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Implications of retaining woody regrowth for carbon sequestration for an extensive grazing beef business: a bio-economic modelling case study

Giselle Whish[A,C], *Lester Pahl*[A] *and Steven Bray*[B]

[A]Queensland Department of Agriculture and Fisheries, Toowoomba, Qld 4350, Australia.
[B]Queensland Department of Agriculture and Fisheries, Rockhampton, Qld 4702, Australia.
[C]Corresponding author. Email: giselle.whish@daf.qld.gov.au

Abstract. A bio-economic modelling framework (GRASP-ENTERPRISE) was used to assess the implications of retaining woody regrowth for carbon sequestration on a case study beef grazing property in northern Australia. Five carbon farming scenarios, ranging from 0% to 100% of the property regrowth retained for carbon sequestration, were simulated over a 20-year period (1993–2012). Dedicating regrowth on the property for carbon sequestration reduced pasture (up to 40%) and herd productivity (up to 20%), and resulted in financial losses (up to 24% reduction in total gross margin). A net carbon income (income after grazing management expenses are removed) of \$2–4 per t CO_2-e was required to offset economic losses of retaining regrowth on a moderately productive (~8 ha adult equivalent^{-1}) property where income was from the sale of weaners. A higher opportunity cost (\$ t^{-1} CO_2-e) of retaining woody regrowth is likely for feeder steer or finishing operations, with improved cattle prices, and where the substantial transaction and reporting costs are included. Although uncertainty remains around the price received for carbon farming activities, this study demonstrated that a conservatively stocked breeding operation can achieve positive production, environmental and economic outcomes, including net carbon stock. This study was based on a beef enterprise in central Queensland's grazing lands, however, the approach and learnings are expected to be applicable across northern Australia where regrowth is present.

Additional keywords: beef production, carbon farming, carbon price, livestock emissions, rangelands, tree regrowth.

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Introduction

The impact of anthropogenic emissions of greenhouse gases (GHG), such as carbon dioxide, methane and nitrous oxide, on the climate system is an issue of global concern due to its impacts on climate, human wellbeing and natural systems (IPCC 2014). Globally, the livestock sector contributes between 8% and 11% (O'Mara 2011) and 14% (Gerber *et al*. 2013) of anthropogenic GHG emissions, of which ~5% can be attributed to enteric methane (Gerber *et al*. 2013).

Mitigating anthropogenic GHG emissions has become a focus for governments and communities around the world, and several carbon markets have been established to assist countries achieve reductions in GHG emissions and climate change mitigation (Dargusch and Harrison 2011; Cowie *et al*. 2012). The Australian Government's voluntary carbon (C) offsets scheme (Emissions Reduction Fund – ERF) was established to create positive incentives for Australian businesses to adopt practices that reduce both their operational costs and C emissions (Commonwealth of Australia 2015*a*). This scheme may provide an opportunity for primary producers to earn additional income by reducing atmospheric GHG.

Livestock enteric fermentation is responsible for ~10% of Australian total GHG emissions (Commonwealth of Australia 2015*b*), with almost half (45%) these emissions generated from the northern cattle industry (Charmley *et al*. 2011). Over half the ~29.3 million head of beef cattle nationally graze ~250 million hectares of predominantly native grasslands, savannas and grassy woodlands of northern Australia (MLA 2014; Australian Bureau of Statistics 2015; State of Queensland 2015*a*). Much of the woodlands and forests in central and southern Queensland have been cleared to improve forage production for livestock (State of Queensland 2015*b*), however, recurrent regrowth control is required for the dominant *Acacia* and *Eucalyptus* tree species and many other sub-dominant species (Scanlan *et al*. 1991).

In the extensive tropical grazing systems of northern Australia there are relatively few options for the abatement of methane and nitrous oxide emissions from ruminant production (Eckard *et al*. 2010). The main options available include management changes that increase herd production efficiency (improved fertility, culling unproductive animals) and individual animal production efficiency (faster growth rates and reduced days to slaughter), both of which decrease the methane emissions per kg saleable product or emissions intensity (Charmley *et al*. 2008; Henry *et al*. 2012; Hristov *et al*. 2013).

However, extensive grazing systems contain large stocks of C in vegetation (Bray and Willcocks 2009) that provide a potential sink for C (Moore *et al*. 2001; Donaghy *et al*. 2010; Ryan *et al*. 2015). Although clearing of remnant trees to increase pasture yields has been widely practiced in Queensland (Bortolussi *et al*. 2005), these trees often regrow from stumps and roots, forming stands of regrowth that require recurring clearing (Scanlan 1988). Retention of this woody regrowth for C storage could provide an alternative revenue stream to bolster enterprise profitability (Henry *et al*. 2012).

Simulation models have value in complementing and extending empirical research, which seeks to evaluate a range of management options. As livestock production systems are large and complex sources and sinks of GHG, whole-system models that capture the dynamic interactions between animals, climate, soil, and plants are needed to effectively evaluate management options (Eckard *et al*. 2014). Additionally, consideration of management options at the property scale is often needed to evaluate the cost-effectiveness, feasibility and the potential trade-off effects of mitigation measures (van Groenigen *et al*. 2008). An ideal biophysical model would be capable of predicting the influence of management changes on GHG sources and sinks, economic production and the consequences for other ecosystem services (Moore *et al*. 2014). However, a balance must be found between complexity and practicality of models (Eckard *et al*. 2014).

An integrated bio-economic modelling (BEM) framework (GRASP-ENTERPRISE) was used to capture the interactions between pastures, animals and climate, and to evaluate the economic and C outcomes of retaining woody vegetation for C sequestration.

