Assessing the economic impact of an emissions trading scheme on agroforestry in Australia's northern grazing systems

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Abstract

Although agriculture generates a significant portion of Australia's greenhouse gas emissions, it also has the potential to sequester large quantities of emissions through changed land use management such as agroforestry. Whilst there is an extensive amount of agroforestry literature, little has been written on the economic consequences of adopting silvopastoral systems in northern Australia. This paper reports the economic feasibility of adopting complimentary agroforestry systems in the low rainfall region of northern Australia. The analysis incorporates the dynamic tradeoffs between tree and pasture growth, carbon sequestration, cleared regrowth decomposition rates and livestock methane emissions in a bioeconomic model. The results suggest there are financial benefits for landholders who integrate complimentary agroforestry activities into existing grazing operations depending on the rules of the carbon accounting framework used.

Keywords: Agroforestry, carbon sequestration, financial analysis, carbon accounting framework

Introduction

Whilst it is widely recognised that Australian agriculture generates 16% of Australia's greenhouse gas (GHG) emissions (Department of Environment and Heritage 2006), there is also the potential for Australia's agricultural land to be managed in a way that allows it to become a carbon sink. Research has identified a wide variety of options for sequestering or mitigating GHG emissions through changed land use and land management. Eady *et al*. (2009) identified within Queensland an overall technical potential of 293Mt CO2-*e*/yr for GHG abatement (equivalent to 77% of Queensland's emissions), with 140 Mt of this potential assessed as attainable with concerted effort in technical and management changes, policy adjustment and shifts in current land management priorities.

Included in the range of sequestration options considered were retaining rather than clearing regrowth vegetation and carbon plantings. Eady *et al*. (2009) estimated that carbon-positive management of regrowth could potentially provide 38 Mt CO2-*e*/yr with an estimated attainable GHG sequestration of 7 Mt CO2-*e*/yr. Changing land use to carbon forestry offers greater sequestration potential (153 Mt CO2-*e*/yr) with an estimated attainable annual sequestration of 77 Mt CO2-*e*/yr. These Kyoto compliant agroforestry options could realistically offset 46% of Queensland's 2007 GHG emissions (Eady *et al*. 2009).

Under the Kyoto rules regrowth or plantation forestry activities on land that was clear of forest pre-1990 and converted to forest as a result of direct human intervention is an allowable sequestration or mitigation option. Eastern and southern Australia has a history of extensive land clearing. Due to the clearing techniques used, vegetative suckering from root stocks and seedling establishment occurs resulting in regrowth control being a persistent problem requiring recurrent clearing for many Queensland graziers (Fensham and Guymer 2009). However, if managed appropriately retained regrowth may provide graziers with a carbon sink that complements existing grazing enterprises (such as silvopastoralism systems) without significantly compromising productive capacity.

Natural regrowth has a number of advantages over forestry plantations as a carbon store; it requires no intensive effort of planting, involves tree species naturally adapted to the site and progresses to mature vegetation approximating the original vegetation thus helping to restore ecosystems and biodiversity (Fensham and Guymer 2009). While there is a growing understanding of the sequestration potential of managed regrowth and forestry carbon dynamics, there is currently little known about the economic implications of establishing complementary agroforestry (including regrowth management and plantation) and pastoral systems (silvopastoralism) in northern Australia, particularly in the lower rainfall areas (600-750 mm/yr).

This paper investigates the economic feasibility of agroforestry in the semi-arid areas (annual rainfall of 600-750 mm) of central Queensland. There is potential for the managers of beef cattle grazing systems to encourage regrowth strips in order to sequester carbon in a manner that complements existing grazing operations. The analysis incorporates the dynamic tradeoffs between tree and pasture growth, carbon sequestration and livestock methane emissions. This information has been used to construct a bio-economic model of potential silvopastoralism systems over two representative land types that are compared on a financial basis with a conventional grazing system using 4 potential carbon accounting frameworks.

The case studies reported are particularly relevant given the size of the central Queensland beef industry and the large area of remnant vegetation that has been cleared during the development of the region's extensive grazing systems. Central Queensland is one of the major beef producing regions in Australia with annual cattle slaughtering and product sales valued at $830 million (20% of Queensland's total beef sales). Central Queensland also has the second highest annual rate of vegetative clearing in Queensland (approximately 25,500 ha/annum) with the majority of this clearing occurring on regrowth of brigalow and eucalypt land types (DNRW 2008).

A review of agroforestry research relevant to Australia's semi-arid regions

Agroforestry broadly refers to the purposeful growing of trees and crops, perhaps with animals, in interacting combinations for a variety of benefits and services (Nair 2008). Unlike the production emphasis of agroforestry in the tropics, environmental protection (including carbon sequestration) and monetary return are the main motivating factors for agroforestry in industrialised nations (Nair *et al*. 2009). Alley cropping, forest farming, riparian buffer strips, silvopasture and windbreaks are the five major agroforestry systems undertaken in north America (Nair *et al*. 2009).

In the lower rainfall and subtropical regions of northern Australia the development and adoption of agroforestry systems has been relatively slow. An assessment of the satellite remote-sensed data from Queensland indicates vast areas of land that could be reforested that meets the 'reforestation' definition of the Kyoto Protocol (Fensham and Guymer 2009).

