred. A major dust storm in eastern Australia in September 2009 carried topsoil C from the rangelands of central and eastern Australia out to sea with some deposition, substantial enough to be readily evident on car windows as far away as New Zealand, over 2 000 km away (AFP, 2009). Very large quantities of topsoil are transported. For example, a large dust storm on 23 October 2002 that traversed eastern Australia was 2 400 km wide, up to 400 km across and 2 km high and contained aloft some 3.4–4.9 Mt of dust estimated for 9 am on that day (McTainish *et al.*, 2004). Of course the total dust transported during the whole event would have exceeded, possibly greatly, the amount aloft at any one moment. The dust picked up is the very topmost topsoil containing the most recently deposited SOM for which people may have been paid money in an agricultural C trading system. The organic content of dusts in Australian dust storms averages 31 percent in contrast to the 1 percent for overall dryland topsoil (McTainish and Strong, 2007). Applying that organic matter concentration, and assuming 55 percent C in the surface-SOM, means that the organic C that was aloft at 9 am on 23 October 2002 in eastern Australia was about 0.7 Mt. At, say, AUD15 per tonne C, which equals AUD10 million worth of recently sequestered C aloft at that time, much of it heading out to sea. In terms of the planetary C budget, it is unclear whether organic matter that is blown about through the atmosphere being deposited elsewhere, including substantially into the ocean, oxidizes back to CO_2 more quickly or more slowly than if it stayed in the soil where it was initially sequestered. However, for the people attempting to conduct C trading on a project scale basis, the phenomenon makes for an accounting nightmare. As with wind erosion, huge flooding events, including regular monsoonal ones, shift large quantities of organic matter around the landscape and out to sea.

Harmonizing a short-term market mechanism for CO_2 emission reduction with a long-term ecological process of carbon sequestration having chaotic episodic elements

Carbon sequestered into ecosystem C stocks represents a removal from the atmosphere only as long as the stocks remain at the elevated levels resulting from sequestration. That requires ongoing C stock management. Maintenance of high rangeland C stocks on the decadal to century timescale needed for climate change mitigation presents special challenges for its management via any short-term market-based incentive schemes operating on annual time steps. There are several considerations. There are two steps to reducing CO_2 emissions from the land: (i) increasing the standing stock of C in the plants and soils; and (ii) holding these increased C stocks indefinitely, once they have reached their steady-state limit under the altered management regime, to keep the net accumulated C stock from returning to the atmosphere. Most emphasis in discussion of C trading concerns remunerating a landholder for step (i). However, the issue of how the ongoing management regime to sustain those higher C stocks, often involving reduced income from animal production, is achieved and rewarded indefinitely, also needs to be addressed. If continual remuneration ceases, then the balance of factors for the landholder that lead to higher animal stocking rates and any associated lower ecosystem C stocks may return – see the next section on Costs to the grazier.

Ownership issues are another consideration. Much of Australian rangeland is publicly owned – so-called "crown land" that is leased to graziers. When the land is managed under leasehold to either a private owner or the state the remuneration regime will be more complex. When the lease or land is sold, the burden of the C sequestration legacy may also have to be sold – or should it be leased?

Costs to the grazier

The costs of C sequestration to the pasture manager can be considerable. Although there can be benefits to production of increased soil organic C, there may also be a conflict between maintaining production and sequestering C (Moore *et al.,* 2001). The grazier may receive less income from animal production where, for example, the increased C stock arises from reduced stocking rates or from non-removal of increased woody shrub and tree density. This reduced income stream would be for ever, or until the cost of repaying society to release the CO_2 back to the atmosphere becomes less than any gain in reintensifying the grazing.

Another cost is that of measuring the baseline C stocks and testing the expected increase in C stocks on an indefinite basis. While a modelling approach may be adopted initially in a scheme to "deem" an annual ecosystem C accumulation rate for a particular agreed change in grazing management, it will be essential to test and reset the modelled rate of accumulation every decade or two for each patch of land. This will be necessary to ensure that C has actually been removed from the atmosphere for the particular land involved and that correct financial compensation is changing hands – in whichever direction it needs to go, depending on whether C was accumulated or was lost from the land. The huge variability, on all space scales, of C stocks per unit area, especially (on fine space scales) for the tussock and hummock grasses so common on the Australian rangelands, makes the detection of ecosystem C change (especially soil C change) against that statistical variability extremely expensive. Funding the eternal burden of checking that sequestered C is still in place long after the C stock increase has saturated will be a major impediment to a cost-effective scheme.

