Carbon farming via assisted natural regeneration as a cost-effective mechanism for restoring biodiversity in agricultural landscapes

Megan C. Evans a,*, Josie Carwardine b, c, Rod J. Fensham c, d, Don W. Butler d, Kerrie A. Wilson e, Hugh P. Possingham c, e, Tara G. Martin b, c

a The Australian National University, Fenner School of Environment and Society, Canberra 0200, ACT, Australia
b CSIRO Land and Water, Ecosciences Precinct, Dutton Park 4102, QLD, Australia
c The University of Queensland, Centre for Biodiversity and Conservation Science, School of Biological Sciences, Brisbane 4072, QLD, Australia
d Queensland Herbarium, Department of Science, Information Technology, Innovation and the Arts, Mt. Coot-tha Road, Brisbane 4068, QLD, Australia
e Imperial College London, Department of Life Sciences, Silwood Park, Ascot SL5 7PY, Berkshire, England, UK

Abstract

Carbon farming in agricultural landscapes may provide a cost-effective mechanism for offsetting carbon emissions while delivering co-benefits for biodiversity through ecosystem restoration. Reforestation of landscapes using native tree and shrub species, termed environmental plantings, has been recognized as a carbon offset methodology which can contribute to biodiversity conservation as well as climate mitigation. However, far less attention has been paid to the potential for assisted natural regeneration in areas of low to intermediate levels of degradation, where regenerative capacity still remains and little intervention would be required to restore native vegetation. In this study, we considered the economics of carbon farming in the state of Queensland, Australia, where 30.6 million hectares of relatively recently deforested agricultural landscapes may be suitable for carbon farming. Using spatially explicit estimates of the rate of carbon sequestration and the opportunity cost of agricultural production, we used a discounted cash flow analysis to examine the economic viability of assisted natural regeneration relative to environmental plantings. We found that the average minimum carbon price required to make assisted natural regeneration viable was 60% lower than what was required to make environmental plantings viable ($65.8 t CO₂e⁻¹ compared to $108.8 t CO₂e⁻¹). Assisted natural regeneration could sequester 1.6 to 2.2 times the amount of carbon compared to environmental plantings alone over a range of hypothetical carbon prices and assuming a moderate 5% discount rate. Using a combination of methodologies, carbon farming was a viable land use in over 2.3% of our study extent with a low $5 t CO₂e⁻¹ carbon price, and up to 10.5 million hectares (34%) with a carbon price of $50 t CO₂e⁻¹. Carbon sequestration supply and economic returns generated by assisted natural regeneration were relatively robust to variation in establishment costs and discount rates due to the utilization of low-cost
1. Introduction

The carbon market has the potential to deliver significant outcomes for ecosystem restoration alongside the abatement of greenhouse gas emissions (Bradshaw et al., 2013). The demand for terrestrial carbon sinks is creating opportunities for avoided deforestation in tropical forests (Phelps et al., 2012; Venter and Koh, 2011), as well as landscape-scale restoration through afforestation and reforestation (Galatowitsch, 2009; Peters-Stanley et al., 2013; Silver et al., 2009). There is particular interest as to whether the carbon market can deliver positive outcomes not only for the climate and local economies, but also for biodiversity (Bekessy and Wintle, 2008; Smith and Scherr, 2003). A too narrow focus on maximizing sequestration of carbon (such as the planting of monocultures) can lead to a range of negative ecological impacts (Lindenmayer et al., 2012; Pittock et al., 2013), and will miss opportunities for co-benefits derived through restoration of natural ecosystems (Bullock et al., 2011; Gilroy et al., 2014; Dwyer et al., 2009; Rey Benayas et al., 2009).

Carbon farming is a term that is used to describe land-based practices which either avoid or reduce the release of greenhouse gas emissions, or actively sequester carbon in vegetation and soils, primarily in agricultural landscapes. Several studies have examined the economics of carbon farming through establishment of monocultures or environmental plantings (Bryan et al., 2014; Bryan and Crossman, 2013; Crossman et al., 2011; Paterson and Bryan, 2012; Paul et al., 2012; Poilgise et al., 2013). Environmental plantings are a mixture of locally indigenous tree and shrub species which are planted or seeded on cleared land, and are not normally harvested (Paul et al., 2013). The potential for environmental plantings to deliver biodiversity co-benefits alongside carbon abatement has been a focus of recent work (Bryan et al., 2014; Carwardine et al., 2015; Goldstein et al., 2006; Lin et al., 2013; Nelson et al., 2008; Pichancourt et al., 2014; Renwick et al., 2014). Yet given the high up-front costs of direct planting (Chazdon, 2008; Schirmer and Field, 2000), it is surprising that there has been limited assessment of the economic viability of carbon sequestration through assisted natural regeneration of vegetation, despite the large potential biodiversity and economic benefits of this approach (Birch et al., 2010; Bradshaw et al., 2013; Butler, 2009; Dwyer et al., 2009; Funk et al., 2014; Smith and Scherr, 2003; Trotter et al., 2005).

Assisted natural regeneration (ANR, also known as managed regrowth) is recognized as a cost-effective forest restoration method that can restore biodiversity and ecosystem services in areas of intermediate levels of degradation, while also providing income for rural livelihoods (Chazdon, 2008; Ma et al., 2014). ANR relies on residual seeds and plants at the site, or dispersed from vegetation nearby. ANR utilizes low-cost techniques to assist in the natural re-establishment of vegetation, such as: restriction of livestock grazing through fencing and direct stocking rate management; cessation of tree control practices like burning and disturbance with machinery; the use of vegetation thinning to reduce competition and promote growth, and; in some circumstances, supplementary planting of seedlings (Smith and Scherr, 2003). Although most frequently applied in tropical forests (Rey Benayas, 2007; Shono et al., 2007), ANR is gaining momentum as an important mechanism for restoring forests across a range of ecosystems (Chazdon, 2008; Gilroy et al., 2014; Shono et al., 2007).

Vegetation that is allowed to naturally regenerate has several advantages for biodiversity conservation over plantings, even when plantings are comprised of native species. First, under ANR, the vegetation is more likely to be comprised of native species adapted to local conditions, resulting in vegetation that is more resilient to local climate variation and disturbance. Second, natural regeneration can result in high species diversity including trees, shrubs, forbs and grasses, whereas under environmental planting, generally only tree species are planted. Third, ANR often provides superior habitat for local fauna as a result of the increased plant and structural diversity (Bloomfield and Pearson, 2000; Bowen et al., 2009; Bruton et al., 2013; Fensham and Guymer, 2009). Finally, under the right conditions, the cost of establishing vegetation through ANR is much lower than active planting (Sampaio et al., 2007; Schirmer and Field, 2000; Smith, 2002).