GRASP, a point-based model that uses daily climate inputs to simulate soil-water balance, aboveground grass growth and animal production, has been widely used in the semiarid tropical grazing lands of northern Australia (McKeon *et al*. 2000; Rickert *et al*. 2000). In these rangeland environments, GRASP has been used to evaluate the impact of different stocking strategies on pasture and animal productivity (McKeon *et al*. 2000), to estimate safe carrying capacities for current and future climates (e.g. Johnston *et al*. 1996; McKeon *et al*. 2009; Whish *et al*. 2014), and to examine the dynamics of grazed woodlands and their effect on GHG emissions (Moore *et al*. 2001). Recently, the effects of grazing management practices on native pastures, livestock production and resource condition were evaluated using a modified version of GRASP (Scanlan *et al*. 2014; Pahl *et al*. 2016).

Simulated outputs from GRASP are used in the dynamic, multi-paddock herd and economic model ENTERPRISE to predict the impacts of changes in management on herd productivity and property economics (MacLeod and Ash 2001). The ENTERPRISE model constructs a herd consistent with estimated branding and mortality rates, the simulated stocking rates from GRASP, and the buying/selling rules within ENTERPRISE. The GRASP-ENTERPRISE linked models have been used to evaluate the implications of grazing management strategies for beef businesses in northern Australia (MacLeod *et al*. 2011; Scanlan *et al*. 2014; Whish *et al*. 2015).

This paper uses a BEM framework (GRASP-ENTERPRISE) to predict the consequences of retaining woody regrowth for C sequestration on GHG sources and sinks and economic production. Through the application of this modelling approach to a case study property in northern Australia, this paper provides further details on the potential implications for incorporating C farming practices into a beef grazing enterprise, including the opportunity costs of retaining woody regrowth for C sequestration.

Methods

This paper uses a case study property to predict the biological, economic and C outcomes of retaining woody regrowth for C sequestration. Landscape characteristics and resource information collected from the property (on-ground measurements, herd composition, herd husbandry, stocking rates and grazing management practices) in combination with remotely sensed tree-cover and land-type spatial data, were used to inform and calibrate the GRASP and ENTERPRISE BEM.

Case study property

The beef grazing property (20°21′S, 149°21′E) was located in central Queensland's eucalypt woodlands where the majority of annual rain (long-term average 653 mm) falls during the summer months, although there is considerable variability from year to year (co-efficient of variation 32%). The property consisted of 10 150 ha of predominantly box (*Eucalyptus populnea*), narrow-leaved and silver-leaved ironbark (*E. crebra*, *E. melanophloia*) and bulloak (*Allocasuarina luehmannii*) land types (Whish 2011; Fig. 1).

BEM – property landscape

The case study property was represented in the BEM as homogenous landscape units (land type, pasture condition, woody vegetation structure and cover). As such, the modelled property comprised 13 unique combinations of five land types, three vegetation structures (pulled regrowth, remnant, tebuthiuron herbicide treated) and two pasture conditions ('B' = fair and 'C' = poor). Tree basal area (TBA), or the cross-sectional area at a height of 1.3 m of all live trees in a stand, was estimated from a foliage projective cover spatial dataset (State of Queensland 2015*c*) and on-ground assessments, and varied across the property from 0.1 to 11 m^2 ha^{-1} (Fig. 1). The size of each paddock was proportional to the area of each unit present on the case study property (Table 1).

BEM – management for regrowth practices

The management practices used on the case study property to control regrowth included the application of the herbicide tebuthiuron and pulling (using a chain between two bulldozers to uproot the trees) in combination with regular burning regimes. It has been estimated (Back *et al*. 2009; Donaghy *et al*. 2009) that with regular burning practices woody vegetation would regrow to 1.5 TBA m^2 ha^{-1} after 10 years following mechanically

