CHAPTER 1: INTRODUCTION

1.1 Global warming and the potential role of landholders

Over the past three decades, the average global surface temperature has been progressively rising to levels warmer than for any preceding decade since the 1850s. The fifth assessment report from the Intergovernmental Panel on Climate Change (IPPC) concluded, with 95% certainty, that greenhouse gas (GHG) emissions from anthropogenic activities are contributing to the warming of the globe (IPCC, 2013). Since the industrial revolution, GHG concentrations in the atmosphere have risen from 280 parts per million (ppm) to in excess of 380 ppm (Srinivasan, 2008) and are predicted to continue rising at a rate of 6ppm per year (Stern, 2007). The main contributing activity is the burning of fossil fuels for energy production resulting in increased levels of carbon dioxide (CO_2) in the atmosphere, although other important anthropogenic GHGs include methane (CH_4), nitrous oxide (N_2O), perfluorocarbons (PFCs), hydrofluorocarbons (HFCs) and sulphur hexafluoride (SF_6).

GHGs are transparent to short-wave (ultraviolet) radiation but absorb long-wave (infrared) radiation. Therefore, energy from the sun in the form of ultraviolet radiation passes unimpeded through these gases in the atmosphere and is absorbed by the earth's surface. The warming of the earth's surface results in the release of infrared radiation which in turn is absorbed and reradiated by atmospheric GHGs back to the earth's surface. This process is known as the greenhouse effect. Despite being a natural phenomenon and essential to warming the earth to a habitable level, the increasing level of GHGs is of global concern.

Without the introduction of interventional policies to substantially reduce atmospheric GHG levels, there will be a continued rise in temperature (IPCC, 2007; Innocenti & Albrito, 2011; IPCC, 2013). Due to the complex interactions involved, much uncertainty

surrounds the magnitude of this warming although the majority of the scientific community agree that, if left unmitigated, significant adverse ecosystem, social and economic impacts will be experienced in the near future (IPCC, 2013). It is predicted that weather patterns will change and sea levels will rise.

It is argued that irreversible impacts from global warming may be experienced if the implementation of GHG-reducing policies or techniques are delayed by even one or two decades (Rogelj *et al.*, 2013). Therefore it is vital in the near future to implement low-cost mitigation policies while technology advancements to reduce emissions are developed. It has been reported that changes in agricultural land uses, or the modification of current management practices, can provide low-cost methods of offsetting GHG emissions (Antle *et al.*, 2002; Garnaut, 2008; Eady *et al.*, 2009).

Garnaut (2008) posited that sequestration of carbon through land-use change by rural landholders and the management of existing forests is potentially the most important contribution to global warming mitigation in Australia. One technique is through growing trees. This could be either for the purpose of sequestering carbon from the atmosphere or for the generation of biofuels as an alternative to fossil fuel sources. The planting of trees has been widely reported to have significant potential to provide low-cost abatement, especially in the early stages while new technologies to reduce emissions are developed (Cacho *et al.*, 2008b). Other contributions from landholders can be achieved through reduction of emissions from current land uses, including reduction of livestock methane emissions through management strategies or breed selection, reduction of fertiliser inputs and the limitation of fossil-fuel powered farm machinery.

The potential supply of carbon mitigation through land-use change depends on the availability and costs of different technologies and resource endowments, and is partly

determined by location. In order to determine this potential supply it is necessary to determine the opportunity costs to landholders for the conversion of their properties to carbon-mitigating land uses. Approaches to estimate these costs can be grouped into four main techniques: bottom-up engineering approach (eg., Moulton and Richards, 1990; Cacho *et al.*, 2013; Nielsen *et al.*, 2014), sectoral optimisation models (eg., Alig *et al.*, 1997; Sohngen and Mendelsohn, 2003), econometric analyses (eg., Plantinga *et al*., 1999; Stavins, 1999; Lubowski *et al.*, 2006) and survey-based approaches (eg., Shaikh *et al.*, 2007). The bottom-up approach does not explicitly measure all factors influencing landholder decisions or account for endogenous price feedbacks resulting from a policy (Stavins, 1999). Econometric analyses, on the other hand, can capture these effects (Plantinga *et al.*, 1999; Dempsey *et al.*, 2010). Despite these shortcomings, the advantage of bottom-up approaches is that they are transparent and easy to interpret (Richards and Stokes, 2004; Dempsey *et al*., 2010). As such, this thesis adopts a bottom-up approach to estimating opportunity costs.

Once estimated, landholders' opportunity costs for converting their properties to carbon-mitigation land uses can be aggregated to provide a marginal abatement cost curve for the potential supply of carbon mitigation (Antle & Valdivia, 2006). The economic returns, and hence the opportunity costs, vary significantly between different agricultural land-use types and also spatially within the same land-use type (Antle *et al.*, 2003; Antle & Valdivia, 2006; Bryan *et al.*, 2009b). This heterogeneity leads to a convex supply function and is a vital component to consider when designing environmental polices (Bryan *et al.*, 2009b).

In addition, to operate in a carbon market, these ecosystem services must be certified by an independent authority before they can be sold. This may involve substantial transaction costs (Woerdman, 2001; Fitchner *et al.*, 2003; Cacho *et al.*, 2005; Cacho &

Lipper, 2007). When transaction costs are included, the supply curve shifts up and to the left, reducing the size of the market and increasing the price required to achieve a given level of mitigation. Although the impact of transaction costs on supply curves is well understood, only a few quantitative analyses have recently been undertaken (Cacho *et al.*, 2013).

To evaluate the feasibility of landholders' participation in carbon markets, both abatement and transaction costs must be considered (Cacho *et al.*, 2005). There is strong evidence that significant potential exists for planting trees to achieve low-cost abatement in Australia (Kirschbaum, 2000; Harper *et al.*, 2007; Lawson *et al.*, 2008; Polglase *et al.*, 2008; Burns *et al.*, 2009). However, despite this potential, and the availability of a mechanism to allow credits from the production of offsets from farms (Macintosh & Waugh, 2012), Mitchell *et al.* (2012) found a significant disparity between the estimated potential and actual uptake of these projects by Australian landholders.

Producing creditable offsets has many practical and policy constraints, including additionality (Sedjo & Sohngen, 2012; Mason & Plantinga, 2013), permanence (Yemshanov *et al.*, 2012; Macintosh, 2013), leakage (Chomitz, 2002; Murray *et al.*, 2007), perverse impacts (Jackson *et al.*, 2005; Bradshaw *et al.*, 2013), spatial heterogeneity (Paul *et al.*, 2013b) and transaction costs (van Kooten *et al.*, 2002; Cacho *et al.*, 2013; McCann, 2013). These issues may be limiting landholder uptake and are discussed in more detail throughout this thesis.

Cacho *et al.* (2008b) noted that it will be necessary for landholders to receive adequate incentives to encourage these changes in land use. They also emphasised that high transaction costs associated with participation may be prohibitive and that most landholders would be unable to directly participate in carbon markets. Several authors

have highlighted that transaction costs can be reduced by taking advantage of economies of scale (Galik *et al.*, 2012; McCann, 2013). This introduces the possibility of an aggregator who purchases offsets from a number of landholders to gain economies of scale (Cacho *et al.*, 2013) and sells these offsets in a carbon market. The feasibility of projects using such intermediaries is reliant on the carbon price received at the farm level being acceptable to both landholders and the aggregator. The transaction costs incurred and the number of contracts secured will determine the feasibility of a project for a given market carbon price.

1.2 Aims of the study

The aims of this study are to determine the:

1. potential supply of carbon offsets from land-use change by Australian farmers;
2. abatement and transaction costs for the supply of carbon offsets and their spatial variability;
3. impact of different techniques for calculating transportation costs for harvested biomass from some land-use options;
4. minimum feasible size of a carbon-offset project that would entice aggregators to act as intermediaries in carbon markets; and
5. optimal species selection, management regime and planting configuration of farm forestry in the presence of carbon markets.

These aims required a case study region with spatial diversity. The Border Rivers-Gwydir catchment in New South Wales, Australia, was selected as a case study as it is characterised by a diverse range of climatic conditions, including annual average rainfall ranging from 1,200 mm in the east to 450 mm in the west, various land types and different farm enterprises and farm sizes. Analyses are undertaken at differing levels of

resolution, ranging from 1.1 km^2 cells in a spatial grid to the farm scale. Results are presented from the perspective of both individual landholders and project aggregators.

1.3 Research context

While numerous authors have estimated the potential of farm forestry to contribute to GHG mitigation policies, both globally and within Australia, most spatial analyses have been at a country, state or regional scale (Nijnik & Bizikova, 2008; Polglase *et al.*, 2008; Eady *et al.*, 2009; Nielsen *et al.*, 2014). Few studies have estimated this potential at a finer (local) scale that accounts for current farming technologies and management practices. Paul *et al.* (2013b) found that accounting for spatial heterogeneity at a local scale allowed for the significant variation, in both the cost of and the potential for farm forestry to sequester carbon, to be identified. This thesis uses a similar approach by undertaking farm-scale analyses in a heterogeneous landscape to evaluate incentives for landholders to produce carbon offsets, which are bundled by a project aggregator for the carbon market.

Costs associated with the transport of harvested biomass are significant in determining the feasibility of carbon offset projects (Shi *et al.*, 2008). This thesis contributes to the literature by investigating the impact of travel distance, estimated using three different algorithms of increasing complexity, on transportation costs. The consequences of the choice of algorithm on the feasibility of carbon projects and on transport emissions are determined.

Previous analyses of the potential contribution of farm forestry to GHG mitigation objectives have assumed homogeneity in transaction costs. Polglase *et al.* (2008) accounted for transaction costs at a general level by applying a single allowance of $10 per hectare up-front for legal transactions and $40 annually for carbon monitoring (in

2006 AUD). Their model did not allow these costs to vary across the different farms or regions of Australia. Likewise, Flugge and Abadi (2006), in their analysis of the potential for Western Australian farmers to plant trees for carbon sequestration, limited transaction costs to a one-off upfront payment of $5,000 (in 2006 AUD), irrespective of the project size. Transaction costs can vary significantly across different farms, regions, land-use changes and policies (Boyd & Simpson, 1999; UNDP, 2006). Fitchner *et al.* (2003) found that transaction costs varied from 7% to more than 100% of the production costs of the set of carbon sequestration projects that they reviewed. Ignoring the variability in transaction costs may produce misleading results and contribute to ineffective policy design. This thesis adds to the literature through an analysis of the transaction costs incurred by participants (both landholders and project aggregators) in carbon markets and considers the spatial heterogeneity of these transaction costs across a case study catchment.

Farm forestry monocultures have been found to sequester higher levels of carbon at lower abatement costs than biodiversity plantings such as the mixed-species methodology currently approved under the Australian Carbon Farming Initiative (Hunt, 2008; Crossman *et al.*, 2011; Paul *et al.*, 2013b). This legislation imposes strict conditions (Australian Government, 2011a), limiting the eligibility for carbon credits of the standing carbon stock and the harvested products from monoculture plantations (Macintosh, 2013). This thesis contributes to knowledge by evaluating the ability of this legislation to encourage landholders to participate in carbon offset projects.

1.4 Overview of the study

This thesis is comprised of four research sub-projects, with each taking the form of an independent paper for journal submission. These papers are presented in Chapters 2 – 5.

Chapters 2, 4 and 5 are closely related in that they spatially simulate the optimal land use for farmers in the presence of carbon sequestration payments. Chapter 3 examines different methods for estimating travel distance in the base model developed in Chapter 2, to determine the optimal method which is used in the remaining chapters. Each chapter builds on the previous chapter, following a progression in complexity and spatial detail in the systems analysed and the methodology used. A schematic illustrating the flow and link between each chapter in the thesis is provided in Figure 1.1.

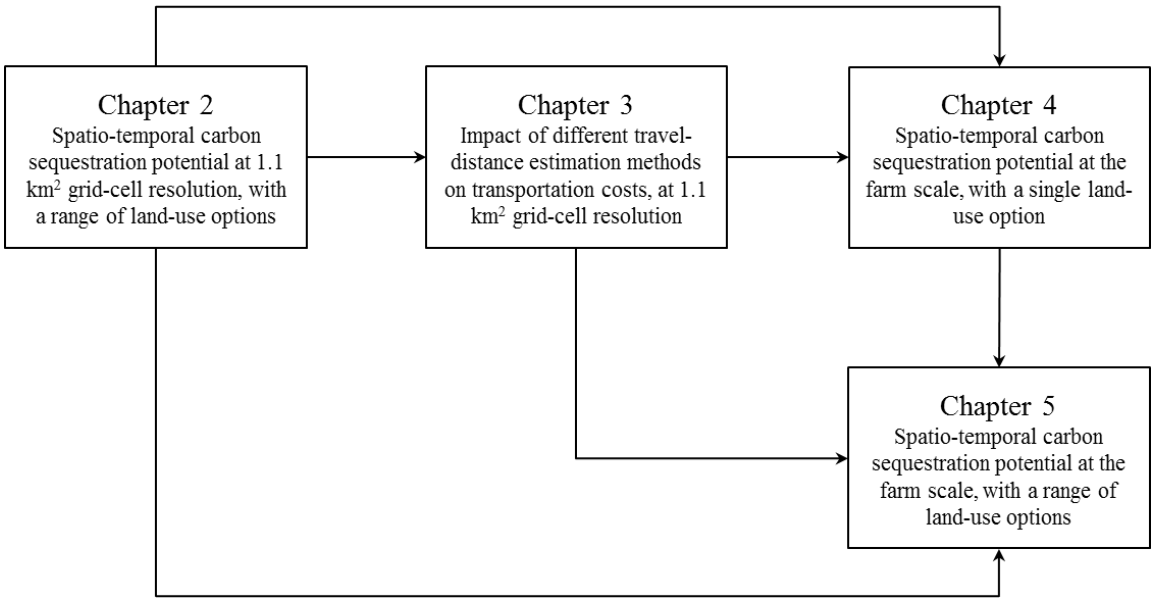

Figure 1.1: Schematic diagram demonstrating the link between chapters in the thesis.