Despite the potential advantages of ANR, a lack of awareness of its benefits and demonstrative results means it remains underutilized (Shono et al., 2007). ANR falls under the definition of afforestation/reforestation (A/R) under the Kyoto Protocol and Clean Development Mechanism (Smith and Scherr, 2003; Smith, 2002), but has attracted little attention as a carbon sequestration methodology compared to mechanisms such as active planting or avoided deforestation (Niles et al., 2002). ANR has most potential in locations that have not been intensively used (cropped or irrigated) or with a relatively short history of intensive land use. Across much of sub-tropical Australia most grassy eucalypt woodlands used for grazing land fall into this category (McIntyre and Martin, 2002). A window of opportunity therefore exists to achieve significant carbon and biodiversity outcomes through assisted natural regeneration across much of northern Australia (Fensham and Guymer, 2009; Martin et al., 2012),
and in the Texas drylands (Asner et al., 2003), central Brazilian pastoral lands (Sampaio et al., 2007), the Gran Chaco in Argentina (Zak et al., 2004), degraded pastoral landscapes in Albania (Deichmann and Zhang, 2013) and in the mountainous Humbo region of Ethiopia (Biryahwaho et al., 2012).

The aim of this study was to evaluate the potential for carbon farming in the extensive agricultural landscapes of the state of Queensland, in north-eastern Australia, by examining the economic viability of ANR relative to environmental plantings. Commercial livestock grazing on pastures with dominant native species is the main land use across Queensland. The extensive, as opposed to intensive (McIntyre and Martin, 2002), nature of grazing in much of Queensland provides ideal conditions for carbon sequestration via ANR. Profitability (profitability at full equity) of grazing throughout Queensland is generally low with many farms losing money in recent years (ABARES, 2013). To determine whether carbon farming could be a viable land use in Queensland, we conducted a spatially explicit analysis of the minimum (‘break-even’) carbon price required for carbon farming to become profitable via environmental plantings and ANR. We also considered a range of hypothetical carbon prices and discount rates to estimate the carbon sequestration supply and profitability of carbon farming over a long (100 years) and medium (25 years) project duration. Finally, we tested the sensitivity of our results to variation in the establishment costs of each methodology.

Fig. 1 – The study extent encompasses 73 sub-bioregions in Queensland, of which agricultural landscapes make up 30.6 million hectares. Areas of remnant vegetation (in black) are excluded from the analysis. The Brigalow Belt bioregion (hatched) covers an extensive part of the study extent.
2. Study region and policy context

Our case study region is in the state of Queensland, in northeastern Australia (Fig. 1). Agricultural development over the past 150 years has led to extensive landscape modification (Dwyer et al., 2009; McAlpine et al., 2002) with the most rapid development occurring in the vast Brigalow Belt bioregion within the latter half the 20th century (Seabrook et al., 2006). As a result, around 34 million hectares of vegetation in Queensland (20% of the state’s total vegetated area) is now considered non-remnant: heavily modified, secondary vegetation. Commercial grazing of livestock is the predominant land use across much of northern Australia, where it occurs in extensively managed grassy eucalypt and acacia woodlands and shrublands (Martin and McIntyre, 2007). Unlike southern parts of the continent, these northern landscapes have not been subject to broad scale intensification via sowing of exotic pastures, fertilization and irrigation. Despite broad scale clearing of trees and shrubs in some regions (Martin et al., 2012), much of the cleared land retains regenerative capacity (small trees and soil seed bank) e.g. Brigalow (Acacia harpophylla) (Butler, 2009; Dwyer et al., 2009; Fensham and Guymet, 2009).

At present, clearing regrowth to maintain high quality forage for livestock represents a substantial management cost to graziers throughout Queensland (Gowen et al., 2012; McIntyre and Martin, 2002). Landholders not clearing regrowth would forgo some pasture, but could attract credits for carbon sequestered if the vegetation was left to regenerate (Commonwealth of Australia, 2013a,b). Extensive restoration of vegetation in these agricultural landscapes is a high priority to avoid potential long term ecological impacts and mitigate extinction debt from past clearing (Martin, 2010; McAlpine et al., 2002).

Australia’s Carbon Farming Initiative (CFI, Commonwealth of Australia, 2011) and climate policies are currently under review; however, there is broad political support for landholders to generate additional income through the provision of land-based carbon offsets in agricultural landscapes. We examine a range of carbon prices, project durations and establishment costs in order to gain an understanding of the economic viability of two key reforestation methodologies in our study region to help guide the carbon farming policy debate.

3. Methods

3.1. Land use data

We restricted the extent of our analysis to sub-bioregions in Queensland where at least 5% of the sub-bioregion is comprised of agricultural production landscapes (resulting in 73 of 130 sub-bioregions being considered). Our study extent encompasses 30.6 million hectares of agricultural landscapes potentially suitable for ANR or environmental plantings. To refine this extent to areas where ANR or environmental plantings are feasible, we used a state-wide vegetation coverage layer (Department of Environment and Resource Management, 2009) to delineate the extent of cleared land in Queensland (Neldner et al., 2005).

We excluded areas of intensive land use (mines, urban areas), irrigated cropping, protected areas and water bodies from the analysis using a national land use dataset (ABARE-BRS, 2010). Land uses included in our analysis were native pasture (88.7% study extent), non-irrigated cropping (hereafter, ‘cropping’, 3.5%), and modified pastures (6.1%). The native pasture category includes land where there has been limited or no deliberate attempt at pasture modification, and vegetation contains greater than 50% dominant native species (ABARES, 2010). For ease of interpretation, we incorporated modified pastures within the ‘cropping’ land use category. Approximately 1.7% of the study extent is formerly rainforest where cropping is now the dominant land use, which is important to delineate given the higher costs of environmental plantings in these areas (Catterall and Harrison, 2006).

Environmental plantings were considered to be feasible across each of our three land use categories (native pasture, cropping and former rainforest). However, ANR is generally not a suitable carbon farming method on sites which have been cultivated, irrigated and sown to exotic pastures, due to lack of local regenerative capacity in native vegetation (Fischer et al., 2009; McIntyre and Martin, 2002). We therefore restricted our analysis of ANR to areas of native pasture.