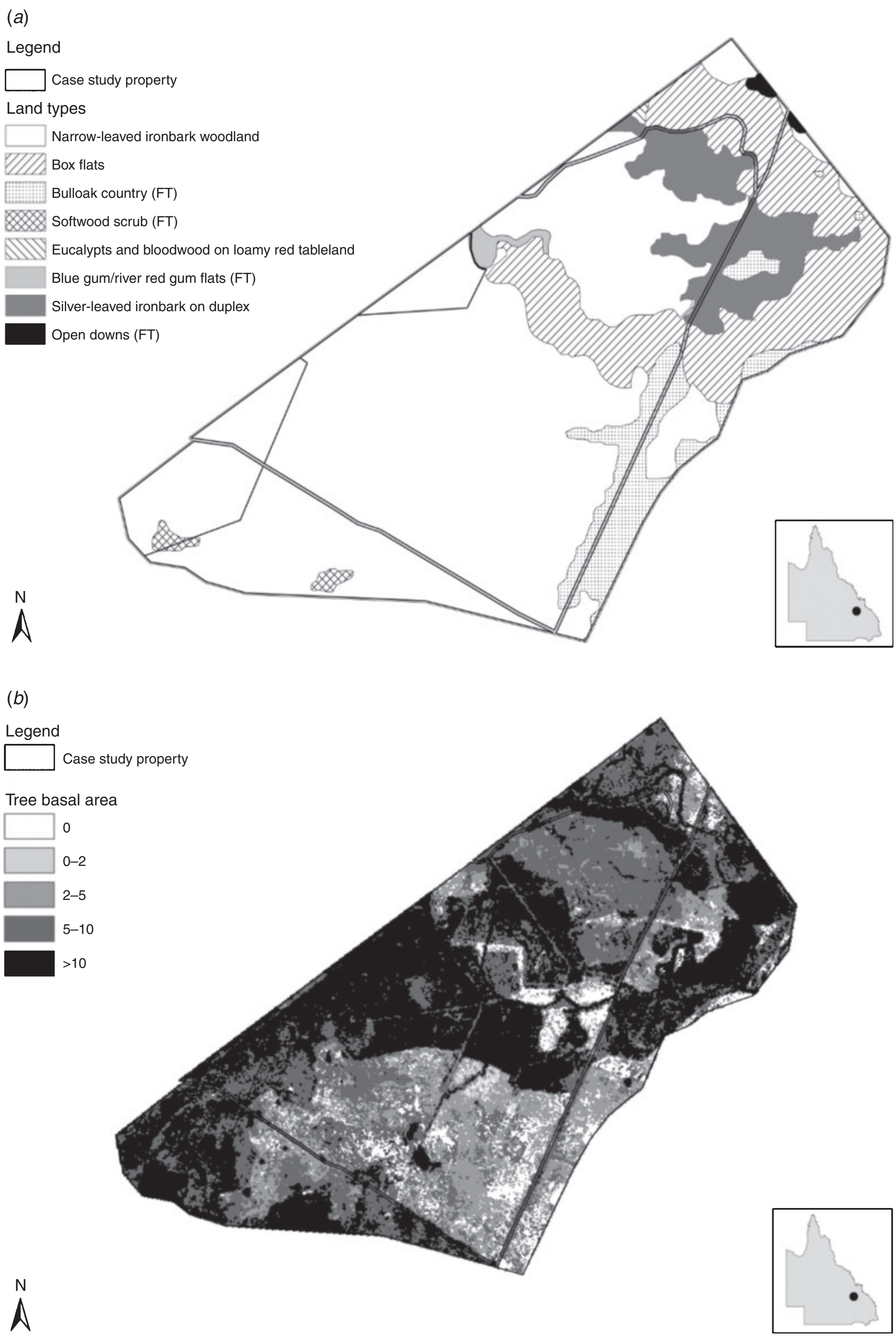

Fig. 1. The (*a*) land types and (*b*) tree basal area m^2 ha^{-1} on the case study property located in central Queensland eucalypt woodlands.

pulling vegetation, and to 1.5 TBA m^2 ha^{-1} after 30 years following application of tebuthiuron. Unless otherwise indicated, regrowth TBA are for a measured height at 130 cm.

Remnant areas were not burnt and a constant TBA was simulated. A simple tree growth sub-model was used in the GRASP model where TBA could change as a function of time

Table 1. The maintenance stocking rate and carrying capacity (AE) for each modelled paddock with specified land type, vegetation structure, tree basal area (TBA) and pasture condition

New paddock	Land types	Vegetation structure	TBA m^2 ha^{-1}	Pasture condition B = fair C = poor	Area ha	Maintenance stocking rate ha AE^{-1}	AE
1	Box flats and blue gum	Pulled regrowth	1.5	B	420	4.8	88
2	Box flats and blue gum	Remnant	9	C	850	16.7	51
3	Bulloak	Pulled regrowth	2.5	C	360	11.5	31
4	Bulloak	Pulled regrowth	2.5	B	1120	7.7	146
5	Bulloak	Remnant	11	C	1170	24	48
6	Gum-topped box	Graslan treated	0.1	B	1380	5.4	255
7	Gum-topped box	Pulled regrowth	0.2	C	1000	5.6	180
8	Gum-topped box	Pulled regrowth	2	C	500	7.7	65
9	Gum-topped box	Remnant	8.5	C	820	15	55
10	Narrow-leaved ironbark	Pulled regrowth	2.5	C	220	11.1	20
11	Narrow-leaved ironbark	Remnant	9	B	1220	14	90
12	Silver-leaved ironbark	Remnant	11	C	830	18.2	46
13	Silver-leaved ironbark	Remnant	9	B	260	10	26
				Total area	10 150		1101

and fire (J. Scanlan, pers. comm.). The dynamic tree model requires specification of a starting TBA, a fixed annual rate of increase, and a maximum TBA. Tree basal area can be reduced if there are fires. GRASP was parameterised so that regrowth paddocks were burnt once every 4 years and re-pulled when a TBA of 1.5 m^2 ha^{-1} or greater was reached. The effectiveness of a burn controlling regrowth was assumed to be the same for all land types, and was dependent on the amount of total standing dry matter (TSDM) kg ha^{-1} available at the time of the burn. A minimum TSDM fuel load of 800 kg ha^{-1} was required for a burn to be achieved, whereas fuel loads of at least 2500 kg ha^{-1} were required for maximum tree death. Over the 20-year simulation period, paddocks that were mechanically cleared and regularly burnt required re-pulling after 8–10 years. No further application of tebuthiuron herbicide was required.

Regrowth in paddocks designated for C farming was allowed to grow unchecked (not pulled nor burnt) and retained for C sequestration. Regrowth will compete for water more strongly than a mature tree as a greater leaf biomass is supported per unit of TBA for small trees (regrowth) than for mature trees (Scanlan 1991, 2002). As the competitive effect of regrowth relative to mature trees may be a third to two-thirds greater (J. Scanlan, pers. comm.), all regrowth TBA was multiplied by a factor of 1.5 in this study.

The cost of burning regrowth (\$2.70 ha^{-1}) was included in the ENTERPRISE model. The cost of clearing and stick-raking regrowth (\$65 ha^{-1}) was treated as a capital cost in the ENTERPRISE model, and as such, these costs were deducted from the calculated total gross margins (TGM).

BEM – property stocking rates

The model was calibrated to carry a ‘safe’ number of breeders and heifers, as defined by Johnston *et al*. (1996), so that the pasture condition was maintained over 20 years (1993–2012) for the specified tree cover and management practices to control regrowth. Pasture condition, defined as the capacity of the pasture to respond to rain and produce useful forage under the ABCD framework (MLA 2007), was quantified in terms of the percentage of perennial grasses in the GRASP model. Hence, pasture with ~60% perennial grasses was considered in fair (B) condition, whereas a poor condition pasture had ~20% perennial grasses (see Scanlan *et al*. 2014; for further detail). With a ‘safe’ stocking rate the average pasture condition (percentage of perennial grasses) over a 20-year simulation period approximated the initial condition. The majority (~57%) of the modelled property was in poor (C) condition, with half of the property burnt regularly to control regrowth. Simulated stocking rates for the five land types ranged between 5 ha AE^{-1} (adult equivalent 450-kg *Bos taurus* steer at maintenance) on box flats and gum-topped box through to 11–18 ha AE^{-1} on ironbark and bulloak (Table 1). The simulated stocking rates were in accordance with those recommended in the literature (see Whish 2011 for relevant sources) and were consistent with the owner-estimated long-term carrying capacities.