The implementation rules under the Kyoto Protocol define afforestation as 'the direct human induced conversion of land that has not been forested for a period of at least 50 years to forested land through planting, seeding and/or the human-induced promotion of natural seed sources; and reforestation. Reforestation is the direct human induced conversion of non-forested land to forested land through planting, seeding and/or the human-induced promotion of natural seed sources, on land that was forested but that has been converted to non-forested land' (Department of Climate Change 2009, p40).

Given the focus of silvopastoralism on integrating trees, pastures and livestock in an agricultural land-use system, the interaction between trees and pasture is a key factor influencing the dynamics of total production and financial viability of such enterprises. Generally, early tree and pasture based research in Queensland focused on the competitive effects of tree density on pasture production for key woodland genera and species, including *Eucalyptus* spp. and *Acacia harpophylla* (Walker *et al*. 1971; Walker *et al*. 1986; Scanlan and Burrows 1990; Scanlan 1991; McIvor and Gardener 1995). Pasture production benefits of tree removal were identified through higher documented pasture yields on sites with lower tree stocking rates, reflecting direct competition between trees and pasture for water, nutrients and light. Trees and grass compete more strongly for water followed by nutrients, while competition for light is thought to be low (McIvor and Gardener 1995; McIntyre *et al*. 2002).

Other ecological studies have focused on the adverse impacts (or costs) of tree and vegetation clearing on a range of ecosystem services, either through on-site (paddock) or off-site (catchment) processes. These impacts are summarised in a Queensland landscape context through a series of articles in *The Rangeland Journal* (2002). The main ecological impacts include: biodiversity loss (McAlpine *et al*. 2002); increased greenhouse gas emissions (Henry *et al*. 2002); decline in nutrient availability and cycling (Schmidt and Lamble 2002); soil and water erosion (Ludwig and Tongway 2002) and increased soil and water salinity (Thorburn *et al*. 2002). The importance of balancing these broader ecological impacts and costs with the benefits from vegetation clearing for agricultural production is recognised from a resource economics perspective in terms of optimising the net societal benefits from such systems (Rolfe 2002).

The benefits of trees in silvopastoral systems can be linked with specific design features to capture a range of stimulatory and complementary effects on total output. The most common designs involve rows of trees described as tree strips, alley belts, shelterbelts or windbreaks. These types of tree strips are used by land managers to reduce wind speed and erosion, provide shelter and beneficial microclimate and increase soil moisture and plant growth in the pasture neighbouring the tree strip (McKeon *et al*. 2008). Woodlands with mature, scattered trees have also been shown to reduce wind speed by up to 50% (McIntyre *et al*. 2002).

Trees used strategically in the landscape can also provide direct benefits for animal production through provision of shade and shelter, particularly during periods of climatic stress and calving (Roberts 1984; Daly 1984; Bird *et al*. 1992). From a tropical perspective, a number of studies have evaluated the impacts of tree shading on nutrient cycling and pasture quality in northern and central Queensland (Wilson 1996, Jackson and Ash 1998, Ash and McIvor 1998; Jackson and Ash 2001). The studies concluded that shading enhanced soil fertility, forage nitrogen and pasture quality under the tree canopy.

The case of GHG abatement

Whilst the quantity of literature exploring the biophysical aspects of agroforestry is extensive, there have been limited studies reporting the financial impacts of adopting agroforestry systems that incorporate GHG sequestration and mitigation opportunities. In a review of the agroforestry literature relevant to North America, Weersink *et al*. (2003) concluded carbon could be sequestered by agriculture at a cost of \$10-35/ t CO2-*e.* Ford-Robertson *et al*. (1999) described a carbon stock and flow model and compared the net carbon balance over 80 years for grazing, agroforestry and afforestation land uses in New Zealand. The net carbon stock for a typical pasture system was substantially lower than for agroforestry and afforestation scenarios based on planting *Pinus radiata*, mainly due to sequestration of carbon in the trees and continuing methane emissions from livestock. For agroforestry systems, gains in total carbon stocks were lower than under afforestation, due to methane emissions and lower accumulated biomass carbon over time.

Shively *et al*. (2004) investigated the incremental costs of increasing carbon sequestration across the Manupali watershed in the Philippines using agroforestry and afforestation systems based on *Paraserianthes falcataria*. They found that the costs of carbon storage (or prices needed to compensate farmers for conversion to forestry based on the opportunity costs of the land for cropping) varied between \$3.30/t on fallow land to \$62.50/t on higher value cropping land. Importantly, carbon storage through agroforestry was less costly than through afforestation due to the addition of annual crops to compensate for some of the opportunity costs of land conversion.

From a forest products perspective, Venn (2005) investigated the financial and economic potential for plantations across Queensland for hardwood sawlog production. Where high growth rates are achievable (20-25 m^3/ha/yr), such as along the high rainfall coastal fringes of northern and southern Queensland, long rotation hardwood plantations were found to be profitable compared to agricultural land values. At intermediate (15 m^3/ha/yr) or lower growth rates (5-10 m^3/ha/yr), hardwood sawlog plantations were either viable under optimistic assumptions or marginal. However, the inclusion of broader social benefits such as carbon sequestration, salinity amelioration and other ecosystem services were seen to justify the establishment of plantations for most regions.