3.2. Estimating the rate of carbon sequestration

The rate of carbon accumulation through forest growth varies temporally, and so understanding the cost-effectiveness of alternative carbon farming methodologies requires this dynamic variation in flows to be explicitly accounted for (Richards and Stokes, 2004). To capture this temporal variation, we emulated the core of the FullCAM forest growth model (Richards and Brack, 2004). We used a dataset known as the Maximum Potential Biomass layer (MaxBio, Department of Climate Change and Energy Efficiency, 2004) to derive estimates of the rate of carbon sequestration under the ANR and environmental plantings methodologies.

MaxBio and FullCAM are components of the National Carbon Accounting System (NCAS), which estimates Australia’s greenhouse gas emissions from land based activities in accordance with the international guidelines adopted by the United Nations Framework Convention on Climate Change (UNFCCC). MaxBio is estimated from a forest productivity index (FPI, Kesteven and Landsberg, 2004) generated with a plant physiology model using bioclimatic parameters, and empirically related to above ground biomass in native forests (Richards and Brack, 2004).

The predicted above-ground tree biomass (t ha−1) at time t is a function of MaxBio, M, and an estimated constant k that determines the rate of approach towards the maximum biomass (Richards and Brack, 2004) is:

\[ M(t) = Me^{-kt^2}. \] (1)

Biomass generally accumulates more rapidly in tree species commonly used in environmental plantings compared to regrowth of native vegetation, hence we consider k = 20 for environmental plantings and k = 24 for ANR in this study, which reflects values used for Australia’s National greenhouse gas accounts (Commonwealth of Australia, 2012b; Commonwealth of Australia, 2014). We accounted for biomass
allocation to coarse roots (root:shoot ratio) using the recommended fraction of 0.25 for acacia forest and woodland (Commonwealth of Australia, 2012b; Snowdon et al., 2000), hence the ratio of total biomass to above ground biomass is 5:4, or 1.25 times.

The long-term average annual increment in biomass accumulation (cumulative above- and below-ground biomass) between \( t \) and \( t + 1 \) years \((\dot{t})\), using Eq. (1):

\[
\dot{t} = 1.25M \left( e^{-k/5} - e^{-k/(t-1)} \right) .
\]

(2)

The carbon content of tree biomass can range between 45 and 50% carbon, but this varies by species (Thomas and Martin, 2012) and tree component (Gifford, 2000). We adopted a conversion factor of 50% to be consistent with the Australian National Carbon Accounting System (Commonwealth of Australia, 2012b; Gifford, 2000).

The annual sequestration rate of carbon by vegetation \( (c_r, t\) CO\(_2\) ha\(^{-1}\)), is therefore:

\[
c_r = 0.5 \times 3.67 (\dot{t} - \dot{t}_{t-1}),
\]

(3)

where 3.67 is the ratio of the atomic masses of CO\(_2\) and C.

For simplicity, we assumed that the carbon stock at project commencement \( (t = 0) \) is zero. Current methodologies to credit carbon sequestration through ANR in Australia require that project areas have evidence of regeneration but little standing carbon stock at commencement. Our model is consistent with estimates from the CFI Reforestation Modelling Tool (RMT, Domestic Offsets Integrity Committee, 2011). FullCAM and the RMT model forest growth for user supplied point locations but our approach allowed us to generate estimates of carbon yield from ANR and environmental plantings over a large spatial extent rather than on a single project basis (see Supplementary material for further details).

3.3. Costs of carbon farming

Opportunity costs of agriculture were derived from the most current map of agricultural profit for Australia, which was based on data for the year 2005/2006 (Marinoni et al., 2012). The map is a grid of profitability at full equity (PFE, $ ha\(^{-1}\)) at a 1 km\(^2\) resolution across the Australian continent, using data on production, revenues and costs for 23 irrigated and rain-fed agricultural commodities, combined with data on land use (2005/2006) and yield estimates. PFE is a measure of profit which is calculated as the difference between revenue from the sale of agricultural commodities and all fixed and variable costs (Bryan et al., 2009a; Marinoni et al., 2012). The system developed by Marinoni and colleagues will enable the production of a more current map of agricultural profitability once the latest land use data set for Australia (2010/2011) is finalized. As this dataset is not yet available, we adjusted PFE to present day values based on a 2.7% annual rate of inflation between 2006 and 2013 (Reserve Bank of Australia, 2014).

We considered a mid-range once-off (incurred at \( t = 0 \)) on-ground establishment cost of $2000 ha\(^{-1}\) for environmental plantings (tree stock, fencing, weed management, labour), but also conducted sensitivity analyses using high and low cost estimates ($3000 ha\(^{-1}\) and $1000 ha\(^{-1}\)) adopted in previous studies (Crossman et al., 2011; Polglase et al., 2013; Schirmer and Field, 2000). Environmental plantings in areas of former rainforest incurred an establishment cost of $8000 ha\(^{-1}\) (Catterall and Harrison, 2006).

Ceasing the routine re-clearing of regrowth vegetation is likely to be sufficient to allow regeneration in many parts of our Queensland study region. The balance between re-clearing costs and production income are part of PFE, hence we considered an establishment cost of $0 ha\(^{-1}\) for ANR in our analysis. However, in areas with a longer history of intensive land use, it may be necessary to restrict livestock access to the carbon farming project site to facilitate regeneration of vegetation (Comerford et al., 2011; Prober et al., 2011; Veski and Westoby, 2001; Wett et al., 2011). We therefore conducted a sensitivity analysis by considering an establishment cost for ANR to cover the cost of erecting fences (including materials, labour, and transport). We derived an estimate for the establishment cost for an ANR project using estimates from Schirmer and Field (2000). This estimate was adjusted to present day values based on a 2.9% annual rate of inflation between 2000 and 2013 (Reserve Bank of Australia, 2014), to reach a final estimate of $460 ha\(^{-1}\) (Fig. S1).

Finally, for both environmental plantings and ANR, we derived annual on-ground management costs from comparable studies (Comerford et al., 2011; Polglase et al., 2008; Schirmer and Field, 2000) and adjusted to 2013 prices (Reserve Bank of Australia, 2014) to reach an estimate of $45 ha\(^{-1}\) year\(^{-1}\). Market participation costs included an initial project establishment cost ($100 ha\(^{-1}\)), as well as annual monitoring and auditing costs of $10 ha\(^{-1}\) year\(^{-1}\), and transaction costs of $10 ha\(^{-1}\) year\(^{-1}\) (Bryan et al., 2014; Comerford et al., 2011; Paul et al., 2013).