BEM – breeding operation

The version of the ENTERPRISE model described in MacLeod *et al*. (2011) was used to compare the herd and economic outcomes of the C farming scenarios. This model was customised to represent the case study property, including its herd management practices, classes of cattle present, their weights and rates of mortality and branding, and expenditure and income.

The case study property ran a self-replacing breeder herd consisting of ~850 breeder cows, which produced 620 weaners annually, 34 bulls, and 138 1-year-old replacement heifers. Heifers were mated at ~2.5 years of age, and cows were culled after eight breeding seasons.

This property operates as one component of several family-owned properties, where all weaner steers were transferred to other family-owned properties for fattening at a liveweight transfer price of \$1.99 kg $^{-1}$. As such, the modelled property only contained breeder cows, calves at foot, bulls and replacement heifers.

Surplus-to-replacement heifers were sold at a liveweight transfer price of \$1.73 kg^{-1} when they were between 1 and 2 years of age. Other sale animals included old bulls (\$1.28 kg^{-1}), cast-for-age cows (\$1.16 kg^{-1}) and up to 35% of cows (\$1.39 kg^{-1}) that did not produce a calf each year. The number of bulls carried each year equalled 4% of the number of breeder cows present.

Annual branding and mortality rates were estimated in ENTERPRISE using equations (refer to Scanlan *et al*. 2013) that are a function of the annual steer liveweight gain predicted by GRASP.

The carrying capacity targets for each paddock as predicted by GRASP were achieved in ENTERPRISE through a combination of recruitment of replacement heifers, and sales of cast-for-age cows, non-reproductive cows and surplus heifers.

The economic metrics used to compare C farming scenarios were average annual TGM, average annual gross margin AE^{-1}, and average annual profit. Annual TGM was calculated from the annual income from cattle sales minus annual direct or operating costs such as mustering, identification tags, freight and marketing, and veterinary/husbandry. Average annual profit was the TGM minus over-head costs such as rates, lease payments, maintenance, plant and equipment, and insurance.

Simulations

Five C farming scenarios, that ranged from 0% to 100% of the property regrowth being retained for C farming (Table 2), were simulated over a 20-year period (1993–2012) using historic climate records accessed from Scientific Information for Land Owners (SILO) climate database (Jeffrey *et al*. 2001). Regrowth occurred on 5000 ha, being 49% of the case study property. The regrowth paddocks required for each C farming scenario were determined according to the proportion of the modelled property to be retained and designated for C farming (Table 2).

Each paddock of the modelled property was stocked with cattle at a 'maintenance stocking rate', which ensured there was adequate fuel loads (a minimum of 800 kg ha^{-1} of forage) in the regrowth paddocks to achieve five burns over the 20 years.

Each C farming scenario model within ENTERPRISE was calibrated to match the total herd size as determined by GRASP each year. In some years between 10% and 20% of the weaner heifers were transferred off the property at the time of weaning to prevent the annual heifer cohort from exceeding the carrying capacity of the heifer paddock. Also, it was necessary to vary the culling rates of non-reproductive cows and retention rates for mature cows to ensure that the ENTEPRISE model achieved the herd size stipulated by GRASP for each year of each C farming scenario. Average annual TGM, profit, and gross margin per AE (GM AE^{-1}) were used to compare the C farming scenarios.

Greenhouse gas emissions and C stocks

Results from the BEM were used to estimate the property-level GHG emissions and sequestration of C for the five C farming scenarios. For this study, the only bio-sequestration considered was by aboveground vegetation growth (trees and pasture), whereas emissions included enteric fermentation from cattle and burning of vegetation. Soil organic C was not considered because an experimental trial at the case study property indicated large within-treatment variability and differences were not significant nor consistent between the grazing and woody vegetation treatments (Bray *et al*. 2015). Any change in C stocks in remnant vegetation was not included in the calculations. Average annual and accumulated totals for the 20 years were calculated for the livestock and savanna burning C emissions and sequestered C in regrowth, and reported as carbon dioxide equivalents (t CO_2-e). Tree regrowth C stocks (t CO_2-e ha^{-1}) were calculated on the area of regrowth retained on the property. Calculations of livestock and savanna burning emissions and pasture C stocks (t CO_2-e ha^{-1}) were based on the total property area (10 150 ha).

In paddocks designated for C farming, the C stocks in woody regrowth were calculated as half that present in Year 1 plus any C sequestered through tree growth over 20 years. This approach provided a simple estimation of the amount of initial tree biomass eligible for sequestration in an area that had been managed for pastoral use, and was based on the exposure draft methodology available at the time work was undertaken (Butler *et al*. 2012).

Paddocks in which regrowth was treated (chemically or pulled), and burnt were not included in the calculation of the C stocks of trees. Predicted TBA (m^2 ha^{-1}) at 130 cm height was multiplied by 1.53 (Krull and Bray 2005) to convert to TBA (m^2 ha^{-1}) at 30 cm. Burrows *et al*. (2002) derived a standing biomass allometric (6.286 t biomass m^{-2} at 30 cm) for remnant

Table 2. The proportion of the modelled property that is composed of regrowth and the regrowth paddocks that are retained for the each of the modelled carbon farming scenarios

Regrowth paddock	Hectares (ha)	Proportion of property	Modelled carbon farming scenarios (% of regrowth retained)				
			1 (0%)	2 (25%)	3 (50%)	4 (75%)	5 (100%)
1	420	8		–	–	–	X
3	360	7		–	–	–	X
4	1120	22		–	–	X	X
6	1380	28		X	X	X	X
7	1000	20		–	X	X	X
8	500	10		–	–	–	X
10	220	4		–	–	X	X
Total	5000	49	0%	28%	48%	74%	100%

eucalypt woodland. In this study, regrowth biomass was calculated as two-thirds that of remnant woodland to account for regrowth size class distribution (Bray and Willcocks 2009), and hence a factor of 4.15 t biomass m^{-2} at 30 cm was used to determine regrowth biomass (t ha^{-1}). Both tree (t ha^{-1}) and pasture (TSDM t ha^{-1}) biomass were assumed to be 50% C (Williams *et al*. 2005) and converted to CO_2-e by multiplying by a factor of 3.67 (Commonwealth of Australia 2014).