The methodologies used in this study involve the application of computer-process models and the development of numerical models to simulate the growth of different tree species under a range of management practices, with respect to spatial, economic and technical constraints. Spatio-temporal carbon sequestration potential was estimated using two carbon accounting models based on the point-based modelling approach of the National Carbon Accounting Toolbox: FullCAM (Richards, 2001; Richards & Evans, 2004) and the Reforestation Modelling Tool (DCCEE, 2011). The numerical computing program MATLAB was used to integrate spatio-temporal biophysical results from these

models with geographic information system (GIS) data, commodity maps, production statistics and economic data. Net present value of private benefits is used as the performance measure. This means results are not necessarily socially optimal except to the extent to which the carbon price reflects the preferences of society.

It should be noted that while an attempt has been made to minimise repetitive material between the different chapters, it has not been possible to remove all repetition, especially in the descriptive sections, without compromising the integrity of the individual chapters as independent articles.

In Chapter 2, the methodology to measure the potential supply of carbon offsets from landholders is developed. This methodology is based on opportunity costs across different areas of land which are estimated using spatial commodity maps, productivity satellite images and statistical price and yield data at a 1.1 km^2 grid-cell resolution. Spatio-temporal carbon sequestration potential for several land-use changes is also estimated. This methodology allows for the combination of land-use changes with the highest technical carbon sequestration potential and the lowest opportunity cost to be estimated at a catchment level. An important finding of Chapter 2 was that the cost of transporting harvested biomass from a source location to a destination processing plant or mill is a major component of the total cost of projects that rely on harvested products.

In Chapter 3, geospatial modelling is used to compare three travel distance estimation methods and their impact on the cost of transporting harvested biomass. The extent to which the increased accuracy of complex methods justifies the additional data and processing time required is assessed. The impact of different methods on the estimation of net carbon dioxide equivalent (CO_2-e) emissions is also investigated.

The model developed in Chapter 2 identifies low-cost carbon sequestration areas at a 1.1 km^2 scale across the case study catchment. The model is extended in Chapter 4 to simulate land-use changes at a farm scale and more accurately accounts for current farming systems and paddock infrastructure. In Chapter 2, transaction costs are added as a percentage of production costs for each 1.1 km^2 grid cell, whereas in Chapter 4, the spatial resolution allows for both fixed and variable transaction cost components to be calculated for each farm. The model presented in Chapter 4 includes carbon-offset contracts between an aggregator and individual landholders and the feasibility of carbon-offset projects can be determined. The model is applied to three regions within the case study catchment, to determine the feasibility of carbon-offset projects utilising permanent native mixed-species plantings.

In Chapter 5, the farm-scale model previously developed is further extended to allow the spatial simulation of several tree species, management regimes and planting geometrics. Feasible projects are determined for a range of carbon prices. This model is applied to the same case study regions as in Chapter 4, allowing the impact of several different policy aspects to be tested.

A brief summary of the main findings, limitations of the study and recommendations for future research are discussed in Chapter 6.

Several appendices are presented: (A) a list of the shapefile spatial datasets used to estimate optimal travel distances described in Chapter 3; (B) a sample application of the Dijkstra and least-cost methods for estimating travel distance; (C) a detailed description of the mathematical model used in Chapters 4 and 5 to determine paddock opportunity costs; (D) parameter values for the estimation of these opportunity costs; (E) a list of the

transaction costs described in Chapter 5; and (F) a sample of the carbon trajectories through time for different tree species and management practices described in Chapter 5.

Finally, a CD-ROM containing supplementary information, including a Microsoft Excel workbook with full details of the 17 livestock gross margins for calculating the opportunity costs in Chapters 4 and 5, a Microsoft Excel workbook with the full details of planting costs for the 22 species and management regime combinations in Chapter 5 and a PDF document containing the carbon trajectories for each of these combinations, is attached to the inside back cover of this thesis.

CHAPTER 2: SPATIO-TEMPORAL CARBON SEQUESTRATION POTENTIAL FROM LAND-USE CHANGE

2.1 Introduction

Greenhouse gas (GHG) concentrations in the atmosphere have risen from 280 parts per million (ppm) to about 400ppm since the industrial revolution (Jones 2013). These concentrations are predicted to continue rising, with anthropogenic influences considered to have contributed to this increase (IPCC, 2007, 2013). There is concern that this increase in atmospheric GHG levels is causing a rise in the earth's temperature. The 2009 Copenhagen Accord suggested that a desirable temperature target is a global mean temperature rise of $2^{o}C$ (Randalls, 2010). These findings are driving a global emphasis on the development of technologies and policies to reduce GHG emissions. Carbon dioxide (CO_2) emissions contribute 90% of the annual increase in GHGs (Hansen & Sato, 2004), therefore its mitigation has received much of the attention in the policy debate.

The agricultural sector has the potential to provide significant contributions to any GHG reduction policies, as it is both a source and a sink of GHG emissions. There is evidence that modification of current agricultural management practices is a relatively low-cost method of offsetting GHG emissions (Antle *et al.*, 2002). There are several ways in which the agricultural industry may contribute, including: (a) reduction of farm emissions (Foley *et al.*, 2011); (b) increase in farm soil carbon (Denef *et al.*, 2008; Varvel & Wilhelm, 2011); (c) production of bioenergy (Cook & Beyea, 2000); (d) production of second-generation biofuels (Walter & Ensinas, 2010); (e) capture of CO_2 through plantations (Kirschbaum, 2000; Golub *et al.*, 2009; Wise & Cacho, 2011); and (f) removal of CO_2 from the atmosphere through the use of conservation forests (Crossman *et al.*, 2011).

Trees grown for the purpose of sequestering carbon have lower marginal costs than emission abatement and other mitigation techniques, in many situations (Plantinga *et al.*, 1999; Stavins, 1999; Kauppi & Sedjo, 2001; Sohngen & Brown, 2006; Golub *et al.*, 2009). The potential supply of carbon offsets depends on the opportunity costs of available land-use options which are partly determined by location. An economically rational landholder will only supply carbon mitigation through land-use change if they receive incentives which outweigh their opportunity costs. Landholders' opportunity costs for converting their properties to carbon mitigating land uses can be aggregated to provide a marginal abatement cost curve for the potential supply of carbon mitigation by land-use change and forestry (LUCF) activities (Antle & Valdivia, 2006). Economic returns will differ between land uses and may vary spatially for a particular land use depending on location (Antle *et al.*, 2003; Antle & Valdivia, 2006; Bryan *et al.*, 2009b). This heterogeneity leads to a convex supply function and is a vital component in the toolbox for designing and evaluating environmental polices (Bryan *et al.*, 2009b).

To operate in a carbon market, the mitigation associated with land-use change must be certified by an independent authority before it can be credited (Cacho *et al.*, 2008b). This may involve substantial transaction costs (Woerdman, 2001; Fitchner *et al.*, 2003; Cacho *et al.*, 2005; Cacho & Lipper, 2007; Cacho, 2009) that will shift the supply curve upwards and to the left, resulting in higher carbon prices. Scientific, institutional and economic factors will influence these incentives.

The aims of this chapter are to investigate the possible supply of carbon mitigation services from land-use change, with an emphasis on spatial heterogeneity, and to identify the areas with low-cost sequestration potential and the optimal land-use changes when incentives for carbon sequestration are provided. This analysis was applied in a case study to the Border Rivers-Gwydir catchment in New South Wales, Australia.

2.2 Materials and method

2.2.1 Study area

The Border Rivers-Gwydir catchment lies within the Murray-Darling Basin in northern New South Wales, Australia (Figure 2.1) and covers approximately 5,000,000 hectares (ha). The catchment is characterised by a diverse range of climatic conditions and land types. Average annual rainfall ranges from 1,200 mm in the east to 450 mm in the west (Australian Bureau of Meterology, 2010). Elevation ranges from 1,450 metres above sea level (MASL) on the eastern boundary down to about 150 MASL in the western plains. Soils in the catchment are predominately of low to moderate fertility (Scott *et al.*, 2004) and include grey, black and brown vertosols, red, brown, grey and yellow chromosols, sandy tenosols, kandosols, red ferrosols and dermosols (Isbell, 2002; DCCW, 2010). The average property size across the region is 1,328 ha (standard deviation of 1,558 ha) with properties ranging up to 31,961 ha (NSW Government, 2011). The current land uses in the catchment are presented in Table 2.1(BRS, 2009).

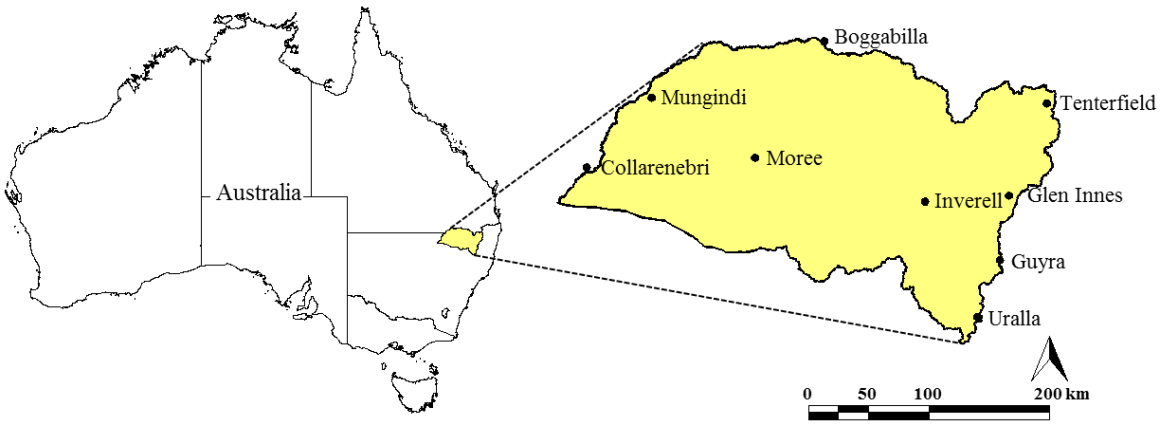

Figure 2.1: Location of the Border Rivers-Gwydir catchment in Australia and the major towns.

Currently, the major land uses are grazing of modified and native pastures and dryland cropping. Grazing enterprises dominate the tablelands in the eastern sections of the catchment, while cropping predominately occurs on the slopes and plains in the western half of the catchment (NSW Government, 2009a).

Table 2.1: Current land uses in the case study catchment (NSW Government, 2009a).

Current activity	Total area (ha)
Grazing of modified pastures	1,534,700
Cropping	1,152,300
Other minimal uses	533,400
Grazing of native pastures	490,000
Irrigated pastures and cropping	184,500
Water	92,000
Forestry	67,700
Nature conservation	58,600
Urban intensive uses	43,000
Rural residential	17,400
Plantations	13,300
Managed resource protected areas	12,500
Irrigated horticulture	1,700
Mining and waste	1,400
Intensive animal and plant production	800
Horticulture	700
Land in transition	600

The analysis presented in this chapter is based on the integration of several models to estimate expected carbon sequestration for a 61-year project period[a] from current land uses in the catchment and a range of proposed land-use changes simulated through a set of computer experiments. The abatement costs in terms of dollars per tonne of carbon sequestered are calculated based on the results of these experiments.

2.2.2 Land-use changes for carbon sequestration

Five proposed land-use change options and management practices proposed by Richards (2001) to increase carbon stocks relative to current practice in the region are investigated. These include the establishment of (1) *Pinus radiata* plantations for commercial soft-wood production; (2) mixed-species plantings for environmental

[a] Due to the data intensive nature of the spatio-temporal simulations in this chapter, the commonly used 100-year project length was impractical so a shorter (61-year) simulation period was adopted. Simulations for 100-year periods are undertaken in Chapters 4 and 5.

purposes with no commercial harvesting potential; (3) a generic eucalypt species plantation for pulpwood and hardwood timber production; (4) *Eucalyptus cladocalyx* plantations for commercial hardwood production; and (5) the conversion of conventionally-tilled cropland to no-tillage systems.