3.4. Economic viability of carbon farming

To determine the economic viability of carbon farming, we used a discounted cash flow analysis to calculate the minimum price on carbon required to generate an economic return via environmental plantings or ANR.

We generated a 1 km\(^2\) vector grid covering the extent of the study area, resulting in 707,530 planning units i. The average annual sequestration rate of carbon by ANR and environmental plantings (Section 3.2) and opportunity cost of carbon farming (Section 3.3) was calculated for each planning unit.

The net present value (NPV) of carbon sequestration in each site i is:

\[
NPV_i = PV_{B_i} - PVC_i
\]

(4)

where \( PV_{B_i} \) is the present value of the benefits at site i, calculated according to a carbon price \( p \) ($ t CO\(_2\)e\(^{-1}\) year\(^{-1}\) discount rate \( r \), and accounting for \( c_0 \) the rate of carbon sequestration at time t in site i:

\[
PV_{B_i} = \sum_{t=0}^{\infty} \frac{PC_i (1 - d_s - d_r - 25)}{(1 + r)^t}.
\]

(5)

For Australian Government cost-benefit analyses, it has been proposed that discount rates over a range of 3 to 10% should be tested (Harrison, 2010). Previous studies have evaluated the economic potential of carbon farming using discount rates ranging from 0% to 12% (Bryan et al., 2014; Funk et al., 2014; Paul et al., 2013; Polglase et al., 2013; Renwick et al.,
Unless otherwise indicated, we present our results where a moderate 5% discount rate has been applied throughout the manuscript. We also test the sensitivity of our findings to rates of 1.5% and 10% to enable comparison to the results of relevant key studies.

We accounted for a risk of reversal buffer (\(d_t\)) of 5%, which is deducted as a percentage of generated carbon credits in order to insure the CFI scheme against residual risks (Commonwealth of Australia, 2012a). Carbon farming policy in Australia currently requires carbon sequestration projects to remain in place for 100 years to meet permanence obligations (Commonwealth of Australia, 2012a; Macintosh and Waugh, 2012). An option for landholders to adopt a 25-year contract for a carbon farming project is currently under consideration (Australian Government, 2014), but under this permanence option 20% of carbon credits would be deducted to reflect the potential cost to Government of replacing carbon stores if 25-year projects are discontinued. Hence we considered two project durations \(T\) of 100 and 25 years, and applied a 20% discount (\(d_T=0.25\)) to the credits earned if \(T=25\).

The present value PVC\(_{ij}\) of the costs of carbon sequestration at site \(i\) is:

\[
PVC_{ij} = EC + \sum_{t=0}^{T} \frac{MC + TC + PFE_i}{(1+r)^t},
\]

where \(EC\) is the sum of the initial establishment and costs (\(\$\ ha^{-1}\)), \(MC\) is the annual on-ground management cost (\(\$\ ha^{-1} \text{ year}^{-1}\)), \(TC\) is the sum of transaction and monitoring costs (\(\$\ ha^{-1} \text{ year}^{-1}\)) and \(PFE\) is the profitability at full equity (\(\$\ ha^{-1}\)) of the current agricultural land use in site \(i\) (Marinoni et al., 2012).

Finally, we converted the NPV in each site to the equal annual equivalent:

\[
\text{EAE}_i = \text{NPV} \left( \frac{r(1+r)^T}{(1+r)^T - 1} \right).
\]

Spatial analyses were conducted using ArcMap version 10 (ESRI, 2011), the discounted cash flow analysis was implemented using MATLAB version 7.10.0.499 (2010), and results were analysed using the R statistical package version 2.15.0 (R Development Core Team, 2012).

4. Results

4.1. Break-even carbon prices

We calculated the minimum price required for carbon farming to become profitable to gain an understanding of the size of payment necessary (\(\$\ t CO_2\text{-e}^{-1}\)) to encourage landholder adoption of ANR or environmental plantings in our study region. The ‘break-even’ carbon price \(p\) occurs when the NPV (Eq. (4)) is equal to 0. A positive break-even price indicates that a payment is required to stimulate conversion from agriculture to carbon farming, whereas a negative break-even price signifies that the current agricultural land use is producing...
negative economic returns and a conversion to carbon farming could occur at no cost.

Overall, the average site-level break-even carbon price ($T = 100$) was considerably lower for ANR ($\$65.8 \text{ t CO}_2\text{e}^{-1}$) as compared to environmental plantings ($\$108.8 \text{ t CO}_2\text{e}^{-1}$, Fig. 2). The break-even price varied according to land use (Fig. S2), whereby environmental plantings were generally more economically viable on sites where cropping was the dominant land use ($\$99.9 \text{ t CO}_2\text{e}^{-1}$) compared to sites on native pasture ($\$109.5 \text{ t CO}_2\text{e}^{-1}$). Environmental plantings on former rainforest sites broke even for $\$153.0 \text{ t CO}_2\text{e}^{-1}$ on average. These averaged estimates mask much of the spatial heterogeneity in the break-even carbon price across the study extent (Fig. 2). Low break-even prices were more frequent in the relatively productive east of the study region, and several areas in the central eastern coast have similar break-even prices under either methodology. Over the 25 year project duration, average break-even estimates for ANR increased to $\$76.1 \text{ t CO}_2\text{e}^{-1}$, and to $\$141.5 \text{ t CO}_2\text{e}^{-1}$ for environmental plantings.

4.2. Carbon sequestration

Using the break-even prices estimated previously for all sites in our study extent, we generated supply curves for carbon sequestration using ANR and environmental plantings under the two project durations (Fig. 3). We also present a third ‘least cost’ curve which is derived by selecting the methodology with the lowest break-even price in each site. This portfolio comprises of sites where environmental planting is the only available carbon farming methodology (where the land use is either cropping or former rainforest), in addition to sites where ANR is the more cost-effective of the two possible carbon methodologies.

With a low carbon price of $\$5 \text{ t CO}_2\text{e}^{-1}$ (similar to the currently trading price in the European market), 63 Mt CO₂ could be sequestered over 100 years by considering environmental plantings alone (Fig. 3a). ANR could supply 110 Mt CO₂ at this price, and a total of 123 Mt CO₂-e could be sequestered if the ‘least cost’ methodology was adopted in each site (Table 1).