Carbon stored in pasture over the 20 years was the accumulated change in biomass (t CO_2-e) from the first year of simulation (1993).

Savanna burning emissions were estimated by multiplying the area burnt by an emissions factor of 0.1 t CO_2-e ha^{-1} per fire (Bray and Golden 2009).

Livestock methane emissions were calculated by multiplying the total AE of animals carried for a whole year by an emission factor of 1.5 t CO_2-e $year^{-1}$, assuming livestock methane production was ~200 g methane AE^{-1} day^{-1} (Charmley *et al*. 2011) and that methane had a global warming potential of 21 (Commonwealth of Australia 2014).

Results

Biological outcomes

Pasture yield (TSDM kg ha^{-1}) and the number of stock carried (AE) were greatest for the business-as-usual scenario when 0% of the regrowth was retained for C sequestration (Fig. 2). When 100% of the regrowth was retained, TBA increased at an average annual rate of 5.8% per annum over 20 years, or 0.13 m^2 ha^{-1} $year^{-1}$ (Fig. 2*c*). The fluctuating annual TBA evident in scenarios where a proportion of regrowth was cleared shows the impact of regrowth pulling treatments and the effectiveness of burns in retarding regrowth. The initial TBA of regrowth paddocks (refer to Table 1) varied from 0.1 to 2.5 m^2 ha^{-1}, with the lowest initial TBA (0.2, 0.1 m^2 ha^{-1}) occurring in the paddocks required to achieve the 25% and 50% C farming scenarios (refer to Table 2). Total increases in TBA in the five regrowth scenarios over 20 years ranged between 0.3 m^2 ha^{-1} for 0% regrowth retained and 2.6 m^2 ha^{-1} for 100% regrowth retained (Fig. 2*c*). The average TBA of regrowth (including cleared and retained regrowth) on the property after 20 years for 0%, 25%, 50%, 75% and 100% regrowth retained C farming scenarios is 0.7, 0.8, 1.0, 1.8, 2.8 m^2 ha^{-1} respectively.

When all regrowth was retained for C farming (100% retained), pasture production (TSDM kg ha^{-1}) was least and the number of stock carried (AE) lowest (Fig. 2*a*, *b*). Compared with business-as-usual (0% retained) average pasture yield (2590 kg^{-1} ha^{-1}) declined by 40% when all regrowth (100% retained) was retained (1570 kg^{-1} ha^{-1}). The year-to-year variability in annual rainfall is reflected in the pasture production with the highest pasture yields occurring in the two wettest years (1999 with 747 mm, 2011 with 1430 mm) (Fig. 2*a*). During years of above-average rainfall (1996–1999, 2008–2011) the competitive effects of trees on pasture production were reduced – with pasture yield increasing in all regrowth scenarios.

Annual herd size declined ~200 AE over 20 years when all regrowth was retained and the amount of available forage was reduced (Fig. 2*b*). When all of the property's regrowth was controlled (0% retained) the total herd size after 20 years (~1160 AE) was 40% higher than that when all regrowth was retained (~830 AE). The average annual herd size after 20 years for 0%, 25%, 50%, 75% and 100% regrowth retained C farming scenarios is 1098, 1063, 1060, 982, 901 AE respectively. Of note is that both the 25% and 50% regrowth retained scenarios carried similar numbers of stock. As both of these paddocks were productive and virtually cleared (initial TBA 0.2, 0.1 m^2 ha^{-1}) the impact of regrowth thickening on carrying capacity was less than other regrowth paddocks. Also, because the paddock required to achieve the 50% retained regrowth scenario was stocked with heifers which do not produce a calf, changes in their numbers has less impact on herd size than does a change in breeder numbers.

Carbon outcomes

Retained woody regrowth dominated the GHG fluxes for the case study property (Table 3). The total amount of C sequestered in regrowth after 20 years ranged from 42.7 t CO_2-e ha^{-1} when all regrowth (5000 ha) was retained (100%) to 28.4 t CO_2-e ha^{-1} when 25% of regrowth (1380 ha) on the property was retained, to none in managed regrowth (0% retained). Paddock variation in area, land type, pasture condition, and initial TBA contributes to the different outcomes between scenarios. Of these, initial TBA, through its influence on carrying capacity and amount of C retained in regrowth, is likely to have the greatest impact on scenario outcomes. After 20 years, partly as a function of the year-to-year variability in rainfall and the maintenance stocking rates employed, C accumulated in the pastures under all scenarios. However, the amount of C sequestered in pasture (7.3 t CO_2-e ha^{-1}) across the property (10 150 ha) was more than double that when no regrowth was retained compared with when all regrowth was retained (3.2 t CO_2-e ha^{-1}) (Table 3). Total livestock methane emissions after 20 years increased slightly in proportion to the higher cattle numbers carried when less regrowth was retained. Savanna burning emissions were an order of magnitude smaller than livestock emissions and were greatest for the business-as-usual (0% retained) scenario (Table 3).

Sequestration in aboveground vegetation (woody regrowth and pasture) resulted in the net removal of CO_2 from the atmosphere for all C farming scenarios (Fig. 3). After 20 years, net CO_2 sink was between ~40 880 t CO_2-e for the 0% retained regrowth scenario and ~219 110 t CO_2-e for the 100% retained regrowth scenario. Methane emissions from livestock negated between ~10% (100% retained) and 30% (0% retained) of aboveground sequestration.