Harris-Adams and Kingwell (2009) estimated the marginal cost of abatement for agricultural shires offsetting their emissions through reforestation in Western Australia. The study identified abundant cost-effective sites for sequestration with the lowest cost sequestration estimated to cost \$18.60 / t CO2-*e.*

This paper expands the understanding of the economic implications of transitioning from a conventional grazing system to a silvopastoralism agroforestry system in central Queensland. It is intended that the findings from this study will contribute to more informed decisions on Australia's GHG abatement strategies.

Research method

The financial feasibility of silvopastoralism in central Queensland was evaluated using a discounted cash flow analysis and regional costs and prices for livestock products. Uncertainties in key variables including tree growth rates and product prices were incorporated using sensitivity analyses. A 1000 ha paddock on a regionally representative cattle property (e.g. average property area and herd size) was used to assess the economic performance of the silvopastoralism option versus business-as-usual (i.e. maintain a largely treeless grazing paddock). The resultant measure of financial viability (i.e. net present value, (NPV) were used to compare a silvopastoralism system to an extensive grazing management system under four different carbon accounting frameworks. The extensive grazing system and each of the four carbon accounting frameworks is defined below. The modelling sought to compare a traditional grazing property conducting a breeding and finishing cattle enterprise on two different vegetation communities (brigalow and eucalypt).

Option 1: Grazing – clear all regrowth (no carbon accounting)

The 1000 ha paddock in this scenario contained 10-year-old regrowth of brigalow (*Acacia harpophyll)* or popular box (*E. populnea)* which was pulled with a bulldozer and chain and raked 10 years previously (Figure 1 and Figure 4). The initial regrowth tree basal area was 5.5 m^2/ha at 30 cm height for brigalow or 3.2 m^2/ha at 30 cm height for popular box. A full description of the methodology used to estimate tree basal area is provided in Back *et al.* (1999).

Fig. 1 Brigalow (*Acacia harpophylla*) regrowth

The modeled paddock had a mature stand of buffel pasture in the case of brigalow and native pasture in the case of popular box, two watering points and a carrying capacity of 1AE[1] to 6 ha in the case of brigalow and 1AE to 10 ha for popular box. In the second year of the analysis for brigalow, all regrowth was blade-ploughed and the paddock spelled for six months (Figure 2).

Fig. 2 Paddock following blade ploughing to control regrowth

In the case of popular box, in Year 2 of the analysis, the regrowth was pulled, stick raked and the paddock spelled for six months. In Year 3 grazing was reintroduced and the carrying capacity for the treated paddock slowly declined over the following 23 years as regrowth competed with pasture for moisture, nutrients and sunlight. The stocking rate was adjusted to match the declining carrying capacity over the life of the analysis.

[1]An Adult Equivalent (AE) refers to a method of comparison between animals of different feed requirements with a recognised standard of a single adult animal feed ration. The international standard being a single non-pregnant, non lactating animal of 455 kilograms live weight. EQUALS 1 AE

Option 2: Retain regrowth strips – no carbon

The retain-regrowth-strips management option begins with the same 1000 ha paddock and 10-year-old regrowth (brigalow or popular box) as in management option 1. In Year 2 of the analysis, the regrowth was cleared (blade-ploughed for brigalow or pulled and chained for popular box) with regrowth strips 20 m wide left every 60 m (similar to Figure 3).

Fig. 3 Blade ploughed brigalow regrowth with regrowth strips

Fig. 4 Poplar box woodland 5-10 years after clearing

The paddock was spelled for 6 months and the carrying capacity for the property adjusted over the following 22.5 years as regrowth in the cleared and uncleared strips slowly increased.

Option 3: Retain regrowth strips accounting for sequestered carbon in strips

The retain-regrowth-strips management option begins with the same 1000 ha paddock and 10-year-old regrowth (brigalow or popular box) as management option 1. In Year 2 of the analysis, the regrowth was cleared (blade-ploughed for brigalow or pulled and chained for popular box) with regrowth strips 20 m wide left every 60 m (similar to Figure 3). The paddock was spelled for 6 months and the carrying capacity for the property adjusted over the following 22.5 years as regrowth in the cleared and uncleared strips slowly increased. Carbon sequestered in the retained strips was sold net of estimated livestock methane emissions. Carbon released from the clearing of regrowth in the inter-row zones was not included as a cost in the economic analysis. Instead it was assumed that any regrowth in the inter-row zone would be in a perpetual cycle of being cleared, regrowing and being cleared again.

Option 4: Retain regrowth strips accounting for sequestered and released carbon (short {10 year} linear decomposition period)

The retained regrowth strips accounting for released carbon option (short decomposition period) is similar to option 3 with the exception of how carbon is accounted for. In this example sequestered carbon is estimated for the entire 1000 ha paddock net of livestock methane emissions and carbon released as a result of clearing regrowth in the 60m inter-row zone. Carbon released as a result of clearing was calculated using the Intergovernmental Panel on Climate Change (IPCC) 10 year default linear decay model (Mackensen and Bauhus 1999). Carbon sequestered in the inter-row zone as regrowth from year 2 on is estimated and included in the annual carbon accounts.