At a moderate carbon price of $\$20 \text{ t CO}_2\text{e}^{-1}$, it is feasible for around 243 Mt CO₂ to be sequestered by a mixture of ANR and environmental plantings over 100 years. Carbon farming becomes more viable under a high carbon price of $\$50 \text{ t CO}_2\text{e}^{-1}$ (comparable to the estimated price required to induce significant cuts in emissions, Pezzey and Jotzo, 2013), with around 1825 Mt CO₂ that could be supplied using a combination of ANR and environmental plantings. ANR however could viably sequester 1664 Mt CO₂-e at this price, whereas carbon farming via environmental plantings alone could supply 770 Mt CO₂-e over 100 years.

Carbon sequestration supply over the 25 year project duration was less than half of what could be achieved over 100 years for each of our hypothetical carbon prices. A total of 710 Mt CO₂-e could be sequestered with a $\$50 \text{ t CO}_2\text{e}^{-1}$ carbon price using a combination of ANR and environmental plantings (Fig. 3b). Supply of carbon sequestration via ANR

Fig. 3 – Carbon sequestration supply curves for ANR, environmental plantings and ‘least cost’ methodology (where the methodology with the lowest break-even price at each site is assumed to be adopted) for (a) 100 year and (b) 25 year project durations. The y-axis is restricted to $\$200 \text{ per t CO}_2\text{e}$ or less for ease of interpretation.
was relatively insensitive to discounting due to negligible establishment costs, whereas a high discount rate (10%) increased the disparity evident in the economic viability of environmental plantings relative to ANR (Fig. S3).

4.3. Economic returns under hypothetical carbon price scenarios

Carbon farming was competitive with agriculture over a fairly limited spatial extent under low and moderate carbon prices (Fig. 4). Environmental plantings was viable over 372,500 ha (1.2% study extent), while ANR viable was over approximately twice that area (626,400 ha) with $5 t \text{CO}_2\text{e}^{-1}$. Increasing to a moderate $20 t \text{CO}_2\text{e}^{-1}$, the area viable for carbon farming increased marginally to 568,700 ha and 1088,200 ha for environmental plantings and ANR, respectively.

With a $50 t \text{CO}_2\text{e}^{-1}$ carbon price, carbon farming via ANR alone was competitive with agriculture over 9.8 million hectares, or 32.3% of the study extent, and could generate $503 M per year over 100 years. Environmental plantings alone was viable across 3.1 million ha (10.2% study extent) and generated less than half of the economic returns possible under ANR ($212 M per year). When we considered the ‘least cost’ methodology in each site, ANR generated the vast majority of economic returns, holding the market share of between 83 and 96% of total net present value over 100 years, under each of our carbon price scenarios and discount rates (Table 1). Assuming the ‘least cost’ methodology was adopted in each site, the total area viable for carbon farming and carbon sequestered at this price was actually comparable under both 1.5% and 10% discount rates with a carbon price of $20 t \text{CO}_2\text{e}^{-1}$ or more. Environmental plantings were more attractive with a lower discount rate, and subsequently made up a greater proportion of the market (Table 1). However, a high discount rate resulted in the ‘least cost’ supply curve shifting upwards, reducing the overall viability of carbon farming (Fig. S3).

Annual economic returns over the 25 year project duration ranged between 69% and 94% of what could be achieved over 100 years for carbon prices of $50 and $5 t \text{CO}_2\text{e}^{-1}$, respectively. The proportion of economic returns via ANR was slightly higher over the shorter project duration (Table 1). However, the total area viable for carbon farming was largely unaffected by project duration (Fig. S4).

4.4. Impact of variation in establishment costs

When we considered a high establishment cost for ANR, the average break-even price for this methodology increased from $65.8 to $80.0 t \text{CO}_2\text{e}^{-1}$. However, this was still less than the average break-even price for environmental plantings with both low ($83.1 t \text{CO}_2\text{e}^{-1}$) and high establishment costs ($134.4 t \text{CO}_2\text{e}^{-1}$).

If a high establishment of $460 ha^{-1}$ was incurred for ANR across all eligible sites, the supply of carbon via this methodology would decrease from 1664 to 1251 Mt \text{CO}_2\text{e} (25%) over 100 years, assuming a $50 t \text{CO}_2\text{e}^{-1}$ carbon price (Fig. 5a). The environmental plantings supply curve was highly sensitive to variation in establishment costs (Fig. 5b), with an increase from $2000 ha^{-1}$ to $3000 ha^{-1}$ leading to a reduction in carbon supply from 770 to 342 Mt \text{CO}_2\text{e} (56%) over 100 years.

However, when the ‘least cost’ methodology was considered in each site (Fig. 5c), variation in the establishment cost for environmental plantings had a minimal impact on the

---

Table 1 - Key results for the ‘least cost methodology’ scenario, assuming the most plausible establishment costs ($2000 ha^{-1}$ for environmental plantings, $0 ha^{-1}$ for ANR), 100 year project duration and three hypothetical carbon prices. Note that it is assumed the methodology (ANR or environmental plantings) with the lowest break-even price is adopted in each site. * Study extent is 30.6 million ha.