The carbon eligible for credit payment is measured relative to a baseline activity, which is normally the current land use. That is, payments apply only for actions that would not have been undertaken in the absence of the payment. The quantity of eligible carbon (C_t) for a given year is (Cacho *et al.*, 2008b)[b]:

$$C_t = C_{P,t} - C_{C,t} \tag{2.1}$$

where $C_{P,t}$ and $C_{C,t}$ are the carbon stocks of the proposed land use and the current land use in year t, respectively. The FullCAM model (version 3.13.8 – research edition), developed by the Australian Greenhouse Office (Richards, 2001), was used to determine annual carbon stocks for baseline systems and for each of the proposed land uses. The FullCAM model is calibrated for Australian regions and land-use types (Paul & Polglase, 2004; Paul *et al.*, 2008). Utilising the same spatial resolution as Polglase *et al*. (2008) and Bryan *et al*. (2009a), simulations were undertaken for 42,046 square cells of 1.1 km^2 representing the catchment as a set of raster map layers.

Simulations indicated that the change from conventionally-tilled to no-tillage systems produced only small gains in carbon stocks. These findings are consistent with the literature which indicates carbon sequestration through soils is a slow process and inefficient for timely, large-scale capture (Hermle *et al.*, 2008; Matsumoto *et al.*, 2008; Young *et al.*, 2009; Dalal *et al.*, 2011). No further analysis was undertaken.

[b] Equation (2.1) assumes that carbon captured in biomass will be released back into the atmosphere at the time of harvest. Wood products can continue to store carbon for long periods after harvest (Ingerson, 2011) and may also be used as a source of bioenergy (Cherubini *et al.*, 2009). This model does not capture these characteristics, resulting in conservative total carbon capture estimates.

2.2.3 Abatement costs

Abatement costs are the cost of producing one unit of (uncertified) carbon sequestration, estimated as the opportunity cost of undertaking the carbon-sequestration activity rather than the current activity (Cacho *et al.*, 2008a):

$$v_{Aj} = a_j \sum_t \left(r_{0t} - r_{jt}\right)\left(1 + \delta_S\right)^{-t} \tag{2.2}$$

where for the jth location in the region, $v_{\text{A}j}$ is the abatement cost, a_j is the total area of the land-use change, r_{0t} and r_{jt} represent the net revenues per hectare in year t for the baseline and proposed land-use change respectively, and δ_S is the landholder's discount rate. Equation (2.2) applies a discount rate to provide abatement costs in terms of net present value.

2.2.3.1 Estimation of baseline revenues

The methodology outlined in Bryan *et al.* (2009a) was adopted in this study to derive the baseline revenues per hectare (r_{0t}). This method involves integration of agricultural yield and price statistics (ABS, 2008; NSW Department of Industry and Investment, 2009; ABARES, 2012), and remote sensing data and land-use maps (BRS, 2009) into a geographic information system (GIS) to calculate the agricultural returns with a profit function at a 1.1 km^2 scale across the region.

The first step of this process was to combine the land-use maps and remote sensing data with the 2006-2007 agricultural census data from the Australian Bureau of Statistics (ABS) using the methodology described by Bryan *et al.* (2009a) to generate commodity maps specifying the location of current agricultural enterprises on a 1.1 km^2 cell-grid

resolution. With these commodity maps, the baseline net revenue per hectare was then determined with the following equation adapted from Bryan *et al.* (2009b)[c]:

$$r_0 = q_1(\beta p_1 + p_2 q_2) - (w_q q_1 + w_a) - (k_O + k_L + k_D) \tag{2.3}$$

where all variables are dependent on the current enterprise at the 1.1 km^2 scale resolution and q_1 is the yield of the primary product for livestock in terms of dry-sheep equivalents (DSE ha^{-1} yr^{-1}) or crops (t ha^{-1} yr^{-1}), q_2 is the secondary product yield for livestock enterprises, which includes wool (kg DSE^{-1} yr^{-1}) or milk (litres DSE^{-1} yr^{-1}), β is the percentage of livestock or crop sold per year, p_1 is the farm-gate price of the enterprises primary product (\$ DSE^{-1}) or (\$ t^{-1}) for livestock and crops, respectively, p_2 is farm-gate price for the secondary enterprise product in terms of \$ kg^{-1} (wool) or \$ litres^{-1} (milk), w_q is the production quantity dependent variable costs for the enterprise in \$ DSE^{-1} yr^{-1} (livestock) or \$ t^{-1} yr^{-1} (cropping), w_a is the variable costs dependent on production area (\$ ha^{-1} yr^{-1}) and k_O, k_L and k_D are the fixed costs in terms of operating, imputed labour and depreciation expenses (all expressed in terms of \$ ha^{-1} yr^{-1}).

A base yield for each enterprise was estimated by dividing total production for each product in a statistical local area (SLA) by the total area of production[d]. In order to capture spatial variance in yield, a greenness index from remote sensing was used in conjunction with historical yield data gathered from the ABS. Normalised difference vegetation index (NDVI) represents vegetation greenness and has been widely adapted as a proxy for estimating agricultural production yields spatially (Bryan *et al.*, 2009a; Bryan *et al.*, 2009b; Wu *et al.*, 2009; Marinoni *et al.*, 2012; Gu *et al.*, 2013). Base yields were adjusted using the maximum monthly NDVI scores from long-term average data and the following equation defined by Bryan *et al.* (2009b):

[c] While the study by Bryan *et al.* (2009b) explicitly specifies water use and water costs in their model, this model captures these aspects through the variable cost estimates rather than accounting for them as separate variables.
[d] In this study, this was derived for the period 2006-2007 (ABS, 2008).

$$q_j = \frac{P\varphi_j n}{\sum_{j=1}^{n} \varphi_j} \tag{2.4}$$

where q_j is the adjusted yield of the agricultural products on the j-th grid cell (t ha^{-1} yr^{-1} or DSE ha^{-1} yr^{-1}), P is the mean yield of the agricultural product (t ha^{-1} yr^{-1} or DSE ha^{-1} yr^{-1}), φ_j is the maximum monthly NDVI index for the j-th grid cell and n is the total number of grid cells classified.

The yield sold each year (β) is the proportion of livestock sold. This is determined as the number of livestock sold divided by the total number of livestock pre-sale. This data was derived from the ABS census data for each SLA. For cropping and horticultural enterprises, it was assumed that the entire yield was sold, so β was set to 1.

Farm-gate revenues for the primary and secondary products of each enterprise were estimated on a per unit of production basis. These were calculated by dividing the total value of production by the total quantity of production across each SLA. Fixed and variable cost parameters were determined from published gross margin budgets (NSW Department of Industry and Investment, 2009) and from the Australian Bureau of Agricultural and Resource Economics and Sciences (ABARES) farm economic database (ABARES, 2012).

Figure 2.2 summarises the process and the corresponding inputs required to estimate the spatially explicit baseline revenues using this methodology.

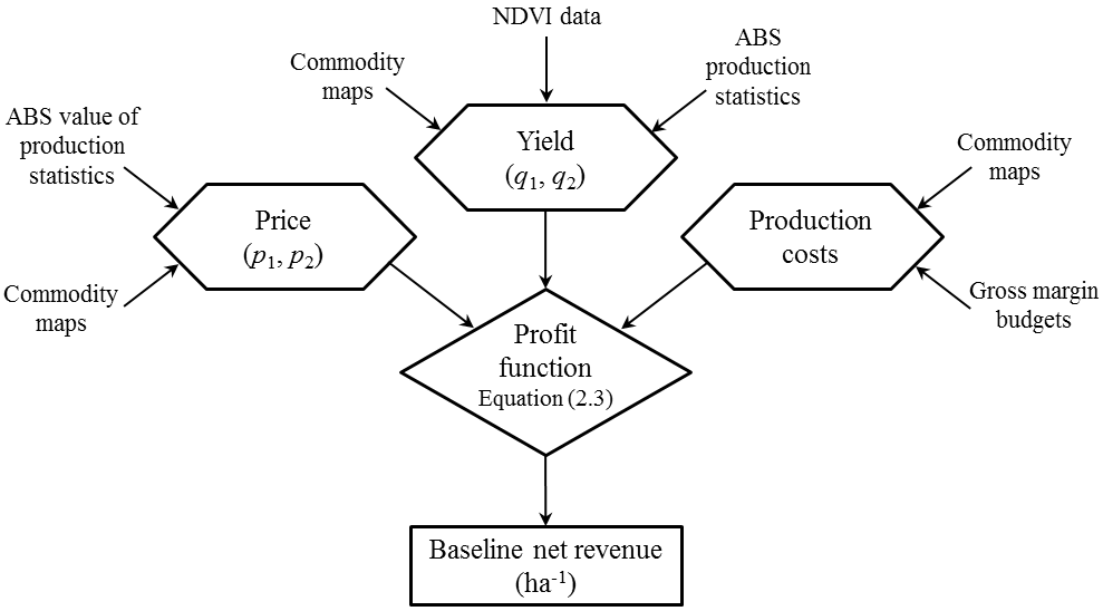

Figure 2.2: Data inputs and structure of the profit function to determine the baseline revenue (r_0) at a 1.1 km^2 grid-cell resolution (adapted from Bryan *et al.*, 2009b, p. 379).

All values used to determine baseline revenue were converted to 2010 Australian dollars using the Consumer Price Index (ABS, 2011) and the baseline revenue for each grid cell replicated for the project length.

2.2.3.2 Estimation of proposed land-use change net revenues

The net revenues for the proposed land-use change strategies were determined by constructing budgets based on the expected expenses and income from each land use, on a per hectare basis. Data for these inputs were derived from Polglase *et al*. (2008), with all values converted to 2010 dollars (Table 2.2).

Table 2.2: Parameters used to determine the net revenues of the proposed land-use changes.

Parameters	Units	Proposed land use			
		P. radiata	Mixed species	Generic eucalypt	*E. cladocalyx*
Establishment					
Planting costs	$ ha^{-1}	1,923	801	1,603	1,282
Follow-up weed control	$ ha^{-1}	160	80	160	160
Annual management	$ ha^{-1}	85	16	85	32
Pruning	$ ha^{-1}	168	NA	224	224
Thinning [#]					
Thinning cost	$ ha^{-1}	240	NA	321	321
Thin II					
Percentage grade C sawlogs	%	30	NA	0	30
Percentage pulpwood	%	70	NA	0	70
Thin III					
Percentage grade B sawlogs	%	0	NA	NA	NA
Percentage grade C sawlogs	%	30	NA	NA	NA
Percentage pulpwood	%	70	NA	NA	NA
Thin IV					
Percentage grade B sawlogs	%	40	NA	NA	NA
Percentage pulpwood	%	60	NA	NA	NA
Clear fall harvest					
Harvest cost	$ m^{-3}	16	NA	18	18
Percentage grade A sawlogs	%	0	NA	40	60
Percentage grade B sawlogs	%	80	NA	26	0
Percentage grade C sawlogs		0	NA	17	0
Percentage pulpwood	%	20	NA	17	40
Price grade A sawlogs	$ m^{-3}	69	NA	102	102
Price grade B sawlogs	$ m^{-3}	64	NA	91	0
Price grade C sawlogs	$ m^{-3}	59	NA	80	80
Price pulpwood	$ m^{-3}	56	NA	80	105
Cartage cost	$ m^{-3} km^{-1}	0.13	NA	0.13	0.13

[#] **Note: products from the first thinning of all species were assumed to have no commercial value.**

Transportation costs can be a significant factor in land-use options with harvested biomass (Searcy *et al.*, 2007). To estimate these costs it was necessary to determine the distance between the location of each biomass source and the destination processing mill.

This distance was determined using the Orthodromic distance method; a technique that estimates the shortest distance between two geographical coordinates. Mathematically, this can be determined using the Haversine formula:

$$d_{m:n} = 111.12 \cdot 2\arcsin\left(\sqrt{\sin^2\left(\frac{\varphi_m - \varphi_n}{2}\right) + \cos\varphi_m \cos\varphi_n \sin^2\left(\frac{\lambda_m - \lambda_n}{2}\right)}\right) \tag{2.5}$$

where d is the distance between the source location m and the location of the mill n, in kilometres; 111.12 is a conversion factor to convert degrees to kilometres; φ_m and φ_n are the latitudes in radians of m and n, respectively; and λ_m and λ_n are the longitudes in radians of m and n, respectively.

The optimal mill for each source location was selected by minimising the travel distance to an existing processing mill. This technique calculates the travel route 'as the crow flies', so provides an inaccurate estimation of the likely distance between locations. However, it has the advantage of requiring no information on the road network.

There are currently 10 operational timber mills within proximity of the case study catchment (Figure 2.3). None of these mills are actually located within the catchment.