Option 5: Retain regrowth strips accounting for released carbon (long {30 year for brigalow and 20 year for popular box} linear decomposition period)

The retained regrowth strips accounting for released carbon option (short decomposition period) is similar to option 3 with the exception of how carbon is accounted for. In this example sequestered carbon is estimated for the entire 1000 ha paddock net of livestock methane emissions and carbon released as a result of clearing regrowth in the 60m inter-row zone. Carbon released as a result of clearing was calculated using a decomposition turnover time of 30 years for brigalow and 20 years for popular box (Mackensen and Bauhus 1999). Carbon sequestered in the inter-row zone as regrowth from year 2 on is estimated and included in the annual carbon accounts.

Derivation of tree growth, biomass and timber product models

Relationships between time since clearing, stand basal area and regrowth height were generated for the brigalow and eucalypt land types from local data in central and southern Queensland. Table 1 provides a summary of the data used in the analysis. Donaghy *et al*. (2009) provided a detailed explanation of the relationships used in generating tree regrowth and plantation basal areas and height growth rates used in the analysis.

Table 1 Data source of regrowth, plantation basal area and height growth rates

Relationship	Data source	Comments
Brigalow stand basal area and time since clearing (tree strip – all data)	McKeon et al. (2008) Chandler et al. (2007) Scanlan (1991) Bradley (2006) and associated unpublished data	Data from southern and central Queensland
Brigalow stand basal area and time since clearing (blade-ploughed strip – <25yr)	Bradley (2006) and associated unpublished data. Using data points less than <25 years since clearing	Data from southern Queensland
Brigalow stand height and time since clearing	McKeon et al. (2008) Scanlan (1991) Bradley (2006) and associated unpublished data	Data from southern and central Queensland
Eucalypt stand basal area and time since clearing	McKeon et al. (2008) TRAPS woodland monitoring site data. Back *et al*. (2009) and Burrows et al. (2002) and associated unpublished data	Data predominately from poplar box (*Eucalyptus populnea)* woodland in central and southern Queensland. Two sites were ironbark (*E. melanophloia* and *E. crebra*) woodland
Eucalypt stand height and time since clearing	McKeon et al. (2008) TRAPS woodland monitoring site data. Back et al (2009) and Burrows et al. (2002) and associated unpublished data	Data predominately from poplar box (*Eucalyptus populnea)* woodland in central and southern Queensland Relationship poor
Plantation stand basal area since planting	Huth (2007)	Data from Central Queensland plantation species trials. Data calculated from individual stem basal area (average of the five best taxa at each site) multiplied by the number of stems at planting and following the two thinning operations in year 5 and 8 (**Error! Not a valid result for table.**)
Plantation stand height since planting	Huth (2007)	Data from Central Queensland plantation species trials. Data is an average of the five best taxa at each site

Pasture production and livestock carrying capacities

The modelling took into account the dynamic relationship between tree and pasture growth. McKeon *et al*. (2008) reported zones of constrained and stimulated pasture growth associated with tree strips which were not accounted for by the tree basal area. Relationships were modeled between relative pasture growth expressed as a percentage of pasture yield with no tree impact and distance from the edge of the tree strip measured in tree heights (e.g. Figure 5). Both relationships were used to derive the constrained and stimulated pasture production factors used in the bioeconomic modelling.

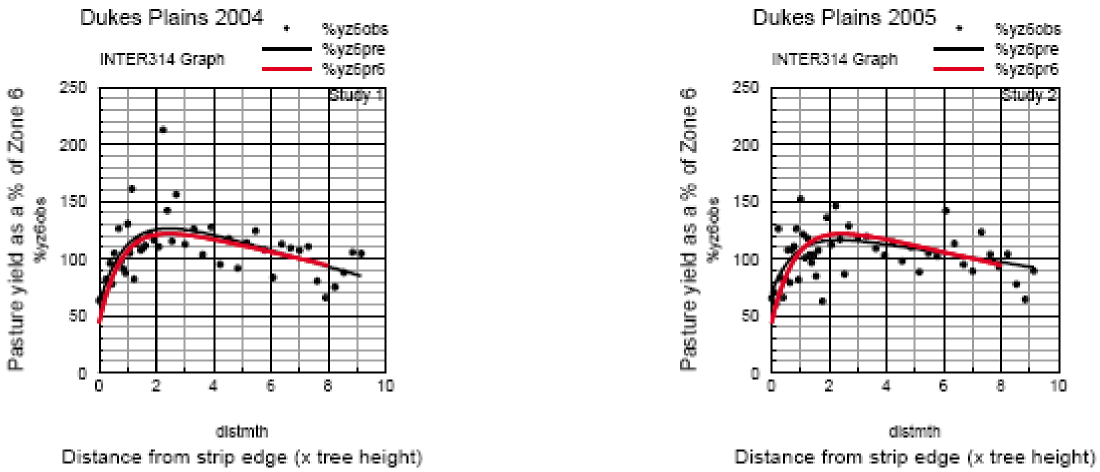

Fig. 5 Relationship between pasture yield expressed as a percentage of pasture yield with no tree impact and distance from edge of tree strip expressed in multiples of tree height. Source: McKeon *et al.* (2008).