Another hidden cost, which might be regarded as an opportunity cost, is the value of the mineral nutrients that are inevitably sequestered along with the C sequestered in organic matter (Passioura *et al.,* 2008). Such minerals are either garnered automatically by ecological processes from the productive outputs of the land or must be applied as fertilizer. Owing to the chemical composition of SOM, each tonne of C in SOM is chemically associated with 100–120 kg of N and 20 kg of P. These amounts, when bound in an enlarged pool of SOM, are effectively unavailable to plant production even though it is a pool that is "turning over", as is the C involved. The value of these elements per tonne of sequestered C, if they were supplied at retail prices of fertilizer, is around AUD150–200 for the N and AUD80–100 for the P at recent prices. Thus, the opportunity cost of the minerals tied up would be around AUD200–300 per tonne of C sequestered. That value greatly exceeds the kinds of value of sequestered C often mentioned (say, AUD15–30). And indeed the current (October 2009) price of C on the Chicago Climate Exchange is only about USD0.5 per tonne of C. These extremely valuable mineral nutrients could be utilized for plant growth in areas of more heavy grazing so that the SOM status declines back down to the presequestration level, and the nutrients thereby released from the organic matter into soluble forms would be available to root uptake while the C is converted back to gaseous CO_2. Thus, well-informed graziers should not accept payment for C sequestered in their ecosystems that is less than the value of the market value of non-C minerals embedded in the sequestered organic matter.

Establishing baseline stocks and flows for carbon trading

In determining the remunerable change in CO_2 emissions associated with a planned change of pasture management regime, there needs to be a baseline year for comparison of ecosystem (or soil) C stocks. Direct measurement of the baseline C, in the baseline year, is generally essential for meaningful C accounting. Nominal deeming rules cannot take account of site history specifics. Direct measurement of the baseline is not, however, possible if the mandated baseline year is in the past (e.g. 1990 for the Kyoto Protocol, or 2000 for the Garnaut and the CPRS proposals).

Not only should there be a baseline in the C stocks of the rangeland, but also a baseline net annual flux (source or sink) in the baseline year (or baseline period) for that rangeland in order to determine the change in flux deriving from the management change. Such a baseline flux will vary with current weather, rate of loss by erosion, current and recent past grazing management, and stage of the rangeland in the resource accumulation/ resource conservation/disturbance/resource release adaptive cycle. The reaction of the ecosystem C sink to a scheme of changed grazing pressure will vary according to where in that adaptive cycle the patch of vegetation was at scheme start. A unit area for the scheme (such as a field, farm, catchment or region) may be a composite of different land patches at various stages in their adaptive cycles and with various land-use histories. The cost and complexity in determining this information for a scheme unit and in finding a way to factor that information into specifying how the agreed management regime has altered those stocks and fluxes on a year-by-year basis are substantial cost impediments to implementation.

A third baseline issue concerns documenting just what the grazing intensity was before the start of the scheme and how it will change after the scheme starts. Prudent graziers already vary grazing pressure enormously over time according to the state of the weather, the state of finances and prices, and land condition. There is rarely an enduring fixed stocking rate in rangelands. During a prolonged drought, a property may carry few stocks for several years. After a heavy rain period or flood event, a property may be able to stock heavily for a couple of years based on the surge of growth. With no fixed grazing intensity or even systematic pattern of varying grazing intensity, it is challenging to define the baseline grazing regime and also the agreed new regime, which it is hoped will lead to net C sequestration unless the new regime is complete destocking. Harper *et al.* (2007), using the Range-ASSESS model for West Australian rangelands, concluded that 50 percent destocking would still lead to some ecosystem C loss in 80 percent of five-year periods. Total destocking was necessary for consistent C accumulation in the ecosystem.

Status of the land from prior greenhouse gas mitigation agreements

Pasturelands were eligible for inclusion under Article 3.4 in national C accounting for compliance under the terms of emissions reductions targets of the Kyoto Protocol to which Australia is a ratified signatory. Unless there is an international agreement to revoke the terms of the Kyoto Protocol, any lands that were submitted to become “Kyoto lands” may have different rules applied by the CPRS to their emission reduction arrangements via subsequent C trading than lands, which are not so constrained by this prior commitment. This may introduce complexity of treatment of C sequestration under new rules, which will need to be accommodated in the arrangements.