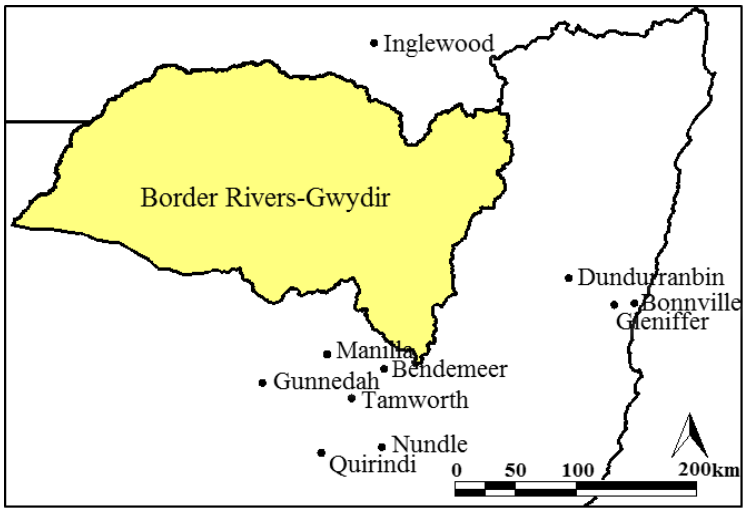

Figure 2.3: Location of existing mills relative to the case study catchment.

2.2.4 Supply curves

In order to derive curves for the supply of carbon, the following steps were undertaken:

1. The opportunity cost (v_A) of undertaking each of the proposed land-use changes was determined across the catchment by applying equation (2.2) to each 1.1 km^2 cell. From these opportunity costs, the proposed land-use system providing the lowest-cost strategy was determined for each source location.
2. The average quantity of additional carbon that can be expected from this lowest-cost strategy was converted to carbon dioxide equivalents (CO_2-e), the unit of measurement applied in emissions trading schemes, by multiplying this additional carbon by 3.67[e].
3. The set of v_A values were sorted in ascending order based on cost per tonne of CO_2-e and the cumulative sum of carbon above the baseline was calculated. The plot of v_A against the corresponding cumulative quantity is the supply curve.

2.3 Results and discussion

2.3.1 Carbon sequestration projections

The regional averages and ranges of changes[f] in carbon biomass above the baseline activities are shown in Figure 2.4 for each of the proposed land-use changes (in which different scales are used in the plots to enhance readability). There are substantial differences in the trajectories of carbon stocks through time. For example, *P. radiata* (Figure 2.4 (A)) exhibits sharp declines due to commercial management practices such as thinning and clear-felling which remove carbon. In contrast, mixed-species

[e] The photosynthesis process of trees converts 3.67 units of CO_2 into one unit of biomass carbon.
[f] Confidence intervals of 5% and 95% are presented in this figure to demonstrate the spatial variance in carbon sequestration potential across the catchment.

environmental plantings (Figure 2.4 (B)) result in monotonic increases in carbon stocks through time.

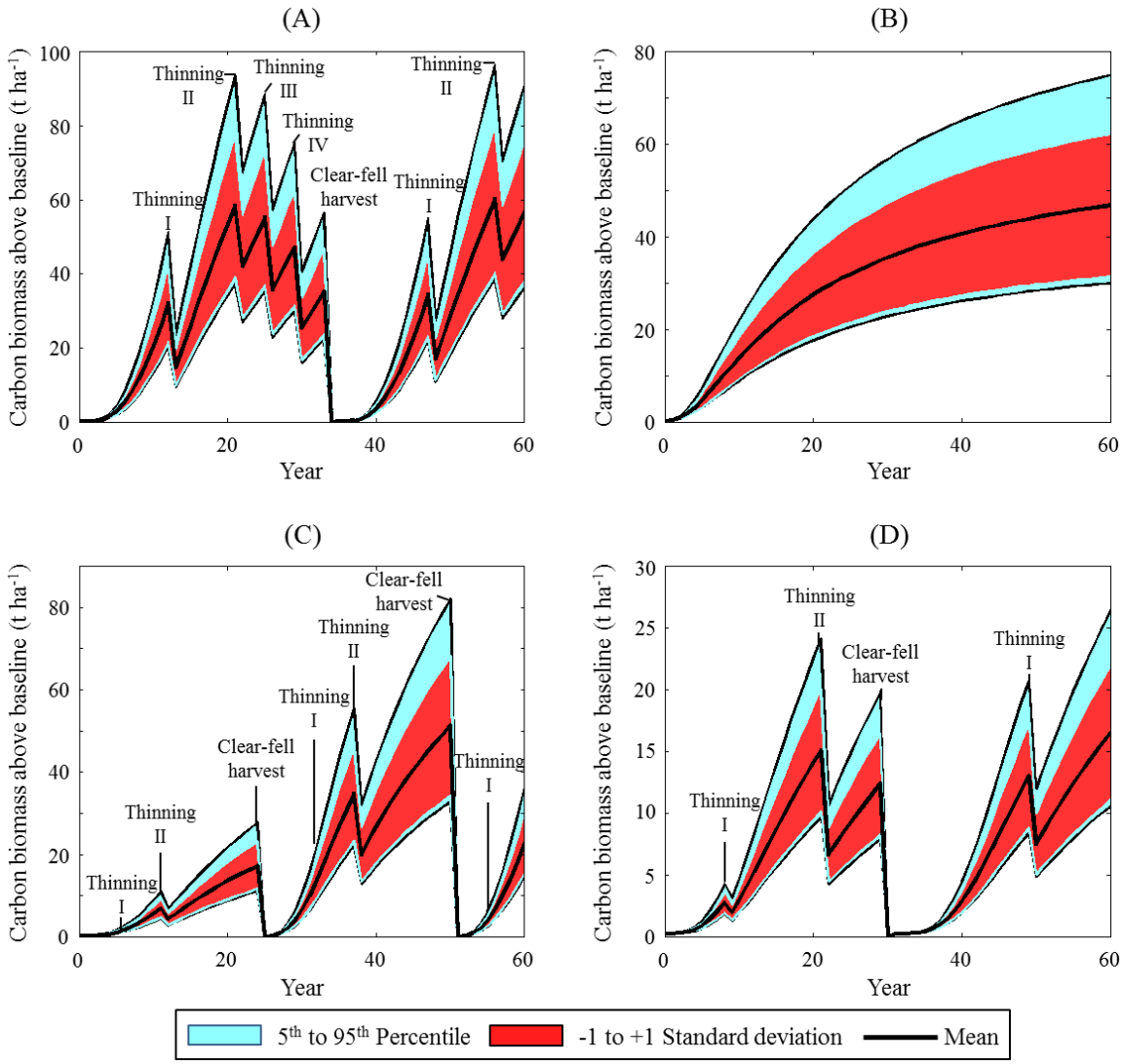

Figure 2.4: Carbon sequestration through time for proposed land uses: (A) *P. radiata* plantations; (B) mixed-species environmental plantings; (C) generic eucalypt plantations; and (D) *E. cladocalyx* plantations.

There is considerable spatial variability within land uses, which is illustrated in Figure 2.5. This variation occurs both across the region and between the proposed land-use changes (see also Table 2.3). Figure 2.5 and Table 2.3 highlight the fact that the highest quantity of carbon is captured through the mixed-species environmental planting, closely followed by the *P. radiata* plantation option. The lowest carbon sequestration potential was found to occur with the *E. cladocalyx* plantation option, with an average

sequestration potential of just over 10% of that of the mixed-species environmental planting.

Table 2.3: Additional average carbon sequestered over the 61-year simulation period, compared with the baseline carbon that would be sequestered by maintaining the current land use.

	Mean[#]	St. dev.[#]	Min[#]	Max[#]	Coefficient of variation (%)
P. radiata plantations	22.48	7.97	7.29	73.88	35.45
Mixed spp. environmental plantings	26.36	9.23	8.91	85.99	35.02
Generic eucalypt plantations	10.51	4.24	2.98	36.44	40.34
E. cladocalyx plantations	2.44	2.28	-4.00	15.28	93.44

[#] All units expressed in tonnes of additional carbon sequestered per hectare over the simulation period.

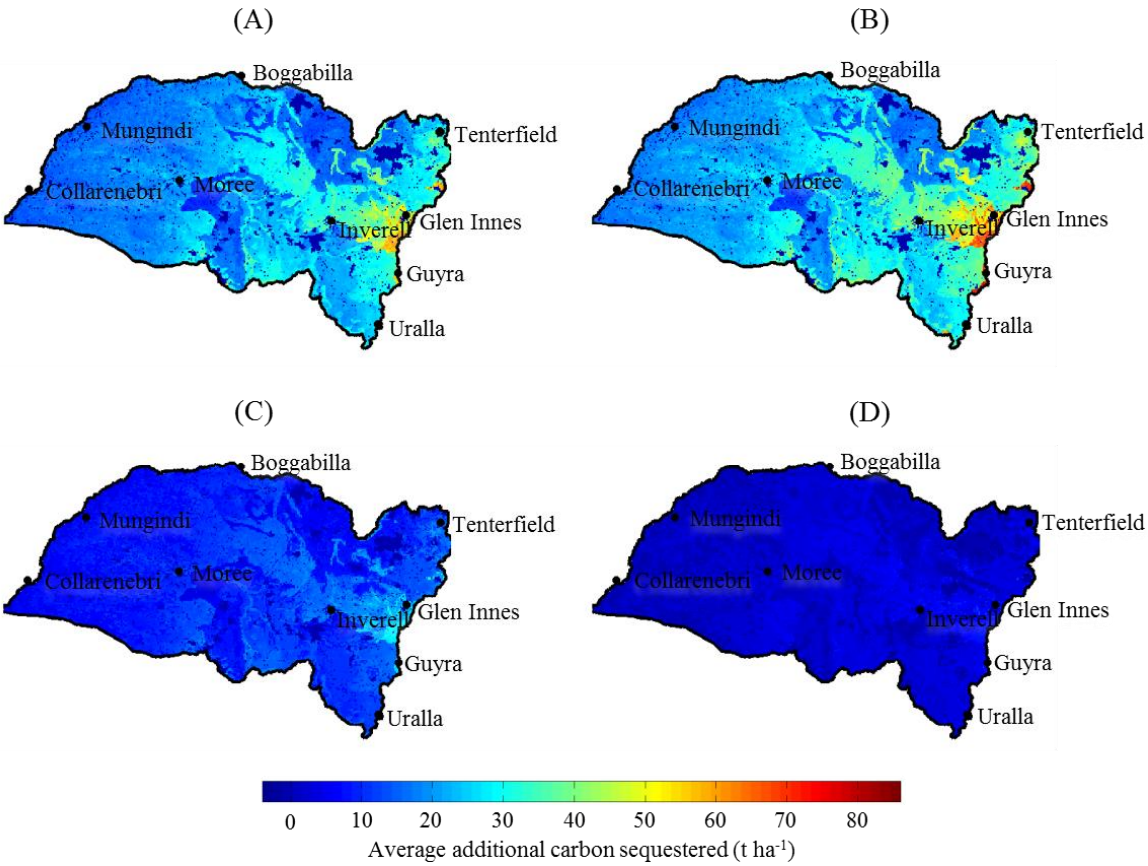

Figure 2.5: Average additional carbon sequestered over the 61-year simulation period for proposed land uses (A) *P. radiata* plantations; (B) mixed-species environmental plantings; (C) generic eucalyptus plantations; and (D) *E. cladocalyx* plantations.

For the mixed-species environmental plantings (Figure 2.5(B)), there are 9,900 ha in the catchment that could achieve additional carbon sequestration levels exceeding 70 tonnes

of carbon per ha, if this land-use option were to be fully adopted. Results indicate that 1,900 ha of these occur around the township of Deepwater, a small town 40 kilometres north of Glen Innes. The remainder of this high carbon sequestration potential area occurs along the eastern boundary of the catchment from Glen Innes to just south of Guyra.

2.3.2 Supply of carbon abatement

The opportunity cost of changing to one of the proposed land uses depends on the current land use, along with the potential net revenue that could be expected from the proposed land-use change. The average net present values per hectare for all the land uses and for a range of discount rates are shown in Table 2.4. With a discount rate of 5% or higher, all the proposed land uses are less profitable than the current land use[g]. This suggests that some form of compensation would be required to entice landholders to undertake the proposed land-use changes. While *E. cladocalyx* plantations had the highest net present value, they also had the lowest additional carbon sequestration potential.

Table 2.4: Average net present values ($ ha^{-1}) of the proposed land uses and current land use over a 61-year simulation period for a range of discount rates.

	Discount rate					
	1%	**3%**	**5%**	**7%**	**9%**	**11%**
P. radiata plantations	-451	-1,426	-1,954	-2,233	-2,372	-2,431
Mixed spp. environmental plantings	-1,585	-1,307	-1,166	-1,086	-1,037	-1,004
Generic eucalypt plantations	578	-860	-1,595	-1,974	-2,164	-2,251
E. cladocalyx plantations	5,249	2,413	742	-235	-803	-1,131
Current land use	1,959	1,238	873	669	544	462

[g] Table 2.4 shows the average net present value of *E. cladocalyx* to be higher than the current land use at discount rates of 1% and 3%. This leads to the question, why are these landholders not already undertaking this activity? A possible explanation is these discount rates are lower than the rates that would usually be applied to analyses comparing private landholder decisions. In a study on the establishment of new forests on agricultural land in Australia, Polglase *et al*. (2013) argued that a discount rate closer to 10% is more appropriate.