Using these principles each modeled paddock was split into five zones (Figure 6) so pasture production and livestock carrying capacity could be estimated specifically for each zone. The zones were dynamic with zonal width changing as the height of the trees in the strips grew each year. The prevailing winds blow from the left of the diagram to the right.

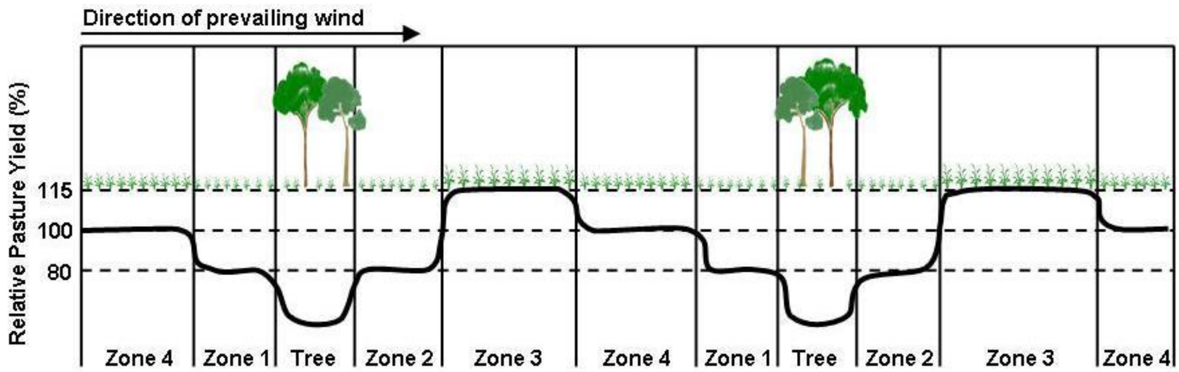

Fig. 6 Schematic diagram of the different zones modelled and the relationship to relative discount or stimulation of forage production in the cleared strips

Table 2 defines the width of each zone and the corresponding suppression or stimulation factor applied to pasture production.

Table 2 Width, constraint and stimulation factors for different zones where strips of regrowth have been cleared or retained (see Fig. 6).

Zone	Width	Relative pasture yield
Tree	Retain regrowth strip width	Based on tree strip basal area
Zone 1	1 times tree height	Discounted by 0.8 of cleared strip basal area
Zone 2	1 times tree height	Discounted by 0.8 of cleared strip basal area
Zone 3	4 times tree height	Stimulated by 1.15 of cleared strip basal area
Zone 4	Remaining width of cleared strip	Based on cleared strip basal area

Pasture production was estimated using tree basal area and pasture production relationships derived from GRASP pasture modelling and extracted from the StockTake database (DPI 2004) (Figure 7). The brigalow/blackbutt and poplar box with shrubby understorey land types were modelled in GRASP using climate data drawn from a data drill for Bombandy station (located north of the Middlemount township in central Queensland).

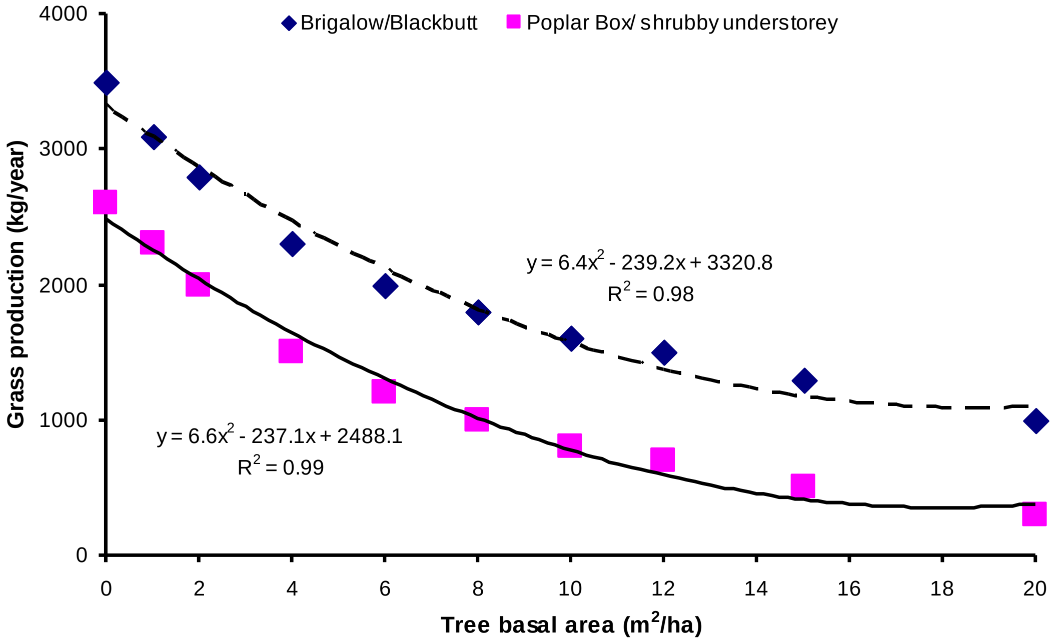

Fig. 7 Relationship between tree basal area and grass production for the brigalow/blackbutt and poplar box with shrubby understorey land type

For each paddock zone, pasture production (kilograms of dry matter) per hectare was estimated annually based on the tree basal area in the zone and applying the associated stimulation or suppression factor for the zone. The livestock carrying capacity was calculated assuming a 25% utilisation rate and 10 kg dry matter per day intake. The total number of livestock carried for that year was the sum of the carrying capacity for each zone by the area of that zone in the paddock. This analysis assumes an even utilisation rate and a matching of livestock numbers to forage production so that land condition was maintained or improved. The modelling also assumes no seasonal variation in rainfall and pasture production.