Economic outcomes

Without C income, the 100% regrowth retained scenario achieved the lowest average annual profit, TGM and GM AE^{-1} of all C farming scenarios (Table 4). The financial outcomes progressively improved as less regrowth was retained. The average annual TGM for 0% retained regrowth scenario was 32% higher ($47 790) than that for the 100% retained regrowth scenario. The reduced accumulated TGM for 0% retained scenario, when adjusted to include the cost of clearing, was only 13% higher ($3.34 million versus $2.96 million) than the accumulated TGM when all regrowth was retained (100%). The gross margin per AE (~$178 GM AE^{-1}) when little or no regrowth on the property was retained was almost 10% higher than that

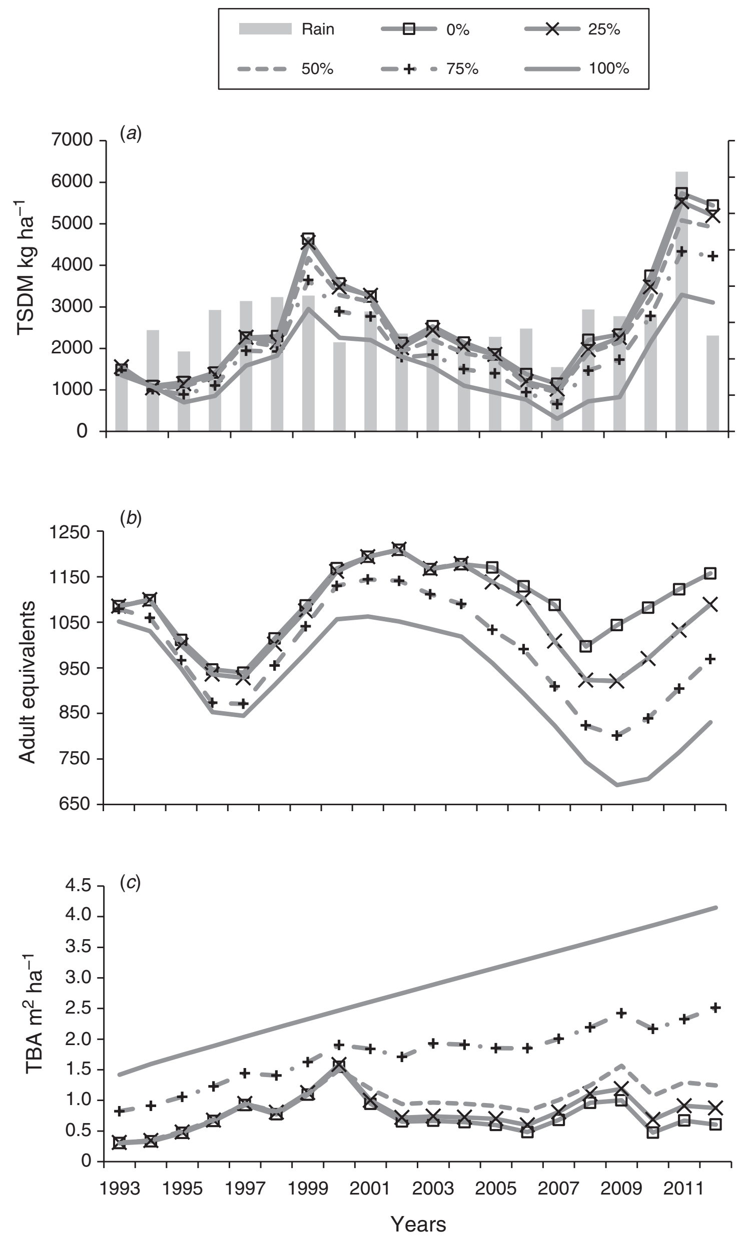

Fig. 2. Average annual (*a*) total standing dry matter (TSDM kg ha^{-1}) and rainfall (mm), (*b*) herd adult equivalents (AE), and (*c*) regrowth tree basal area (TBA m^2 ha^{-1}) achieved for the five carbon farming scenarios (0% regrowth retained, 25% regrowth retained, 50% regrowth retained, 75% regrowth retained, 100% regrowth retained).

for the scenarios with high proportions of retained regrowth (~\$162 GM AE^{-1}), exacerbating the impact of reduced livestock numbers.

When C income is considered, a net C price (price received after management expenses are removed) of \$1.75 per t CO_2-e for the C sequestered in trees (213 489 t CO_2-e) was required to make up the difference in TGM (\$374 613) between the 0% and 100% regrowth retained scenarios (Table 5). However, a C price of between \$2.95 and \$3.93 was needed to make up the difference in TGM when 50% or less regrowth was retained. The higher opportunity cost (\$/t CO_2-e) for the 25% and 50% scenarios results from the productive land type and minimal initial regrowth (TBA 0.2, 0.1 m^2 ha^{-1}) of each paddock involved in the scenarios (refer to Tables 1 and 2).

Discussion

Dedicating a proportion of the regrowth on the property to C sequestration reduced pasture (up to 40%) and herd productivity (up to 20%). In the absence of C income, this resulted in financial

Table 3. The derived greenhouse gas fluxes (t CO_2-e ha^{-1}) for the 10 150 ha property after 20 years for each of the five carbon farming scenarios (0% regrowth retained, 25% regrowth retained, 50% regrowth retained, 75% regrowth retained, 100% regrowth retained)
Woody regrowth C sequestration is based on the area of regrowth retained on the property (ha)

Carbon farming scenario % regrowth retained (ha)	Total woody regrowth sequestration	Total pasture sequestration	Livestock emissions	Savanna burning emissions
0% (0 ha)	0.0	7.3	3.2	0.35
25% (1380 ha)	28.4	6.7	3.1	0.30
50% (2380 ha)	35.1	6.2	3.1	0.25
75% (3720 ha)	40.5	5.1	2.9	0.15
100% (5000 ha)	42.7	3.2	2.7	0.00

losses (up to 24% reduction in TGM) for the beef grazing enterprise. As such, C storage through retaining tree regrowth would be unlikely unless this generated income at least equivalent to that lost due to decreased cattle productivity.