The bigger Earth System management problem

GHG emissions are not the only environmental management externality that is in need of special arrangements to compensate for failure of mainstream market mechanisms to take account of the common collective good and intergenerational equity. It is becoming increasingly recognized that the interplay between such global change issues on continental and global scales requires integrated international, interjurisdictional and interagency policy coordination because the environmental issues are interconnected. This spawns the need for a coherent Earth System science that informs an integrated Earth System approach to global environmental governance.

For example, one of the several interconnected issues is the hydrological balance of catchments. In Australia, the state of the nation's water supplies for industry and commerce, irrigated agriculture, stock watering, domestic use and for ecological biodiversity ("environmental flows") is at least as major a topic of recent political debate as is climate change. Major catchments and watercourses straddle different states that have their own jurisdiction over water rights. Large sums of public money are being used to purchase water access rights from producers for rediversion to "environmental flows". The repercussions for regional water storages above and below ground, and of increasing the tree cover, need consideration with regard to appropriate market-based mechanisms for building up ecosystem C stocks and for biodiversity conservation. Hydrologically, there can be both benefits and costs of building up ecosystem C stocks. Trees on rangelands tend to increase the interception and retention of rainfall, but also, being deep rooted, to lower water tables and use more water than the purely grassy/herbaceous vegetation. Thus re-treeing, be it by plantation or woody thickening under grazing, to increase C stocks above and below ground could reduce runoff into rivers, surface storages and aquifers. However, the reduced runoff can in some catchments be primarily from reduced storm flow rather than from reduced base flow (Wilcox, Huang and Walker, 2008), which is favourable for reducing surface soil C losses via water erosion. The balance of hydrological pros and cons will therefore vary with rainfall regime, soil infiltration properties and geology, and will differ for each region.

This hydrological interaction with C biosequestration exemplifies how the use of market-based mechanisms for selected subsets of the complex Earth System problems arising from the still burgeoning deleterious imprint of human beings on the planet can lead to further problems that might be averted if an integrated Earth System analytical approach were used to inform coherent policy and decisions.

CONCLUSIONS

While there is doubtless substantial technical potential to increase C storage in grazed Australian ecosystems above and below ground, an adequate information base for accurately quantifying that expected potential for any specific changed management regime does not exist. It is not yet clear if reduced animal production is always necessarily a concomitant to achieving increased soil C stocks, although that seems logical for most situations. This poor state of the information base will be inhibitory to the uptake of any market-based C trading or GHG trading system for grazing land-based approaches. There are numerous complicating factors that will need to be addressed and dealt with explicitly in any market-based GHG trading scheme that involves C sequestration into grazed ecosystems. These include: linked emission and/or uptake of CH_4 and N_2O associated with management changes for achieving changed C sequestration; the impact on C stocks of wildfire frequency and intensity; compensatory non-domesticated animal grazing; large-scale movement of high C surface topsoil by flood and wind; difficulties in defining baseline C stocks and baseline GHG fluxes from each patch of land under consideration, especially when the requisite baseline is in the past; long time frames (several decades) required; high expense for measuring change in C stocks in each patch of land under a scheme; the high actual input value or opportunity value of the mineral elements chemically associated with increased organic C stocks; the special status of any lands that have already been defined as "Kyoto Lands" by coming under Kyoto Protocol arrangements; and the interaction of C sequestration with other environmental externalities that are coming under different management policy arrangements such as interactions with hydrological and biodiversity policies.

The existence of the above and other real-life complexities will render market-based C trading schemes involving pastures exposed to the risks of complicated, ill-conceived, ill-understood, poorly regulated financial instruments and arrangements that are replete with opportunity for fraudulent scams and inappropriate diversion of community wealth to the personal fortunes of scheme managers and traders, while not delivering the scheme objectives, reminiscent of those involved in the Global Financial Crisis of 2007–09. Thus considerable attention to transparency of the scheme details, the demonstration of actual C sequestration in each scheme by direct measurement of changing C stocks and fluxes from measured baselines, and independent regulation of the arrangements by well-informed regulatory agencies would be needed to deliver the objective of actually slowing the rate of global climate change and sustaining community support for such a venture.

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