The opportunity cost of each land-use change was measured in terms of price per hectare ($ ha^{-1}), price per tonne of carbon ($ t C^{-1}) and price per tonne of carbon dioxide equivalents ($ t CO_2-e^{-1}). The average opportunity cost ranges from -$283 to $349 t CO_2-e^{-1}. The lowest average opportunity cost, for all units of measurement and all but the 1% and 3% discount rates, was associated with *E. cladocalyx* plantations (Table 2.5). However, across the different land-use systems simulated, and across all discount rates assessed, no single proposed land-use change provided the minimum-cost strategy across the entire catchment.

Table 2.5: Summary of average opportunity costs for the proposed land-use changes, in terms of price per hectare and price per tonne of additional carbon and per CO_2 equivalents sequestered.

		Discount rate					
Units	**Proposed land use**	**1%**	**3%**	**5%**	**7%**	**9%**	**11%**
$ ha^{-1}	*P. radiata* plantations	2,410	2,664	2,827	2,903	2,916	2,893
	Mixed spp. environmental plantings	3,545	2,545	2,039	1,756	1,582	1,467
	Generic eucalypt plantations	1,382	2,098	2,468	2,644	2,709	2,713
	E. cladocalyx plantations	-3,289	-1,175	132	904	1,347	1,593
$ t C^{-1}	*P. radiata* plantations	149	147	147	147	145	143
	Mixed spp. environmental plantings	140	101	82	71	64	59
	Generic eucalypt plantations	233	271	292	301	302	299
	E. cladocalyx plantations	1,280	276	-344	-711	-922	-1,039
$ t CO_2-e^{-1}	*P. radiata* plantations	41	40	40	40	40	39
	Mixed spp. environmental plantings	38	28	22	19	17	16
	Generic eucalypt plantations	64	74	80	82	82	82
	E. cladocalyx plantations	349	75	-94	-194	-251	-283

[#] **All units expressed in terms of average price over the total 61-year simulation period.**

Table 2.6 displays the area of each proposed land-use option that contributes to the lowest-cost strategy across the catchment for a range of discount rates. Only 3,897,400 ha are eligible for land-use changes; other areas are designated national park or not available for agricultural use. For a 1% discount rate, *E. cladocalyx* plantations are the lowest-cost option across 2.3 million ha (60.03%), mixed-species environmental

plantings is the lowest-cost option for 1.2 million ha (29.68%) and generic eucalypt plantations and *P. radiata* plantations are the lowest-cost options for 0.25 million ha (6.36%) and 0.15 million ha (3.93%), respectively. As the discount rate increases, the areas of *P. radiata*, *E. cladocalyx* and the generic eucalypt species plantations decline, while the mixed-species environmental plantings increase. At an 11% discount rate, the mixed-species environmental plantings are the lowest-cost option for 2.8 million ha (72.95%) of the catchment, *E. cladocalyx* plantations for 1.0 million ha (26.85%) and the generic eucalypt species and *P. radiata* plantations for only 0.008 million ha (0.20%). It is interesting to note that, at a lower discount rate, the mixed-species environmental plantings are mainly constrained to the western part of the catchment (not shown). As the discount rate increases, this proposed land-use option expands towards the east, with *E. cladocalyx* plantations expanding towards the west of the catchment.

Table 2.6: Total area (ha) within the catchment where each of the proposed land-use change forms the minimum-cost strategy when viewed in terms of \$ t CO_2-e^{-1}.

	Discount rate					
Proposed land-use change	**1%**	**3%**	**5%**	**7%**	**9%**	**11%**
P. radiata plantations	153,100	112,500	62,600	3,700	4,800	7,600
Mixed spp. environmental plantings	1,156,900	1,278,500	1,389,000	2,029,500	2,516,400	2,843,200
Generic eucalypt plantations	247,700	130,700	15,200	3,600	100	100
E. cladocalyx plantations	2,339,700	2,375,700	2,430,600	1,860,600	1,376,100	1,046,500
Total eligible area	3,897,400	3,897,400	3,897,400	3,897,400	3,897,400	3,897,400

These findings highlight two important issues. One being that the choice of discount rate can greatly influence the lowest-cost strategy, and the second being that any policy to encourage carbon sequestration from land-use changes needs to allow for a range of land-use options.

Results indicate that the land-use change and geographical regions providing the highest carbon sequestration potential may not necessarily provide the lowest cost. This is

illustrated by a map of the additional CO_2-e sequestered if the lowest-cost strategy is adopted (Figure 2.6 (A)), presented next to the opportunity cost of sequestering this additional CO2-e (Figure 2.6 (B)). It can be seen that the regions both north and south of Guyra, which exhibit the highest CO_2-e sequestration potential (Figure 2.6 (A)), do not have the lowest opportunity cost (Figure 2.6 (B)). In fact, the region to the south-west of Boggabilla, which has a relatively low CO_2-e sequestration potential, is a lower-cost region for sequestering CO_2-e. This result is not unexpected as an area with higher CO_2-e sequestration potential could also be expected to have higher production of agricultural outputs.

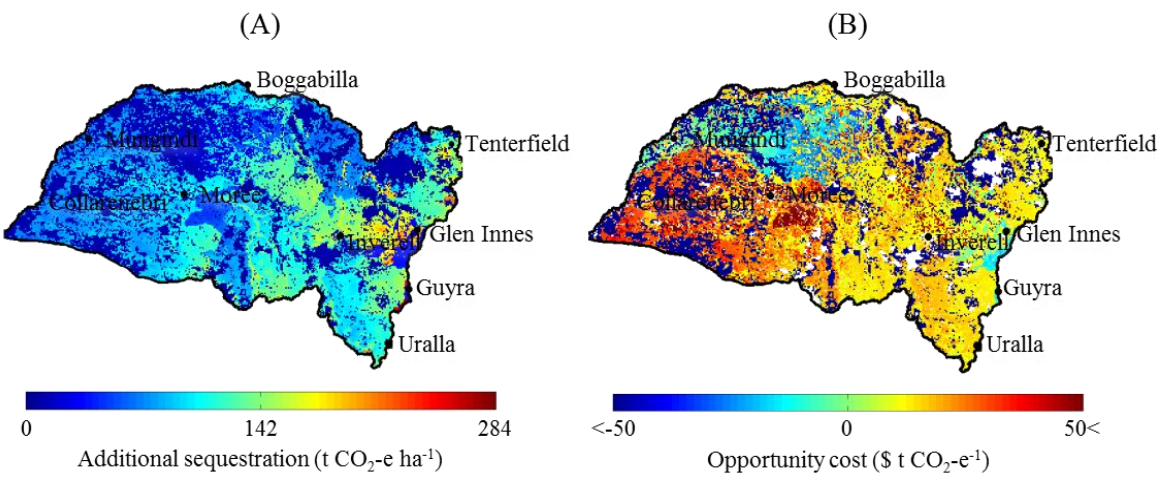

Figure 2.6: (A) The average additional CO_2-e for the lowest-cost land-use strategy; and (B) the opportunity cost of each tonne of additional CO_2-e, for a discount rate of 9%.

The supply curves derived from this analysis are shown in Figure 2.7. These curves demonstrate increasing marginal cost of carbon sequestration, from which the incentives required to encourage landholders to participate in carbon markets can be estimated. The discount rate drastically influences both the shape and position of the supply curves. However, it is important to note that transaction costs are not included when estimating these supply curves and their exact effect on the size and shape of the curves is an empirical question which is addressed in Chapter 4.

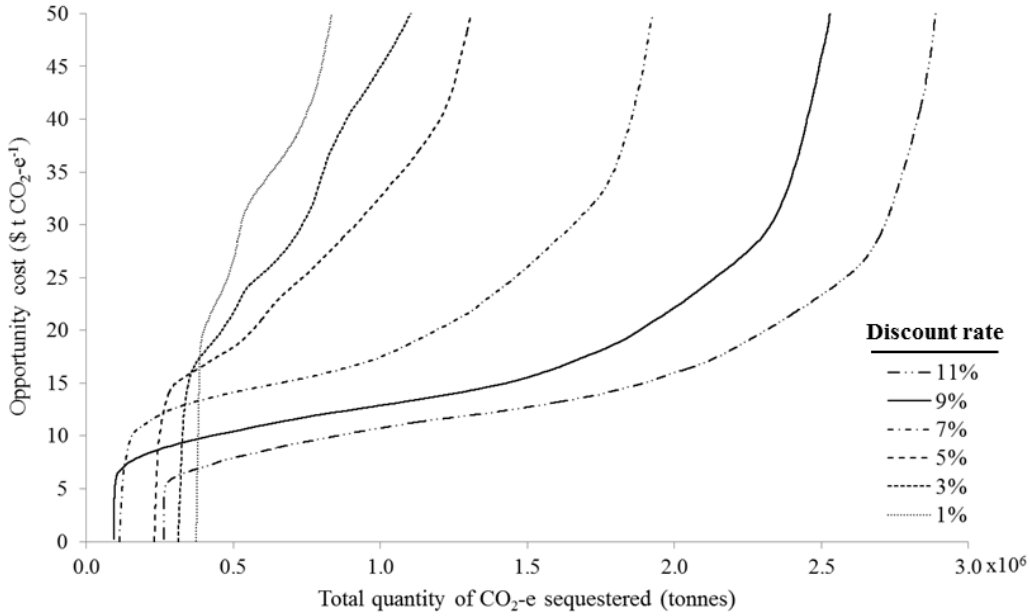

Figure 2.7: Estimated supply curves for the additional CO_2-e from the lowest-cost land-use strategies for the six different discount rates.

There is a great deal of uncertainty regarding the global price of carbon. The modelling in this chapter allows the quantity of CO_2-e that could feasibly be sequestered at different carbon prices to be estimated, assuming that landholders will accept a payment equivalent to their opportunity cost. The estimated total quantities of CO_2-e sequestered over a 61-year project, for a range of prices and discount rates are shown in Table 2.7. For example, if a 9% discount rate is assumed, approximately 423,000 tonnes could feasibly be sequestered at a price of \$10 t CO_2-e^{-1}. If, however, this price is increased to \$20 t CO_2-e^{-1}, almost 1.9 million additional tonnes of CO_2-e could feasibly be sequestered by providing incentives to landholders in the catchment to change their current land use.

Table 2.7: CO_2-e (t) that could feasibly be sequestered at different prices over a 61-year project.

Price cap ($t CO_2-e^{-1})	Discount rate					
	1%	3%	5%	7%	9%	11%
10	380,054	326,911	249,010	153,070	422,720	849,680
20	399,658	461,861	564,561	1,190,325	1,881,054	2,308,168
30	526,828	733,646	916,527	1,655,999	2,327,136	2,715,891
40	741,262	885,014	1,204,220	1,853,397	2,450,197	2,826,407
50	834,196	1,102,587	1,306,661	1,924,977	2,528,358	2,887,248
Unlimited	1,373,523	1,442,447	1,509,769	2,062,151	2,605,837	2,933,234

2.3.3 Transaction costs

Fitchner *et al.* (2003) found that transaction costs varied between 7% to more than 100% of the production costs of carbon sequestration in the studies they reviewed. In this chapter, transaction costs were estimated as a proportion of production costs. The impact of different transaction costs on the total quantity of carbon potentially sequestered is shown in Figure 2.8.

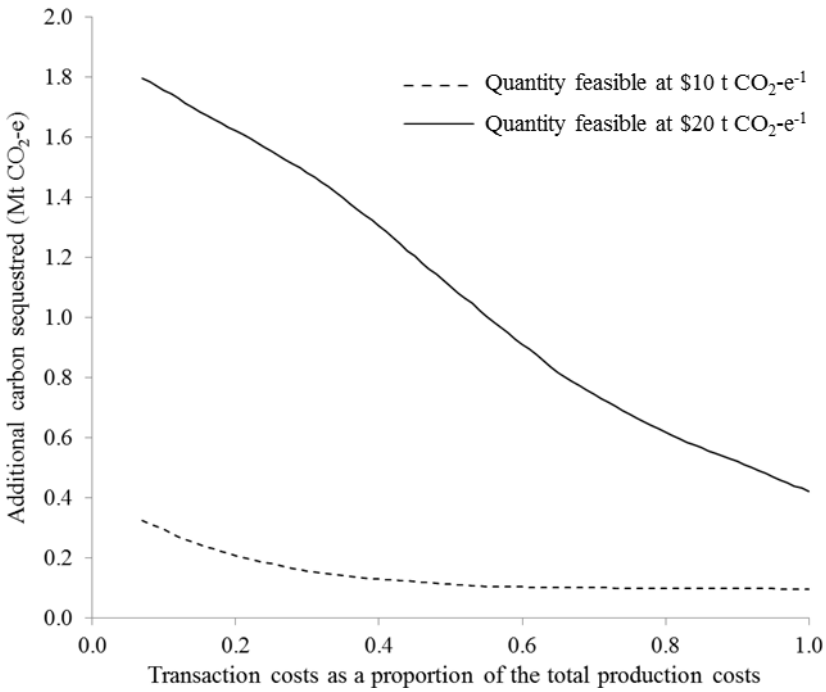

Figure 2.8: The feasible quantity of carbon sequestration (Mt CO_2-e) for different transaction costs, for a discount rate of 9%.