A minimum net C income (income after grazing management expenses are removed) of \$2–4 per t CO_2-e is required to offset the economic loss of retaining regrowth. These analyses do not include the transaction costs associated with the sale of sequestered C and the continued monitoring and reporting of C stocks, which can be substantial (e.g. \$9200 per annum, Walsh and Cowley 2016). Also, a higher C price would be required if cattle prices were to increase. For example, cattle prices at the end of 2015 were approximately double those used in this modelling study. Additionally, this modelled breeding operation, although

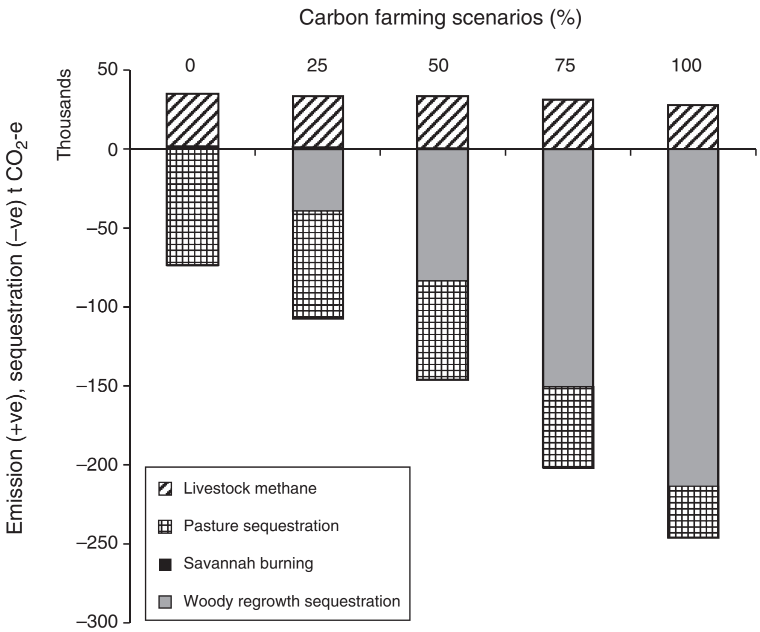

Fig. 3. Magnitude of livestock methane and savanna burning emissions and sequestration of carbon in retained woody regrowth and average annual pasture biomass (t CO_2-e) accumulated over 20 years for each of the five carbon farming scenarios (0% regrowth retained, 25% regrowth retained, 50% regrowth retained, 75% regrowth retained, 100% regrowth retained).

Table 4. Average annual financial outcomes, costs of clearing, and total gross margins minus clearing costs for the 20 years for each of the five carbon farming scenarios (0% regrowth retained, 25% regrowth retained, 50% regrowth retained, 75% regrowth retained, 100% regrowth retained)
TGM, total gross margin; GM, gross margin; AE, adult equivalent

Carbon farming scenario % regrowth retained	Average annual TGM	Average annual profit	Average annual GM AE^{-1}	Annual clearing costs	Accumulated TGM – clearing costs
0%	\$195 883	\$57 581	\$179	\$29 055	\$3 336 562
25%	\$188 187	\$47 445	\$177	\$29 055	\$3 182 642
50%	\$177 045	\$36 079	\$167	\$22 555	\$3 089 801
75%	\$161 552	\$16 622	\$163	\$9490	\$3 041 233
100%	\$148 097	\$714	\$162	\$0	\$2 961 950

Table 5. Differences between carbon farming scenario 1 (0% regrowth retained) and the four other scenarios (25% regrowth retained, 50% regrowth retained, 75% regrowth retained, 100% regrowth retained) for total gross margin (TGM) minus clearing costs, the carbon sequestered in woody regrowth (t CO_2-e), and the breakeven carbon price (t CO_2-e) required to match the loss of income

Carbon farming scenario % regrowth retained	Difference between carbon farming scenario 1 (0% regrowth retained) and other scenarios		Carbon price t CO_2-e
	TGM – clearing cost	Woody regrowth carbon sequestration t CO_2-e	
25%	–$153 920	39 136	$3.93
50%	–$246 762	83 627	$2.95
75%	–$295 329	150 580	$1.96
100%	–$374 613	213 489	$1.75

representative of the case study property where weaners were transferred to other family-owned properties, is likely to underestimate the income from cattle when steers are sold at an older age. As such, a higher C price would likely be required to compensate for the higher income derived from feeder steer operations (sold to a feedlot for finishing before slaughter). These considerations are consistent with the conclusions of Donaghy *et al*. (2009), who revealed that landholders who retained strips of regrowth and continued to graze livestock at carrying capacity would be financially better off over 25 years with a net C price of greater than $10 per t CO_2-e.

There is considerable uncertainty around the price received for abatement of emissions. During the 10 years (2003–2012), which the NSW Greenhouse Gas Reduction (or Abatement) Scheme scheme operated (IPART 2013), the price received by electricity providers for abatement certificates fluctuated between $5 and $14 per t CO_2-e (Johnson and Coburn 2010). Abatement purchased by the Australian Government in the first ERF auction was approximately $14 t CO_2-e, with over half the proposed emission reductions being sequestered in soil and trees (Commonwealth of Australia 2015*c*). However, under the ERF, there is a price discount if the sequestration project nominates 25 years rather than 100 years as the period for which C stores must be maintained (Commonwealth of Australia 2015*d*). Generally, trees sequester C at the highest rate between age 10 and 20–30, after this age the sequestration rate slows gradually until plateauing when trees are ~80–100 years of age (Johnson and Coburn 2010). The high risk of fluctuating and lower C prices in the future, the permanence obligations for sequestration, and the opportunity cost of foregone value on land all add to the uncertainty around C farming projects.