Estimating decomposition rates of coarse woody debris

Currently the release of carbon from land use change (including clearing) is calculated using the intergovernmental panel on climate change (IPCC) default 10 year linear decay model. However the IPCC default decomposition model does not take into account a range of variables linked to decay such as size of distribution litter, density, content of extractives and the situation in which wood is decaying (Mackensen and Bauhus 1999). By referring to international studies on decomposition rates and to durability studies on Australian timbers Mackensen and Bauhus (1999) concluded that an assumed turnover time of 10 years for all litter may be a considerable overestimation of decay rates if the majority of the litter was in the form of coarse woody debris (CWD), and concluded that CWD-turnover in Australia would in most cases exceed 25-30 years.

North of 30° latitude Mackensen and Bauhus (1999) found durability was found to decrease substantially due to the increasing importance of wood destroying agents such as termites. As a result turnover times were adjusted down to >30, 20, 11 and 4 years for the durability classes 1,2,3 and 4 (Mackensen and Bauhus 1999). For the modelling reported here a turnover period of 30 years and 20 years was assumed for brigalow and popular box respectively.

Financial analysis

For this study a standard discounted cash flow (DCF) investment analysis was used to evaluate the proposed farming practice changes where capital investment is required. The DCF analysis estimates the NPV or lump sum present value equivalent of the incremental net cash flow stream over an investment period (e.g. 25 years). It arises directly as a result of estimating the difference in the annual cash flow pattern for the property, with and without any proposed changes in management options. The net present value is calculated as:

$$NPV = \sum_{t=1}^{n} \frac{C_t}{(1+r)^t}$$

where n = number of periods in the investment
r = the discount rate
t = the year of the cash flow
C_t = cash flow at year t

The economic analysis reported here compares the net present value of conventional grazing systems to a range of alternative scenarios. The analysis takes into account regrowth clearing costs, carbon budgeting, changes in pasture production and carrying capacities as a result of changes to tree basal area and herd gross margins to estimate the expected cash flows and economic returns from each production system.

In options 3, 4 and 5, sequestered carbon sales (net of livestock methane emissions) are included in the analysis. To determine the relative profitability of the conventional grazing system (option 1) to each of the alternative systems, the NPVs of management options 2, 3, 4, and 5 are compared with the returns of the conventional grazing systems (option 1) based on net present value. A 6% discount rate was used for the analysis.

Costs and prices

All clearing costs are based on industry estimated contractor rates. In each analysis it was assumed that the land was already owned and used for extensive grazing – that is, the sale and purchase of the land was not included in any of the comparative partial budgets.

Gross margins per adult equivalent including interest on livestock capital ($155.65/AE for brigalow land and $105.33/AE for eucalypt land) were sourced from Best (2007). It was assumed for Scenarios 3, 4 and 5 that any sequestered carbon would be valued at $10/t CO^2e. Transaction costs associated with the sale of sequestered carbon and the continued monitoring and reporting of carbon stocks were not included in the analysis. Sequestration rates were based on changes in estimated annual tree basal area and above and belowground allometrics (Scanlan 1991, Burrows *et al*. 2002; Zerihun *et al*. 2006). Livestock methane emissions were estimated to be 1.5t CO2-*e*/yr per adult equivalent (Charmley 2009), pers.comm., 20th May). Results

A summary of the financial consequences of choosing to retain regrowth strips and continue grazing (with and without carbon sales) are presented in table 3.

Table 3 Results Summary

Land type	Option 1 NPV of clearing and grazing (no carbon accounting)	Option 2 NPV of retaining regrowth strips (no carbon accounting)	Option 3 NPV of retaining regrowth strips and selling sequestered carbon (excluding carbon released from clearing)	Option 4 NPV of retaining regrowth strips and selling sequestered carbon assuming short (10 year) CWD turnover	Option 5 NPV of retaining regrowth strips and selling sequestered carbon assuming long (30 or 20 year) CWD turnover
Brigalow land type	$268,391	$253,659	$317,212	$189,259	$310,808
Eucalypt land type	$119,853	$118,152	$238,901	$393,594	$416,858

The decision to clear all the timber and continue grazing (option 1) resulted in a NPV of $268,391 for brigalow and $119,853 for popular box. Clearing and retaining regrowth strips 20m wide every 60m for 25 years for grazing purposes only (management option 2) actually left the grazier $14,732 worse off in the case of brigalow and $1,701 worse off in the case of eucalypt land.

The inclusion of a carbon sequestration budget dramatically altered the outcomes. Figures 8 and 9 present the modelled annual carbon stocks for options 3,4 and 5 for brigalow and poplar box incorporating released carbon from clearing, sequestered carbon from regrowth and livestock methane emissions.