<table>
<thead>
<tr>
<th>Carbon price</th>
<th>Discount rate</th>
<th>1.5%</th>
<th>5%</th>
<th>10%</th>
</tr>
</thead>
<tbody>
<tr>
<td>$5 per t \text{CO}_2\text{e}$</td>
<td>Area of carbon farming (ha)</td>
<td>762,700</td>
<td>696,700</td>
<td>622,700</td>
</tr>
<tr>
<td>Area (% study extent)</td>
<td>2.5%</td>
<td>2.3%</td>
<td>2.0%</td>
<td></td>
</tr>
<tr>
<td>Total carbon sequestered (Mt \text{CO}_2\text{e})</td>
<td>136</td>
<td>123</td>
<td>110</td>
<td></td>
</tr>
<tr>
<td>Net present value ($M)</td>
<td>4230</td>
<td>1513</td>
<td>691</td>
<td></td>
</tr>
<tr>
<td>Equal annual equivalent ($M/year)</td>
<td>82</td>
<td>76</td>
<td>69</td>
<td></td>
</tr>
<tr>
<td>% NPV from ANR</td>
<td>84%</td>
<td>89%</td>
<td>94%</td>
<td></td>
</tr>
<tr>
<td>$20 per t \text{CO}_2\text{e}$</td>
<td>Area of carbon farming (ha)</td>
<td>1135,300</td>
<td>1212,200</td>
<td>984,400</td>
</tr>
<tr>
<td>Area (% study extent)</td>
<td>3.7%</td>
<td>4.0%</td>
<td>3.2%</td>
<td></td>
</tr>
<tr>
<td>Total carbon sequestered (Mt \text{CO}_2\text{e})</td>
<td>223</td>
<td>244</td>
<td>194</td>
<td></td>
</tr>
<tr>
<td>Net present value ($M)</td>
<td>5805</td>
<td>2252</td>
<td>983</td>
<td></td>
</tr>
<tr>
<td>Equal annual equivalent ($M/year)</td>
<td>112</td>
<td>113</td>
<td>98</td>
<td></td>
</tr>
<tr>
<td>% NPV from ANR</td>
<td>83%</td>
<td>89%</td>
<td>94%</td>
<td></td>
</tr>
<tr>
<td>$50 per t \text{CO}_2\text{e}$</td>
<td>Area of carbon farming (ha)</td>
<td>8038,400</td>
<td>10508,600</td>
<td>8106,100</td>
</tr>
<tr>
<td>Area (% study extent)</td>
<td>26.3%</td>
<td>34.3%</td>
<td>26.5%</td>
<td></td>
</tr>
<tr>
<td>Total carbon sequestered (Mt \text{CO}_2\text{e})</td>
<td>1532</td>
<td>1825</td>
<td>1489</td>
<td></td>
</tr>
<tr>
<td>Net present value ($M)</td>
<td>19,720</td>
<td>10,956</td>
<td>4089</td>
<td></td>
</tr>
<tr>
<td>Equal annual equivalent ($M/year)</td>
<td>382</td>
<td>552</td>
<td>409</td>
<td></td>
</tr>
<tr>
<td>% NPV from ANR</td>
<td>85%</td>
<td>91%</td>
<td>96%</td>
<td></td>
</tr>
</tbody>
</table>
overall supply of carbon, since ANR was the most viable methodology in the majority of sites in our study region. At a $50 \text{ t CO}_2\text{e}^{-1}$ carbon price, a high establishment cost for environmental plantings reduced overall carbon supply from 1824 to 1752 Mt CO$_2$e (4%). ANR retained the market share when we considered an optimistic low establishment cost for environmental plantings (83–84% for all carbon price scenarios), and also under a high ANR establishment cost (87% for all carbon price scenarios).

5. Discussion

Carbon farming in agricultural landscapes presents an important opportunity to deliver biodiversity, economic and social co-benefits alongside terrestrial carbon abatement (Lin et al., 2013). Identifying low-cost options for carbon abatement which can contribute to biodiversity conservation and other co-benefits is a high priority (Bryan et al., 2014; Gilroy et al., 2014; Nelson et al., 2008; Phelpe et al., 2012). Our study has highlighted the potential for carbon farming to establish as a viable land use in agricultural landscapes in north-eastern Australia. In particular, our research illustrates the ability of ANR to provide a cost-effective alternative to environmental plantings for sequestering carbon and providing biodiversity co-benefits in areas of low to intermediate levels of degradation.

In our Queensland study region, we found that carbon farming was a viable alternative to agricultural production across a fairly limited spatial extent under low and moderate carbon prices. Nonetheless, this is still a significant result as it highlights the marginal economic nature of the dominant grazing land use in some parts of the landscape. The level of payments delivered either by an appropriate incentive scheme or a market price on carbon would only need to be minimal to stimulate adoption of carbon farming in such areas, but would provide an alternative viable land use as well as deliver environmental and biodiversity co-benefits. Under a scenario where carbon is priced at a level needed to stimulate significant cuts in global greenhouse gas emissions ($50 \text{ t CO}_2\text{e}^{-1}$, Pezzy and Jotzo, 2013), carbon farming could become a viable land use across up to 10.5 million hectares of agricultural landscapes (34% of our study extent), and sequester 1825 Mt CO$_2$e over 100 years. These findings demonstrate the importance of gaining an understanding of future land use change across a range of possible scenarios, in order to inform the development of policies which will have implications for climate mitigation, agriculture and biodiversity conservation.

Our study is the first to quantify the economic and carbon sequestration opportunities derived from assisted natural regeneration of vegetation in Australian agricultural landscapes, and one of few internationally (Birch et al., 2010; Funk et al., 2014; Gilroy et al., 2014). We found that where it is possible, ANR was almost always a more cost-effective methodology for sequestering carbon than the direct planting of trees. Environmental plantings were competitive with ANR on areas of native pasture only when the discount rate was very low, or in the situation where the cost of establishing environmental plantings is very low ($1000\text{ ha}^{-1}$), and a high establishment cost

---

**Fig. 4** – Equivalent annual returns ($ per ha per year) for (a) ANR and (b) environmental plantings, under hypothetical carbon prices ($5, $20 and $50) and 5% discount rate.
($460 \text{ ha}^{-1}$) is assumed for ANR. It is unlikely that the establishment cost of an ANR project would be as high as $460 \text{ ha}^{-1}$, particularly in our study region where regeneration can be facilitated simply by ceasing to re-clear regrowth vegetation (Dwyer et al., 2009; Fensham and Guymer, 2009).

Previous studies which have examined the economics of carbon farming via environmental plantings have found that the viability of this methodology is highly sensitive to variation in establishment cost (Polglase et al., 2013). Under one particular scenario evaluated by Polglase and colleagues, ($20 \text{ t CO}_2\text{e}^{-1}$, 5% discount rate), the area profitable for environmental plantings across Australia declined from 32 M ha to only 1 M ha when the establishment cost was increased from $1000 \text{ ha}^{-1}$ to $3000 \text{ ha}^{-1}$. Our results were also sensitive to changes in the establishment cost, but the overall viability of carbon farming was affected minimally when accounting for the option of ANR. In our study region, we found that the area viable for carbon farming using environmental plantings alone halved under the same circumstances (868,600 ha to 387,700 ha). However, when we considered environmental plantings in combination with ANR, the reduction in total viable area was just under 10% (1.3 M ha to 1.2 M ha), as ANR was more viable methodology in the vast majority of sites and contributed the greatest proportion of economic returns. Since ANR establishment costs are largely negligible, carbon sequestration supply and economic returns generated using this methodology are likely to be more robust to variable economic conditions.