The rate at which biomass of regrowth accumulates over 20 years is fundamental to the estimation of sequestered C. Woody regrowth sequestration rates vary with age, species mix, climate, soil characteristics and management practices (Scanlan 2002; Henry *et al*. 2015*a*). Without clearing and regular burning, the average increase in TBA of regrowth (~0.23 m^2 ha^{-1} $year^{-1}$ at 30 cm) was higher than reported rates for remnant woodlands (0.076 m^2 ha^{-1} $year^{-1}$, Burrows *et al*. 2002; 0.045 m^2 ha^{-1} $year^{-1}$, Bray and Golden 2009), but lower than the rate (~0.85 m^2 ha^{-1} $year^{-1}$) reported for brigalow regrowth stands that were pulled 15 years previously (Scanlan 1991). Few studies report the growth rate of regrowth for different vegetation types, regions, and clearing treatments. The relationships between time since clearing and stand basal area used in this study were based on data in central and southern Queensland (Donaghy *et al*. 2009), and reflect the expected higher annual basal area growth increment for a regrowth site compared with that of a mature stand (Burrows *et al*. 1990).

In this study, these locally relevant regrowth relationships and property-specific information (tree-cover, land type, management practices, climate) were used to estimate woody regrowth biomass and C sequestration. The Full C Accounting Model (FullCAM), which is used for both national GHG accounting and project-scale sequestration activities, provides an alternative approach for predicting biomass production and C sequestration rates (Commonwealth of Australia 2015*e*; Paul *et al*. 2015). The rates of accumulated C (1.4–2.1 t CO_2-e ha^{-1} $year^{-1}$) in retained woody regrowth from this study were similar to the modelled rates for 20-year-old brigalow regrowth in southern central Queensland (1.4 t CO_2-e ha^{-1} $year^{-1}$, Bray and Golden 2009), as well as the predicted rates using FullCAM for planted mixed native tree species in a wool-producing area of Western Australia (2.0 t CO_2-e ha^{-1} $year^{-1}$), but lower than those rates in New South Wales (4.4 t CO_2-e ha^{-1} $year^{-1}$) (Henry *et al*. 2015*b*).

In our study, we assumed only half the initial woody regrowth biomass was eligible for sequestration. The amount of initial biomass included in sequestration projects will have an impact on the C income received from these activities. For example, the ERF methodology for 'avoided clearing of native regrowth' and 'native forest from managed regrowth' has set rules, which apply to how much of the initial regrowth stock can be claimed (Commonwealth of Australia 2015*f*). These include a 'materiality test', which determines the baseline used in calculations of accumulated C depending on the proportion of biomass present in the 10 years before commencement of the C farming activity. Landholders considering the retention of woody regrowth for C sequestration will need to consider all the aforementioned factors (vegetation type, age, climate, soil characteristics, management practices, ERF methodologies) when assessing the viability of a C offsets activity.

The potential impacts of woody regrowth on pasture and animal productivity will be a major factor determining the financial viability of C farming activities on beef enterprises. In this study, although C stocks were dominated by woody regrowth, average pasture yield and the C it contained decreased by 40% as all the regrowth on the property was retained. However, the reduction in pasture C stocks was partly offset by the 20% reduction in the number of cattle carried all year, thereby reducing livestock emissions. The accumulated C stored in pasture (3.2–7.3 t CO_2-e ha^{-1}) under a conservatively stocked property was more than double the methane emitted from livestock (2.7–3.2 t CO_2-e ha^{-1}) over 20 years. The net C position of retained regrowth, albeit with fewer components contributing to the C balance, was within range of the reported values from a grazing property in northern Australia (Bray *et al*. 2014).

Pasture C stocks are not included in the current ERF methodologies, however, the substantial C stocks in forage and litter biomass in Queensland grazing lands (estimated 633 Mt CO_2-e, Bray and Willcocks 2009) can provide additional cattle productivity and environmental benefits. Accumulation of pasture C stocks, and subsequent maintenance of ground cover, will reduce erosion and sediment entering waterways and improve water quality in the Great Barrier Reef lagoon (Fraser and Stone 2016). An unexpected but important outcome of this study is that a conservative stocking strategy that maintains the condition of land is able to achieve positive GHG outcomes along with sustainable economic and productivity outcomes. Although not estimated in this study, reductions in stocking rate with subsequent improvements in land condition and individual animal performance could improve the emission intensity of a grazing beef operation (Burrows *et al*. 2010). This has been demonstrated by a major pastoral operation which, through the adoption of innovative practices and technologies, enhanced the efficiency with which beef was produced while also moving towards achieving their financial and environmental goals (Bentley *et al*. 2008).

The case study modelling approach used in this paper included the complexity of landscape variability and specific property data. Land types and their areas present, the condition of pastures, and the initial TBA all had some influence on the productivity and C scenario outcomes, but little impact on the conclusions of this study. The engagement of producers and the use of their property data improved the likelihood that modelled outcomes were realistic and applicable to the extensive beef industry in northern Australia where woody regrowth occurs.

Conclusions

Retaining areas of woody regrowth for storage of C reduced the productivity and profitability arising from beef production on a case study grazing enterprise in northern Australia. Nonetheless, a conservatively stocked breeding operation can achieve sustained cattle productivity, environmental and economic outcomes, including net positive C stock. The Australian Government's ERF provides an opportunity for landholders to generate and sell C offsets generating income through C sequestered in woody regrowth, which has potential to more than offset the reduction in livestock income. However, uncertainty about the rules which apply to achieve successful ERF auction bids, transaction costs and C prices act as a barrier to the implementation of this and similar schemes. Although this study was based on data and information from a beef enterprise in central Queensland's grazing lands, the approach and learnings are expected to be applicable across northern Australia where regrowth occurs.

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