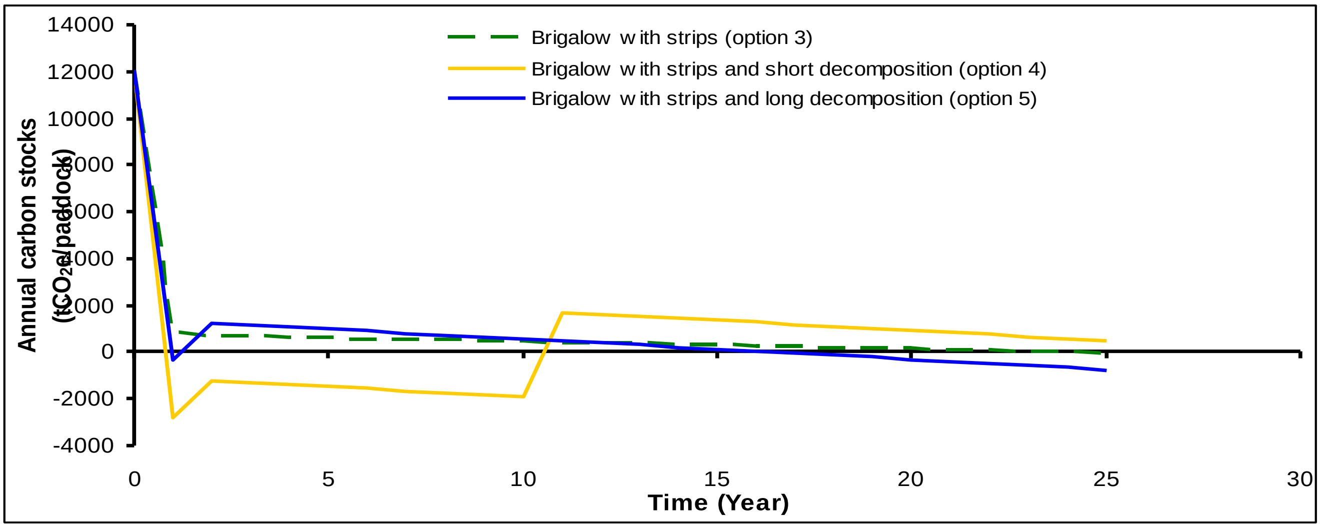

Fig. 8 Brigalow annual carbon stocks $tC0_2$-e for 1000 ha. Annual carbon stocks under 3 alternative carbon accounting frameworks incorporating livestock emissions, carbon released from clearing and carbon sequestered as regrowth

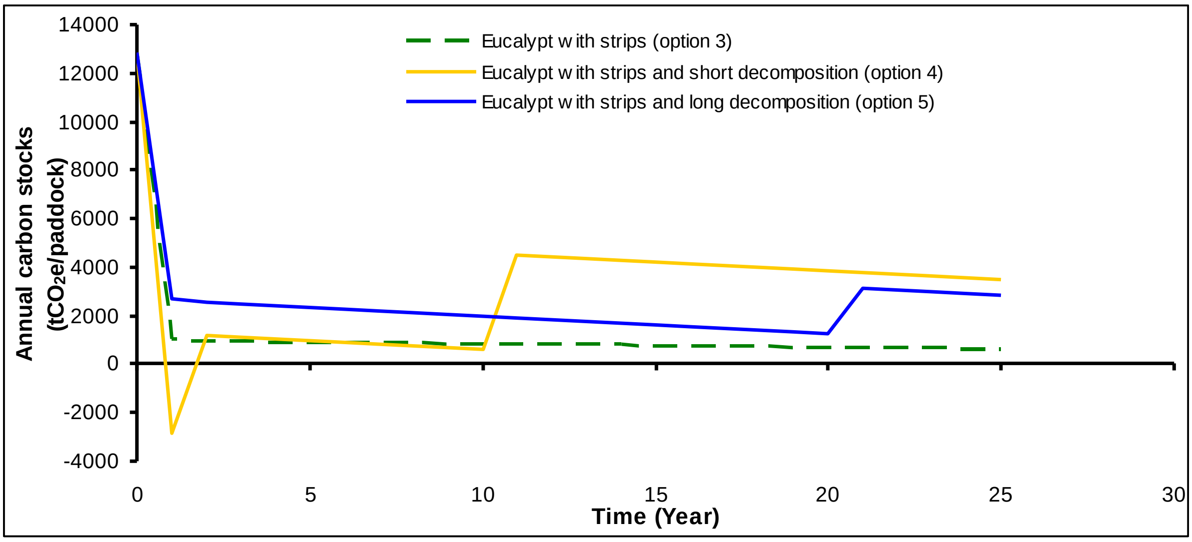

Fig. 9 Popular box annual carbon stocks $tC0_2$-e for 1000 ha. Annual carbon stocks under 3 alternative carbon accounting frameworks incorporating livestock emissions, carbon released from clearing and carbon sequestered as regrowth

At \$10/t CO_2e the grazier would be \$48,821 better off over 25 years retaining tree strips, continuing to graze and selling any sequestered carbon from brigalow land (option 3). In the case of popular box, higher rates of sequestration and lower opportunity costs from foregone grazing translate into higher NPVs. At \$10/t CO_2-e the grazier is \$119,048 better off retaining tree strips and selling sequestered carbon.