Assuming that a landholder will change land use if the sum of the opportunity costs and the transaction costs are less than the price of the carbon, the quantity of carbon that could be sequestered over a 61-year project will, not surprisingly, decline as transaction costs increase.

On the low end of the scale, if the transaction costs account for 7% of the production costs, at a \$20 t CO_2-e^{-1} price cap, and assuming a 9% discount rate, approximately 1.8 million tonnes of additional CO_2-e could be sequestered. If, however, these transaction costs are the same as the costs of production, only 422,000 tonnes of additional CO_2-e could be sequestered at the same price and discount rate. This represents a 76.5% decrease in possible project sequestration if transactions costs are at the higher end of the spectrum. It is therefore evident that exact quantification of transaction costs is of critical importance when determining the feasibility of land-use change projects.

2.3.4 Transportation costs

Transportation costs are a significant component of total costs involved with land-use changes that require the delivery of harvested biomass (Shi *et al.*, 2008). Previous studies have estimated that transportation costs account for between 18% and 29% of the total delivered cost of harvested biomass (Graham *et al.*, 1997). The transportation costs for the three proposed land-use options[h] that produce harvestable biomass were found to vary between 3.02% and 40.68% of the total project costs (Table 2.8).

Table 2.8: Transportation costs as a proportion of total project cost for land uses with harvested biomass.

Proposed land-use change	Mean	St. Dev.	Min	Max
P. radiata plantations	25.37%	6.80%	4.71%	40.68%
Generic eucalypt plantations	17.62%	4.95%	3.02%	28.01%
E. cladocalyx plantations	24.27%	6.34%	4.50%	37.10%

[h] Mixed-species environmental plantings do not produce harvestable biomass.

Transportation costs are a function of the cost per kilometre and distance between the source location and the destination processing mill. The Orthodromic distance technique is based on a simplifying assumption that the route is a straight-line 'as the crow flies'. A range of techniques to estimate distances using geographical and road data exist (Zhan & Noon, 1998), but have not been commonly used when determining the feasibility of projects with harvestable biomass (Healey *et al.*, 2009). As transportation costs are a significant component of total production costs, more sophisticated techniques that take account of geographical features are assessed in Chapter 3.

2.3.5 Sensitivity analysis

The feasibility of carbon sequestration projects estimated in this chapter is specific to the parameter values used in the FullCAM model and the input and output prices in the economic model (equations (2.2) – (2.5)). The default parameter values in FullCAM were generally used. Input and output prices were drawn from the literature. Sensitivity analysis was undertaken to evaluate the robustness of the results. Elasticities are presented in Table 2.9 for the percentage change in the total quantity of carbon that could be sequestered over a 61-year project at a price of \$20 t CO_2-e^{-1} given a percentage change in the parameter values assumed.

Table 2.9: Elasticities for the percentage change in the total carbon sequestered at \$20 t CO_2-e^{-1} for percentage changes in a range of parameters over a 61-year simulation period.

Parameter	Elasticity
Annual carbon sequestration (C_t)	1.17
Discount rate (δ_S)	1.13
Prices of harvested biomass[#]	0.63
Variable harvesting costs[#]	-0.07
Fixed planting costs[#]	-0.11
Revenue from current land use (r_{0t})	-0.39
Fixed management costs[#]	-0.59

[#]As per Table 2.2.

An elasticity of 1.17 for a change in the annual quantity of eligible carbon demonstrates the importance of having a spatially validated model to ensure accurate sequestration estimates. This elasticity also highlights that different management techniques, species selection and planting configurations, which all influence annual carbon sequestration over time (Hunt, 2008; Crossman *et al.*, 2011; Hodgman *et al.*, 2012; Paul *et al.*, 2013b), are important factors when estimating the feasibility of carbon sequestration projects. The impacts of these factors on project feasibility are explored in Chapter 5.

As already noted earlier in this chapter, the choice of discount rate can have a significant positive impact on the feasibility of carbon sequestration, and this is reflected here in the elasticity of 1.13. At higher discount rates, the total quantity of carbon that can be sequestered at a given price increases. This occurs because the timing of production costs is captured through discounting in the model, whereas the time preference for carbon sequestration is not. Given the urgency of reducing atmospheric carbon, removing a unit of carbon sooner is more important than removing it later. In a review of different carbon valuation equations, Boyland (2006) argued that failing to capture the time preference for carbon sequestration could give misleading results and concluded that discounting carbon sequestered over time is required to ensure meaningful economic results. The model used in this chapter is extended in Chapter 4 to include the discounting of carbon sequestered over time.

Assumptions about revenues and costs are also important when estimating feasible project size, with the price of harvested biomass and the fixed costs of managing plantations having the most impact, with elasticities of 0.63 and -0.59, respectively.

In addition to these elasticities, simulations of carbon sequestration potential under a range of future climate scenarios were undertaken in FullCAM for the different land-use

options considered in this chapter. The climate scenarios were derived from CSIRO and Bureau of Meteorology projections (Gordon *et al.*, 2010; Whetton *et al.*, n.d.). Changes in carbon sequestration across the entire catchment were negligible and are not shown here. This result was surprising and may reflect a possible deficiency in the FullCAM model.

2.3.6 Willingness to undertake land-use changes

The model used in this study assumes that landholders are economically rational and will switch land use if the financial benefit from the proposed land-use change is the same or greater than the opportunity cost of their current land use. The model does not take into account other motives, such as behavioural and attitudinal, which would contribute to a landholder's decision to undertake land-use change (Wilson, 1997; Langpap, 2004; Clayton, 2005; Kabii & Horwitz, 2006). The single profit maximisation objective used in this study could be extended to represent a utility function which includes several landholder objectives.

The model also calculates opportunity costs for 1.1 km^2 grid cells and landholders would be unlikely to base their land-use decisions at this level of resolution. Changes in land use would need to fit within their farming system. The model could be adapted to undertake further analysis on the degree of land-use change that would realistically be expected at a farm scale. This is done in Chapters 4 and 5.

Accounting for changes in soil carbon for climate mitigation has featured in academic discourse in recent years (Lal, 2004; Baker *et al.*, 2007; Lal, 2010; Aguilera *et al.*, 2013), however, simulations undertaken here showed minimal increases in soil carbon from land-use change. This is consistent with the findings of Smith *et al.* (2008) that showed that soils act as large carbon sinks, but their net flux is small. Carbon sequestration in

above-ground biomass as a result of land-use change shows more potential (Cacho *et al.*, 2008b; Kragt *et al.*, 2012; Mujuru *et al.*, 2014).

2.4 Conclusions

This study has found that additional carbon may be sequestered through land-use change in the Border Rivers-Gwydir catchment. Carbon sequestration potential varied significantly depending on location. Mixed-species plantings and *P. radiata* plantations had the highest carbon sequestration potential of all the land-use options assessed. However, they weren't necessarily the lowest-cost options and they did not dominate the lowest-cost strategies identified for the entire catchment.

No single land-use option defines the lowest-cost strategy for the entire catchment, which implies that an efficient strategy will need to allow for a combination of land-use changes across the catchment. Results were found to be most sensitive to the choice of discount rate and the annual carbon sequestration parameter. It was highlighted that failing to capture the time preference for carbon sequestration would also influence the results.

For land-use options with harvestable biomass, transportation costs were found to be a significant component of total costs. Transportation costs are a function of the distance from the source location to the destination processing mill. Here, transportation costs were estimated using a simple distance function. More complex functions and their impact on project feasibility are assessed in Chapter 3.

The inclusion of transaction costs incurred in changing land use and joining a carbon trading scheme will reduce the quantity of additional carbon that can be sequestered at a given carbon price. Results in this chapter have shown that the price can have a

significant impact on the feasibility of projects. Further quantitative analysis of the transaction costs incurred through participation in carbon markets will allow more accurate assessment of the potential supply of carbon mitigation products. The impact of transaction costs are assessed through simulations undertaken in Chapters 4 and 5.

CHAPTER 3: ESTIMATING OPTIMAL TRANSPORTATION COSTS FOR HARVESTED BIOMASS

3.1 Introduction

With the current trend towards finding low-cost options for sequestering carbon and the recent technological advances in the production of second-generation biofuels (Kumar *et al.*, 2009), the distance of travel between the biomass source and the processing destination is gaining increasing importance (Sims *et al.*, 2010). When comparing the optimal size and location of biomass processing plants, Searcy *et al.* (2007) demonstrated that travel distance was an important factor as it impacts the trucking costs associated with moving biomass to processing plants. Leduc *et al.* (2009) highlighted that one of the most important factors influencing the selection of locations for producing biomass for production into biofuels included areas where the travel distance was minimised. Transportation costs account for a significant percentage of the total cost of delivered biomass for biofuel production (Noon & Daly, 1996; Shi *et al.*, 2008), and have been estimated at between 18% and 29% (Graham *et al.*, 1997). Transportation costs were estimated to account for an average of between 18% and 25% (and up to 41%) of the total project costs for land-use changes with harvested biomass in Chapter 2.

While techniques to estimate distance between locations, along road networks exist, they have seldom been applied in studies of carbon offset projects. The total cost of biomass at the processing mill depends on distance from the source, how the biomass is transported, the type of material (such as bark, sawmill residue or whole-tree chips) and demand by competing markets (Wright *et al.*, 2008; Sims *et al.*, 2010; Fortenbery *et al.*, 2013). Therefore, in order to find the lowest-cost locations for establishing carbon mitigation projects, it is important to have an accurate measure of travel distance for estimating transportation cost.

Healey *et al.* (2009) noted that estimating transport of biomass is usually limited to tracking individual drivers and/or mill operators. They pointed out that using this type of method would be unacceptably onerous at the landscape level to determine average travel distances to mills, and is limited to estimating costs for existing projects. Therefore, in order to estimate travel distances and transportation costs to determine optimal locations for land-use change projects involving forestry or biomass outputs at a landscape scale, desktop modelling techniques are required. Several techniques for estimating shortest distance along road networks are reviewed by Zhan and Noon (1998).

In addition to reducing costs, it is important to fully account for all impacts of any greenhouse gas (GHG) mitigation scheme to ensure mitigation techniques incur a net reduction of emissions. The life-cycle assessment (LCA) technique accounts for the emissions associated with all stages of a product or program (Klöpffer, 1997). Geyer *et al.* (2010) pointed out that conventional LCA techniques do not utilise geospatial information. They noted, however, that there is potential to use geospatial modelling to provide more precise estimates of travel distances, transportation costs and greenhouse gas emissions.

Chapter 2 estimated the potential sequestration of carbon for different land-use options in a catchment in northern New South Wales, Australia. Several of the options examined in that chapter included the harvesting of biomass for both timber products and the production of second-generation biofuels from waste. The objective of this chapter is to introduce geospatial modelling to determine the transportation costs for potential carbon sequestration projects. The impact of different travel distance estimation methods for calculating transportation costs is determined to ascertain whether the increased accuracy of more complex methods justifies the additional data and processing time required.

Also investigated is the impact of using these different methods on net CO_2-e emission estimates.

3.2 Methods

3.2.1 Study area

The Border Rivers-Gwydir catchment is approximately 5,000,000 hectares (ha) and lies within the Murray-Darling Basin in northern New South Wales, Australia (Figure 3.1). The catchment is characterised by different climatic conditions and land types. Average annual rainfall ranges from 1,200 mm in the east to 450 mm in the west (Australian Bureau of Meterology, 2010). Varied topography exists with elevation ranging from 1,450 metres above sea level (MASL) on the eastern boundary down to about 150 MASL in the western plains and an average elevation of 499 MASL. Soils in the catchment are predominately of low to moderate fertility (Scott *et al.*, 2004) and include grey, black and brown vertosols, red, brown, grey and yellow chromosols, sandy tenosols, kandosols, red ferrosols and dermosols (Isbell, 2002; DCCW, 2010).

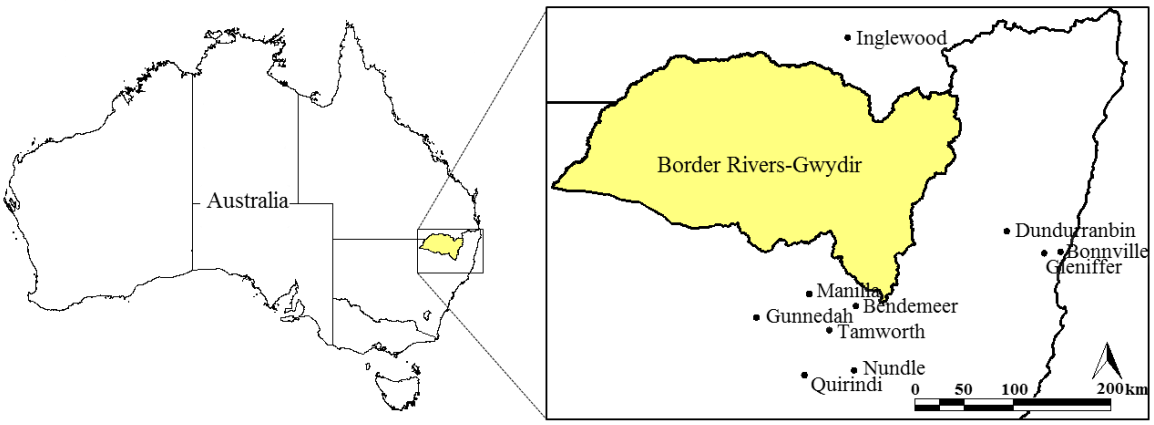

Figure 3.1: Location of the case study region and existing timber processing mills.