The amount of carbon which could profitably be sequestered using ANR was roughly twice the amount possible if only environmental plantings were considered. A $50 \text{ t CO}_2\text{e}^{-1}$ carbon price could incentivize the sequestration of 1664 Mt CO$_2$e using ANR, whereas environmental plantings alone could supply 770 Mt CO$_2$e over 100 years. Environmental plantings are still an important methodological option for carbon farming, particularly in areas where natural regenerative capacity has been diminished to the point where ANR is not viable. However, our findings do indicate that ANR holds considerable potential for restoring vegetation in agricultural landscapes, particularly when the costs associated with establishing environmental plantings are high or uncertain. Our study contributes to a growing body of work which demonstrate the potential for ANR as a low-cost reforestation methodology which can benefit biodiversity conservation alongside carbon sequestration (Birch et al., 2010; Funk et al., 2014; Gilroy et al., 2014), in a literature which has so far been dominated by studies focused predominantly on environmental plantings and fast-growing monocultures (Bryan et al., 2014; Bryan and Crossman, 2013; Carwardine et al., 2015; Crossman et al., 2011; Paul et al., 2013; Polglase et al., 2013; Renwick et al., 2014).

While the biodiversity value of regrowth or secondary forests may generally not be as high as in unmodified forest

Fig. 5 – The impact of alternative establishment costs on the economic viability of carbon farming. (a) A high establishment cost for ANR ($460 \text{ ha}^{-1}$) results in a shift up for the ANR supply curve. (b) Low ($1000 \text{ ha}^{-1}$) and high ($3000 \text{ ha}^{-1}$) establishment costs for environmental plantings have a large impact on the carbon supply curve for environmental plantings, but (c) makes minimal change to the shape of the ‘least cost’ methodology curve.
(Álvarez-Yépez et al., 2008; Gibson et al., 2011; Sampaio et al., 2007), ANR has a key role to play as a pragmatic forest restoration method which can cost-effectively sequester carbon and restore biodiversity in landscapes of low to intermediate levels of degradation (Gilroy et al., 2014; Shono et al., 2007). Restoration of deforested landscapes can provide crucial habitat for highly threatened species (Bowen et al., 2009; Butler, 2009; Munro et al., 2007), by supplementing important refugia (Shoo et al., 2011) and enhancing structural complexity (Munro et al., 2009; Woinarski et al., 2009). The outcomes of restoration are highly dependent on geographic and historical context (Suding, 2011), and it should be noted that ANR is most suitable for restoring areas where some level of natural succession is in progress (Chazdon, 2008; Shono et al., 2007). In our study region, it has been shown that management history can affect density of regrowth and rates of recovery in Brigalow forest (Dwyer et al., 2010a). Management of fire and grazing will play an important role in forest regeneration. In some instances, fire may be useful for thinning which has been demonstrated to enhance growth rates in some forest types in Australian rangelands (Dwyer et al., 2010b). Likewise, the management of grazing pressure will be important to allow early establishment of trees. In regions which also contain high fuel load exotic grasses, grazing may also be necessary to manage fire risk.

It is important to consider some caveats to our approach. We have assumed that monitoring costs are incurred annually, whereas carbon farming projects often have a defined crediting period (15 years for reforestation projects under the CFI) (Commonwealth of Australia, 2012a). Monitoring costs could be reduced by undertaking measurements of carbon stocks at longer intervals (Cacho et al., 2012). Carbon farming offers considerable economies of scale (Cacho et al., 2013; Charnley et al., 2010) which we have not accounted for here. While some establishment and management costs are proportional to project size (tube stock, pest management), many components of market participation and transaction costs are fixed. It is therefore likely that carbon farming projects will be more profitable over large areas, for example where several landholders could collaborate, thereby reducing management and transaction costs (Polglase et al., 2013). Such economies of scale may be particularly significant for the ANR methodology, given there is no need for intensive restoration of vegetation and where fencing is needed, the cost will scale in proportion to project area (Schirmer and Field, 2000).

We have also assumed the full opportunity cost of agricultural production is incurred to establish carbon farming on a property, but this is likely an overestimate. Grazing by livestock is permitted on sites with environmental plantings 3 years after project establishment (Commonwealth of Australia, 2012c), and on ANR sites once forest cover has been re-established (Commonwealth of Australia, 2013b) or earlier if evidence can be provided that grazing has not prevented the regrowth of native forest (Commonwealth of Australia, 2013a). Future analyses should factor in a model of diminishing returns from grazing as a function of vegetation growth rate (Scanlan, 1991), so as to better understand the costs and benefits of ANR versus environment plantings. We have also not accounted for the cost savings associated with ceasing re-clearing of regrowth vegetation in this study, which could make ANR an even more attractive option in landscapes with high natural regenerative capacity (Dwyer et al., 2009; Gowen et al., 2012). Such an analysis will need to take into consideration the variation in regrowth clearing costs, which are dependent on the local dominant vegetation.

A potential source of uncertainty in our results is the error contained within the agricultural profitability layer (Marinoni et al., 2012) which we used as a proxy for the opportunity cost of carbon farming. Two main sources of uncertainty are inherent in spatial estimates of agricultural profitability: mapping uncertainty, and estimation uncertainty (Bryan et al., 2009). Mapping uncertainty emerges due to inaccuracies in the underlying land use data layer, as well as the use of NDVI mapping as a proxy for agricultural yield. Estimation uncertainty is mainly due to the temporal and spatial variability of individual parameters of the agricultural commodity profit function, particularly costs, which cannot be fully captured over large geographical areas and for multiple commodities (Bryan et al., 2011). It should also be noted that Marinoni et al. (2012) derived estimates of agricultural profitability for the year 2005/2006, which was a time of drought in our Queensland case study region. The viability of the beef industry in particular was affected during this time (ABARE, 2009), resulting in some negative estimates of profitability mainly in the south-west of our study extent (Marinoni et al., 2012). Our central finding of the economic viability of ANR relative to environmental plantings should be robust to this source of uncertainty, given that 89% of our study extent is devoted to livestock grazing on native pastures, and any price volatility due to drought would affect this landscape fairly evenly. We therefore expect that the overall impacts of uncertainties in agricultural profitability estimates are low, and do not change the general conclusions of our study.