If the grazier was required to account for carbon released from the routine clearing of regrowth using the IPCC's linear decay model prior to selling any sequestered carbon (option 4) the grazier could be significantly worse-off or better-off depending on whether the land was brigalow or popular box. In the case of brigalow the grazier would be \$79,132 worse-off under option 4 or \$274,000 better-off if the land was popular box. Extending the decomposition period (option 5) and hence the period over which the release of carbon from clearing is accounted for in the cashflow again dramatically alters the outcome. In the case of brigalow the grazier would be \$42,417 better-off and for popular box \$297,005 better-off when compared to option 1.

Discussion and Conclusions

Rangeland grazing research has previously focused on the direct impacts of animal stocking rate and tree basal area on pasture biomass and livestock production, with an emphasis on the competitive effects of tree density on pasture growth. This focus essentially regards woody vegetation (i.e. trees) as an impediment to grazing profitability. The promising results presented here for alley belt systems capture the holistic value of multiple-use grazing systems compared to grazing only systems by incorporating carbon sequestration benefits. For these scenarios, encouraging natural regrowth is a potentially valuable activity that gives rise to the combined natural resource management benefits associated with increased trees in the landscape, including soil and water function, carbon sequestration and biodiversity.

The results of the bioeconomic modelling allow several important conclusions to be drawn. First, there is no financial incentive for landholders to retain natural regrowth strips in the absence of carbon payments even after the positive impacts of trees on pasture growth are accounted for. Second, the recognition of carbon benefits at even low carbon prices ($10/t CO_2-e) is sufficient to make the retention of regrowth strips financially viable for landholders, even after the methane emissions for associated livestock are accounted for. Third, the net benefits of carbon sequestration in regrowth strips are higher on lower productivity country for example eucalypt woodlands.

The results also demonstrate how dramatically the rules underpinning a carbon accounting framework can alter the financial impact of graziers choosing to participate in a carbon market. Retaining regrowth strips and selling carbon without the need to offset carbon released from clearing (option 3) is likely to result in regrowth strips being retained on both fertile (brigalow) and less fertile (popular box) grazing lands. If landholders were required to account for methane and clearing emissions prior to selling sequestered carbon using the IPCC turnover period of 10 yeears, it would be unlikely regrowth strips would be retained on brigalow land and highly likely that landholders with poplar box country would choose to forgo some grazing potential in return for selling sequestered carbon. Extending the CWD period to more accurately reflect Australian conditions (option 5) is likely to result in some grazing potential being replaced with retained regrowth strips on both the brigalow and popular box land.

There are some caveats to note with the analyses presented in this paper, concerning CWD decomposition rates, impacts of degraded land, identified data gaps and transaction costs.

- The absence of accurate decomposition data for cleared forests in Australia makes it difficult to estimate the rate of carbon released from cleared regrowth. Its also unclear from the literature if the decomposition rates that are available are relevant to remnant forests only or regrowth and remnant forests.
- All modelling undertaken assumes the land is in reasonable condition. For degraded land the analysis would be expected to favour the adoption of silvopastoral systems.
- A major constraint to the analysis is the availability of relevant data and statistically significant relationships for regrowth, tree heights and above and below ground allometrics. Caution should be used in extrapolating these results beyond central Queensland.
- The only carbon emissions included in the analysis were methane emissions and carbon released from the clearing of regrowth. No attempt was made to incorporate carbon emissions from the use of fossil fuels, fertilizers, feed supplements and electricity use on farm.
- A potential impediment to silvopastoralism being viewed as an efficient GHG offset strategy is transaction costs particularly for small landholders. Carbon sequestered in agroforestry projects needs' to be accounted for in a way that ensures carbon charges are real, directly attributable to the project and in addition to what would have occurred in the absence of the project (Cacho and Lipper 2007). The effort required by market participants, both buyers and sellers, to meet, communicate, exchange and validate information represents the transaction costs of buying or selling sequestered carbon. Transaction costs have not been included in the analysis reported here.

Whilst the analysis reported here highlights the opportunities for Australia to use silvopastoral systems as a voluntary mechanism to meet its emissions abatement targets, silvopastoralism remains under-recognized as a GHG reduction strategy and income diversification opportunity. One reasons for this is an apparent lack of scientific and economic data available to policy-makers, graziers and extension professionals.

Schoeneberger (2008) identified a similar extension constraint in the USA where agroforestry's cross cutting nature put it at the interface of agriculture and forestry where it was not strongly supported or promoted by either. Overcoming these challenges is critical to silvopastoralism being integrated into the broader scope of sustainable agricultural management and viewed as a legitimate means of obtaining 'bankable' carbon.

Given the contribution of agriculture and land-use change to Australia's GHG emissions, the results presented here are important in a policy context. The beef industry operates over large areas of land suitable for silvopastoralism activities. The financial results reported suggest there are net benefits for landholders who integrate complementary carbon sequestration activities into existing grazing operations at even modest carbon prices dependent on the rules of the carbon accounting framework used. A key policy implication is that there appears to be opportunities to engage graziers in biosequestration activities at relatively low cost levels. The rules of the carbon accounting framework are likely to determine not only landholder participation but also the landtypes used to sequester carbon.

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