Major land uses in the region are grazing of modified and native pastures and dryland cropping (BRS, 2009). Grazing enterprises dominate the tablelands in the eastern sections of the catchment, while cropping predominately occurs on the slopes and plains

in the western half of the catchment (NSW Government, 2009a). The catchment is represented by a raster grid of 42,046 cells of 1.1 km^2 as described in section 2.2.3.1.

3.2.2 Proposed land-use changes

Four land-use changes with the potential to sequester additional carbon are investigated. These include the establishment of (1) *Pinus radiata* plantations for commercial softwood production; (2) a generic eucalypt species plantation for commercial hardwood production, pulpwood and as a biofuel source; (3) *Eucalyptus cladocalyx* plantations for commercial hardwood production; and (4) a mixed-species environmental planting with no commercial value, but potential to provide income from offset schemes such as the Carbon Farming Initiative.

Carbon sequestration potential across the case study catchment, for each of these proposed land uses, was estimated with the FullCAM model (version 3.13.8 – research edition), which was developed by the Australian Greenhouse Office (Richards, 2001) and has been calibrated for Australian regions and land-use types (Paul *et al.*, 2008; Polglase *et al.*, 2008). Carbon sequestration from current land uses were also estimated using the FullCAM model. This allowed for the quantity of eligible carbon (C_t) for a given year to be calculated by (Cacho *et al.*, 2008b):

$$C_t = C_{P,t} - C_{C,t} \tag{3.1}$$

where $C_{P,t}$ and $C_{C,t}$ are the carbon stocks of the proposed land use and the current land use in year t, respectively.

The optimal locations across the catchment for land-use change, based on both optimal sequestration potential and lowest opportunity cost, are determined using the model and parameter values described in Chapter 2.

Additional CO_2-e emissions from transportation of harvested biomass materials are included in the analysis by utilising an emissions factor of 214 g CO_2-e t^{-1} km^{-1} (Lindholm & Berg, 2005; Healey *et al.*, 2009).

3.2.3 Travel distances and transportation costs

Three of the four proposed land uses, *P. radiata* plantations, generic eucalypt species plantations and *E. cladocalyx* plantations, have a cost associated with transporting biomass from the source location to a processing mill. There are currently 10 operational mills within proximity of the case study catchment (Figure 3.1). None of these mills are actually located within the catchment. This chapter determines the optimal mill to which to transport biomass. It does not investigate the optimal position for possible new mills or biofuel processing sites, which has been considered by other authors (for example, Dunnett *et al.*, 2008; Schmidt *et al.*, 2009; Kim *et al.*, 2011).

Employing the same methodology as Polglase *et al.* (2008), \$0.13 m^{-3} km^{-1} was assumed to be the fixed cost per kilometre for travel. Transportation costs are calculated by multiplying this fixed cost by the number of kilometres between the biomass source location and the destination mill. The shortest distance from each of the 42,046 1.1 km^2 cells in the catchment to each of the 10 mills was calculated using three different travel distance estimation methods. The first method calculates the shortest distance 'as the crow flies' from each cell to each mill using the Orthodromic algorithm. The second method, called the Dijkstra algorithm (Dijkstra, 1959), calculates the shortest distance between each cell and mill, along a road network. The third method is called the least-cost algorithm and is based on the Dijkstra algorithm. It accounts for differences in travel speed and fuel consumption along different road classes, formations and gradients (Rees, 2004; Choi & Nieto, 2011) and determines the shortest route while

simultaneously accounting for both travel time and fuel consumption. These three methods are now discussed in further detail.

3.2.3.1 Orthodromic algorithm

The Orthodromic algorithm estimates the shortest distance between geographical coordinates. For each 1.1 km^2 cell in the case study catchment, the distance to each mill is determined using the Haversine formula:

$$d_{m:n} = 111.12 \cdot 2 \arcsin\left(\sqrt{\sin^2\left(\frac{\varphi_m - \varphi_n}{2}\right) + \cos\varphi_m \cos\varphi_n \sin^2\left(\frac{\lambda_m - \lambda_n}{2}\right)}\right) \tag{3.2}$$

where d is the distance between the source location m and the location of the mill n, in kilometres; 111.12 is a conversion factor to convert degrees to kilometres; φ_m and φ_n are the latitudes in radians of m and n, respectively; and λ_m and λ_n are the longitudes in radians of m and n, respectively.

The optimal mill for each source location is selected by minimising the travel distance to the mill. This method calculates the travel route 'as the crow flies', so provides an inaccurate estimation of the likely distance between locations. However, it has the advantage of requiring no information on the road network. This method was used in Chapter 2.

3.2.3.2 Dijkstra algorithm

The calculation of minimum travel distance using the Dijkstra algorithm allows the shortest route from each source location to each mill to be estimated along a road network. The optimal mill for each source location is the mill to which the travel distance is the shortest. The algorithm proposed by Dijkstra (1959) has been applied and

assessed in minimum distance calculation studies by numerous authors (Dreyfus, 1969; Zhan & Noon, 1998; Nannicini *et al.*, 2010).

For the purpose of the Dijkstra algorithm, a road network is divided into nodes (intersections) and branches (road sections). The shortest path between two locations is solved by an iterative process described by Dijkstra (1959) and summarised below.

To initialise the algorithm, all nodes are subdivided into three sets:

i. nodes for which the minimum length from the starting node is known;
ii. nodes from which the next node to be added to *set i* will be selected; and
iii. all remaining nodes.

The branches are also subdivided into three sets:

a. branches which form the minimum path from the starting node to nodes in set i;
b. branches from which the next branch to be moved to *set a* will be selected; and
c. branches that have either not been considered or have been rejected.

As the distance of the origin node is known, this node is labelled with a distance of zero and is the only node placed in *set i*. All branches are placed in *set c*. The following iterative steps are undertaken repeatedly until the destination node is solved and placed in *set i*:

Step 1:

Find all the unsolved nodes that are directly connected by a single branch to any node in *set i*. These nodes are placed in *set ii* and their corresponding branches moved to *set b*. For each of these unsolved nodes, determine the candidate distance (*d*) as follows.

$$d_{a:n} = d_{a:n-1} + d_{n-1:n} \tag{3.3}$$

where d is distance, subscript a is the starting node and n—1 is a solved node directly connected and immediately preceding the unsolved node n.

Step 2:

Select the node with the smallest distance and move this node to *set i*; the minimum distance to this node is now solved and the node is labelled accordingly with this distance. The corresponding branch is now moved from *set b* to *set a*.

Step 3:

If the node moved to *set i* in Step 2 is not the destination node, return to Step 1. Otherwise, the optimal solution has been found and is the distance of the last node moved to *set i.* The shortest route can also be recovered by tracing backwards from the destination node to the starting node.

This algorithm requires data on the road network, which was obtained from several shapefiles from the GeoScience Australia database (Geoscience Australia, 2012). A list of these shapefiles is included in Appendix A. Distance of each connected road section was determined using the Euclidean distance:

$$d_j = 111.12 \cdot \sum_{i=1}^{N-1} d_{i:i+1} \tag{3.4}$$

where d_j is the distance of the road section between nodes with coordinates q and r, in kilometres; $d_{i:i+1}$ is the 'Orthodromic distance', calculated with equation (3.2), between the nodes; and i represents the coordinates q and r.

The application of the Dijkstra algorithm to solve the shortest route from a source location to a destination mill across a simplified road network is provided in Appendix B.

As the catchment is represented by 1.1 km^2 cells, rather than actual farms, not all start locations are adjacent to a road. Therefore, an additional 'starting distance' algorithm was used to find the nearest adjacent road for each source location. In general terms, this 'starting distance' algorithm takes the following steps to find the distance from the starting location to the nearest adjacent road.

Initialisation:

To initialise the algorithm, all cells are subdivided into three sets:

I. cells which have been assessed and for which no roads intersect;

II. set containing cells from which the next iteration will assess whether any road intersects;

III. set containing all cells not yet assessed.

Step 1:

Assess whether the starting location cell is intersected by the road network. If so, the minimum start distance to the road network is zero. Otherwise, place the starting location cell in *set I* and select all cells adjacent to this starting location cell and place them in a *set II* and continue to Step 2.

Step 2:

Starting at the cell in the north-eastern corner and working in a clockwise fashion, assess all cells in the adjacent bundle, to determine whether any of the cells are intersected by part of the road network. Once a cell with an intersecting road is found, stop and use equation (3.2) to determine the distance from the starting location to this cell. This location becomes the starting node for the Dijkstra algorithm and the distance calculated using equation (3.2) is added to the optimal distance estimated with the Dijkstra algorithm. If no cells in the adjacent bundle are found, find all adjacent cells to the cells

currently in *set II* from *set III* and move to *set II* while simultaneously moving the cells currently in *set II* to *set I.*

3.2.3.3 Least-cost algorithm

This travel distance estimation method builds on the Dijkstra algorithm, by including several physical characteristics of the road that may be critical for determining the optimal route. Several authors have shown that road class and formation are important factors when calculating shortest feasible routes (for example, Dubuc, 2007; Healey *et al.*, 2009). These authors have applied weights or penalties to the Dijkstra algorithm to account for different road characteristics. This allows for optimal routes to be determined based on a cost penalty as well as distance. In this chapter, equation (3.5) uses cost penalties from Healey *et al.* (2009) that reflect the impact of road class and formation on travel:

$$\delta_{ij} = \psi_i \cdot d_j \tag{3.5}$$

where δ_{ij} is the cost penalty of travelling along road class i for road section j, ψ_i is the penalty assigned to road class i (Table 3.1), and d_j is the length of road section j.

GIS shapefiles containing road network data, including class and formation, across the catchment were obtained from Geoscience Australia (2012).

Table 3.1: Penalties and speed assumptions used for the least-cost algorithm.

	Class penalty (ψ)		**Speed assumption (km h^{-1})**	
	Road formation		**Road formation**	
Road class	**Sealed**	**Unsealed**	**Sealed**	**Unsealed**
Dual carriageway	1.0	NA	110	NA
Minor road	7.0	8.4	60	48
Principal road	3.0	3.6	100	80
Secondary road	5.0	6.0	80	64
Track	12.0	14.4	30	24

As pointed out by Dubuc (2007), considering topography when assessing the time and cost of travel is uncommon. Boriboonsomsin and Barth (2009) undertook a study on the consumption of fuel by trucks, relative to the road gradient and the speed of travel. Road gradient is also included in the least-cost algorithm as a cost penalty, using topographical data at a 1:250,000 scale across the study region (Figure 3.2). This data was obtained from Geoscience Australia (2012).

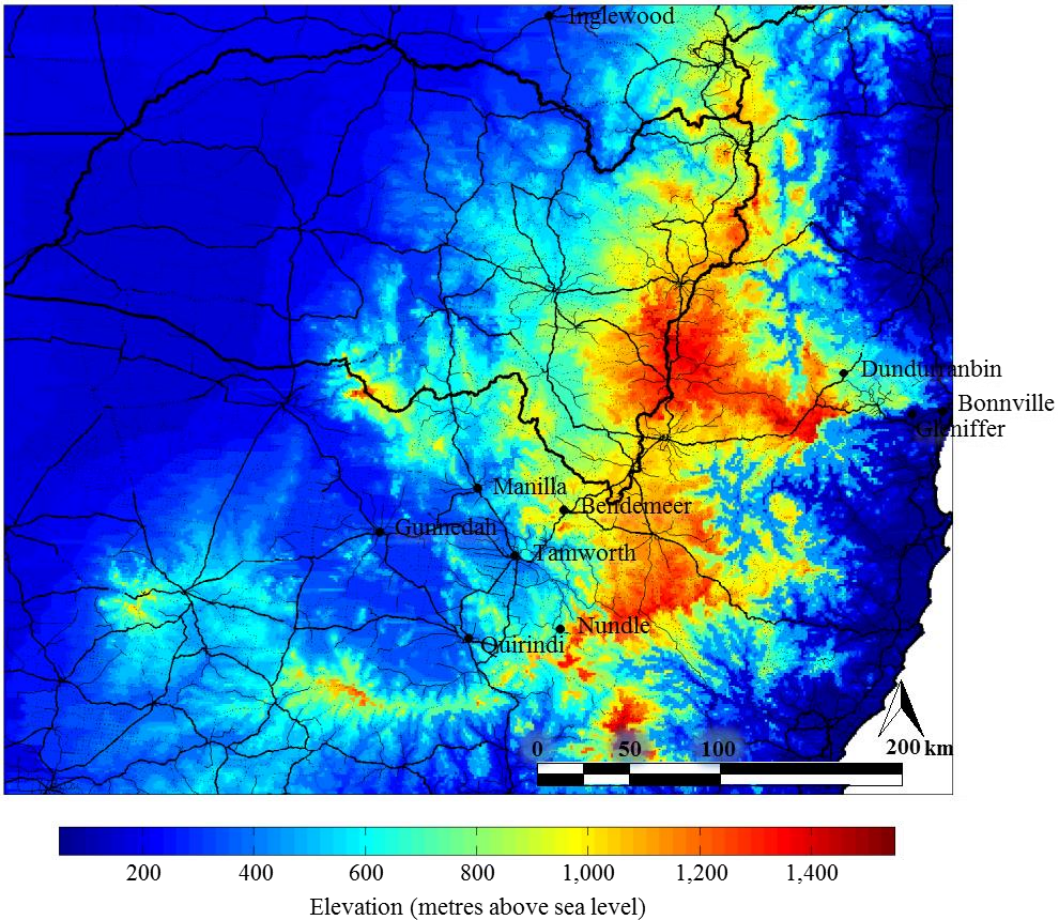

Figure 3.2: Digital elevation map of surrounding region.