We considered the influence of project duration on the viability of carbon farming, as it is clear that in addition to policy risk and market uncertainty, long-term contracts can present a significant barrier to private landholder participation (Ando and Chen, 2011; Charnley et al., 2010; Mitchell et al., 2012). In our analysis, we found that the total area viable for carbon farming and the annual economic returns over a 25 year project duration did not differ substantially to what was expected over 100 years. Although this result was of course influenced by discounting, this effect was diminished by the predominance of the low-cost ANR methodology. Our findings suggest that the 25 year contract option would offer a similar degree of financial benefit compared to a long term contract, despite the proposed 20% discount on credits earned over a 25 year project duration (Australian Government, 2014). However, a key consequence of this shorter contract option is that approximately half as much carbon would be sequestered relative to a 100 year carbon farming project. Reducing barriers to landholder participation in carbon farming schemes does not necessarily mean that only short contracts should be offered, as establishing a carbon farming project requires long term investment, and the carbon sequestration and biodiversity benefits from restoring vegetation increase over time. Alternatives to strict long-term permanence obligations such as insurance policies and premiums (van Oosterzee et al., 2012) and arrangements where the contract duration is selected based on a sliding scale with the estimated risk of project reversal (Macintosh, 2012) deserve further investigation.
In our analysis, we have demonstrated the economic viability of ANR relative to environmental plantings in Australian agricultural landscapes for the first time. An important future extension to this work would be to explicitly consider the expected biodiversity benefits derived from ANR to understand how the supply of carbon sequestration and contribution to biodiversity conservation can be jointly maximized. Targeted payments to areas of high conservation value could augment economic returns from carbon farming to facilitate ‘win-win’ carbon and biodiversity outcomes (Bryan et al., 2014; Carwardine et al., 2015; Crossman et al., 2011; Nelson et al., 2008; Phelps et al., 2012). However, trade-offs exist between biodiversity and carbon sequestration potential over both space and time. While above-ground carbon storage generally increases in a monotonic fashion as stands age and mature (Law et al., 2001; Stephenson et al., 2014), it can take substantially more time for regrowth vegetation to provide habitat values similar to mature remnant vegetation (Hatanaka et al., 2011; Woinarski et al., 2009). Carbon sequestration potential, biodiversity values and opportunity costs are unevenly distributed throughout landscapes (Crossman et al., 2011; Nelson et al., 2008). An important area of future research would be to determine how biodiversity co-benefits can be delivered alongside the economic and carbon sequestration benefits generated by the carbon market, while taking into account these potential trade-offs. In particular, future work should specifically focus on how the cost efficiencies of ANR could deliver improved outcomes for biodiversity relative to what is possible with environmental plantings, as examined by previous studies (Bryan et al., 2014; Carwardine et al., 2015; Crossman et al., 2011).

Despite the requirement for carbon sinks to remain in place over a very long timeframe, consideration of future global change and associated risks to carbon farming projects are noticeably absent in current Australian carbon farming policy. We have calculated the economic viability of carbon farming using a discounted cash flow analysis, but such a deterministic methodology is unable to account for the uncertainties inherent over long time frames in the face of climate change (Dobes, 2008; Stafford Smith et al., 2011). Future carbon prices are subject to high uncertainty and fluctuations, as evidenced by the 2012 crash of the carbon price in the European market and recent climate policy changes in Australia. The costs and benefits of carbon offsets are rarely considered within a climate adaptation framework, and in particular, the projected climate impacts on offset projects which aim to mitigate the effects of climate change are often unaccounted for. Unless analyzed appropriately, mitigation responses to climate change could ultimately prove to be maladaptive in the future. A key research gap therefore exists on how to analyze the costs and benefits of carbon offset projects in the face of uncertainty.

6. Conclusions

Carbon farming in agricultural landscapes presents an important opportunity to deliver biodiversity, economic and social co-benefits alongside carbon abatement. Although carbon farming is only one of many policy options available to stimulate abatement of greenhouse gas emissions, it is an active policy space in Australia (Australian Government, 2014; Bradshaw et al., 2013; Bryan et al., 2014), New Zealand (Funk et al., 2014; Trotter et al., 2005), Canada (Anderson et al., 2014; van Kooten, 2000) and internationally (Benítez et al., 2007; Gilroy et al., 2014; Ma et al., 2014).

We have presented the first spatially explicit assessment of potential carbon supply via assisted natural regeneration relative to environmental plantings, in a region which is significant for its biodiversity values as well as its rural history (Dwyer et al., 2009; McAlpine et al., 2002; Seabrook et al., 2006). Our findings show that carbon farming is a viable alternative to agricultural production in the marginal areas within our study region even with low and moderate carbon prices, whereas a $50 t CO₂e⁻¹ carbon price could make over 10 million hectares of land attractive for carbon sequestration projects.

Crucially, the vast majority of carbon sequestration and economic potential of carbon farming in our study region is derived from assisted natural regeneration. In addition to providing a low-cost option for terrestrial carbon sequestration, there is considerable potential for ANR to make an important contribution to biodiversity conservation within modified agricultural landscapes (Bowen et al., 2009; Butler, 2009; Martin and McIntyre, 2007; McAlpine et al., 2002).

Acknowledgements

We wish to thank Tim Capon, Rebecca Gowen, Andrew Macintosh, Stuart Whitten and Rochelle Christian for helpful discussions around this work. Sue McIntyre and Karen Hussey gave important feedback on an earlier version of the manuscript. Oswald Marinoni provided an updated agricultural profitability dataset. We also thank Jeff Hanson for advice with the spatial analysis. This research was conducted with the support of funding from the Australian Government’s National Environmental Research Program and an Australian Research Council Centre of Excellence for Environmental Decisions. M.C.E was supported by an Australian Postgraduate Award and a CSIRO Climate Adaptation Flagship scholarship. K.A.W was supported by an Australian Research Council Future Fellowship. H.P.P. was supported by an ARC Federation Fellowship. J.C was supported by CSIRO’s Climate Adaptation and Sustainable Agriculture and Forestry Flagships.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.envsci.2015.02.003.

REFERENCES


Domestic Offsets Integrity Committee, 2011. Endorsement of the Methodology for Quantifying Carbon Dioxide Sequestration by Permanent Environmental Plantings of Native Species using the CPI Reforestation Modelling Tool. Domestic Offsets Integrity Committee.


Ecosystems and Vegetation Communities in Queensland, Version 3.1. The State of Queensland, Brisbane.