The road gradient cost penalty was included based on the additional petrol used when travelling on slopes, relative to flat ground. Quadratic regressions in the form of equation (3.6) were undertaken on the data provided in Boriboonsomsin and Barth (2009) to create functions for the additional quantity of petrol used on the different road classes when the effect of gradient was introduced.

$$y = \beta_0 + \beta_1 X + \beta_2 X^2 \tag{3.6}$$

where y is the proportion of fuel used relative to travelling on a flat road and X is the road gradient. A very good fit was achieved across all travel speeds, with all regressions having an R^2 of at least 0.9983 (Table 3.2). The additional fuel required for different road gradients is shown in Figure 3.3.

Table 3.2: Coefficients for determining the proportion of fuel used on different road classes in relation to road gradient.

Road Class (*i*)	β_0	β_1	β_2	R^2
Dual carriageway	0.9976	0.1411	0.0066	0.9983
Minor road	1.0320	0.1984	0.0096	0.9995
Principal road	0.9836	0.1639	0.0080	0.9989
Secondary road	0.9838	0.1946	0.0092	0.9989
Track	1.0307	0.1404	0.0068	0.9985

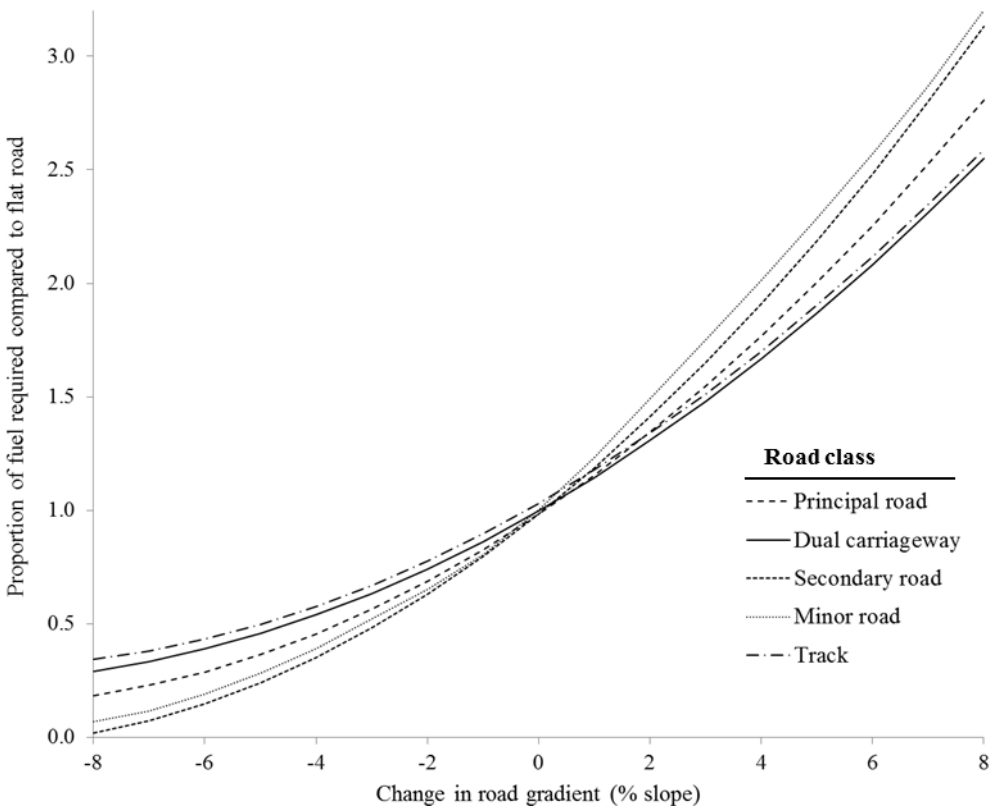

Figure 3.3: Fuel required for different road gradients across different road classes, relative to a flat road.

The cost penalties associated with each road class and formation (equation (3.5)) were further modified to include the influence of road gradient. Each section of road across the region was assigned a cost penalty according to the following equation:

$$\delta_{ij} = \psi_i \left(\beta_{i0} + \beta_{i1} X_j + \beta_{i2} X_j^2\right) \cdot d_j \tag{3.7}$$

where δ_{ij} is the cost penalty of travelling along road class i for road section j, ψ_i is the penalty assigned to road class i (Table 3.1), X_j is the gradient of road section j, β_{in} are the coefficients for the function to determine proportion of fuel consumption for road class i (Table 3.2) and d_j is the length of road section j.

The cost penalties that are estimated for each road section replace the lengths of each road section in the Dijkstra algorithm and the algorithm is executed using the iterative process described in section 3.2.3.2. Once the least-cost route is determined, the distance of this route is estimated. The optimal mill is the mill with the least-cost route from the source location and is not necessarily the mill to which the distance is shortest.

As with the Dijkstra algorithm, not all locations used as a starting point adjoin a road network. Therefore, it was necessary to estimate the distance to the nearest road using the 'starting distance' algorithm.

3.3 Results

3.3.1 Differences between methods

Distance from each of the 42,046 1.1 km^2 cells in the catchment to each of the 10 mills was estimated using the three travel distance estimation methods just described. The optimal mill from the source location for the Orthodromic and Dijkstra algorithms is the mill for which the distance is shortest. For the least-cost algorithm, it is the mill for which the route is of least cost and is not necessarily of shortest distance. The distances to the optimal mill estimated using the three algorithms are shown in Figure 3.4.

The Orthodromic distance method estimated the shortest average distance to the optimal mill. The average distance to the optimal mill using this method was 122 km. This compares to average distances to the optimal mill of 160 km and 173 km with the Dijkstra and the least-cost algorithms, respectively. These results are not surprising. The Orthodromic algorithm assumes a straight-line along the curvature of the earth and ignores any transportation constraints imposed by physical characteristics such as the requirement to travel along a road network and would be expected to produce the shortest route. The Dijkstra algorithm improves the estimation by constraining the optimal route to the road network, but it still ignores important physical characteristics such as road class and gradient. The least-cost algorithm would be expected to produce the longest optimal route because it more realistically accounts for travel characteristics including travel time and fuel consumption.

Statistical differences between the results of the three algorithms are shown in Table 3.3. The distance between cells and their optimal mill is 7.60% longer on average (up to a maximum of 42.34% longer) using the least-cost algorithm rather than the Dijkstra algorithm. It is ever longer when using the Orthodromic algorithm (41.32% longer on average, up to a maximum of 184.30% longer).

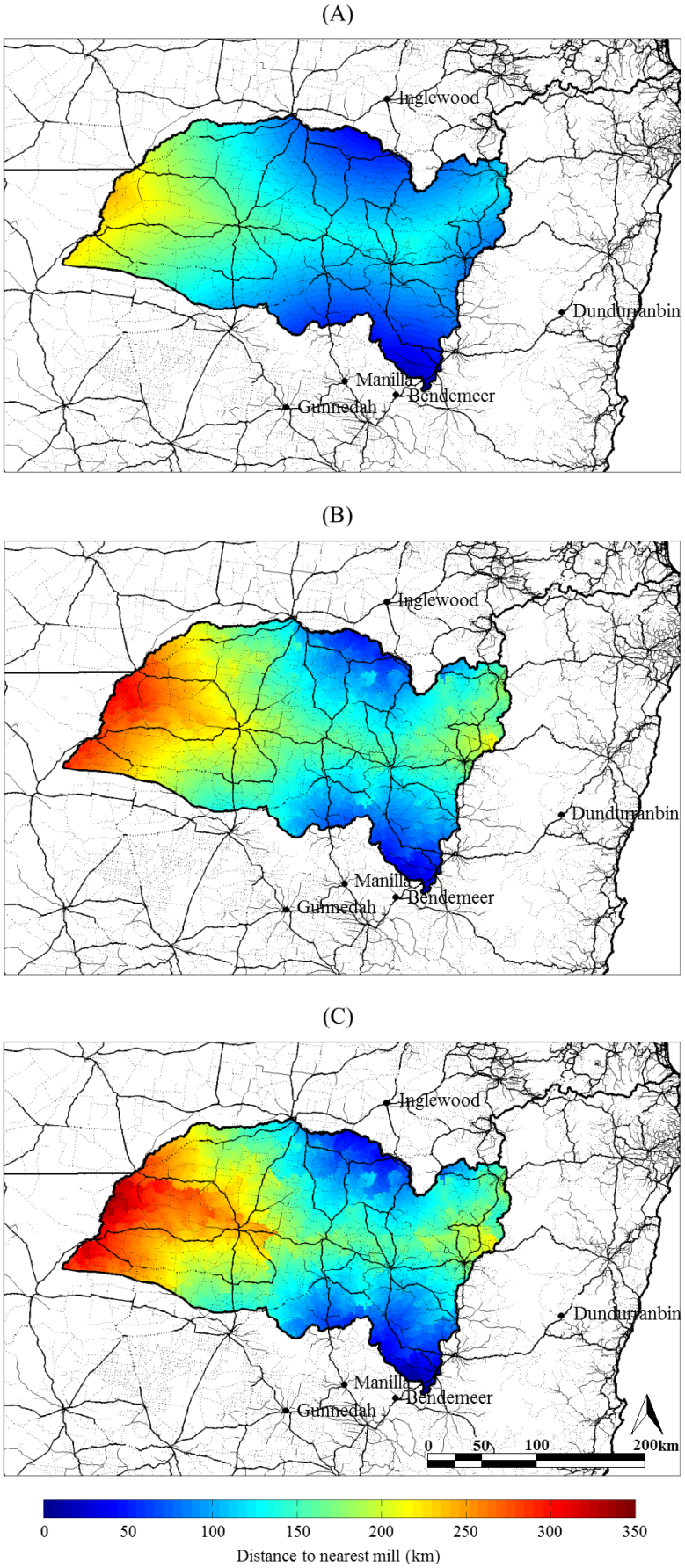

Figure 3.4: Distance to the optimal mill using the (A) Orthodromic algorithm; (B) Dijkstra algorithm; and (C) least-cost algorithm.

Table 3.3: Statistical differences for the three travel distance estimation methods.

Difference between methods*	Mean	St dev	Min	Max
Δ LC:DA	7.60%	6.05%	-13.34%	42.34%
Δ LC:OD	41.32%	16.40%	-2.42%	184.30%
Δ DA:OD	31.51%	15.27%	-5.43%	171.36%

*** Where Δ LC:DA is the percentage difference between optimal distances using the least-cost algorithm compared with the Dijkstra algorithm; Δ LC:OD is the percentage difference between optimal distances using the least-cost algorithm compared with the Orthodromic algorithm; and Δ DA:OD is the percentage difference between optimal distances using the Dijkstra algorithm compared with the Orthodromic algorithm.**

3.3.2 Optimal mills

As already highlighted, the optimal mill from the source location for the Orthodromic and Dijkstra algorithms is the mill to which the distance is shortest. For the least-cost algorithm, it is the mill to which the route is of least cost. The optimal mills for each location in the catchment and for each of the three methods are shown in Figure 3.5. For all three algorithms, only five of the 10 mills were found to be optimal locations to which to transport harvested products. These mills are located at Bendemeer, Dundurranbin, Gunnedah, Inglewood and Manilla. The optimal mill varies depending on the algorithm. This is particularly evident in the eastern quarter of the catchment where the inclusion of topography alters the choice of optimal mill in the least-cost case.

In order to calculate the transportation costs to the optimal mill for the different methods (i.e. the \$0.13 m^{-3} km^{-1} fixed cost of travel from section 3.2.3 multiplied by the distance of the optimal route), the total amount of biomass transported between sources and mills needs to be introduced. In the absence of actual data, it is assumed that in an 'extreme case' all cells in the catchment are converted to forest. The percentage of total harvested biomass for the catchment that would be transported to each of the optimal mills for this 'extreme case' for the three algorithms is shown in Table 3.4.

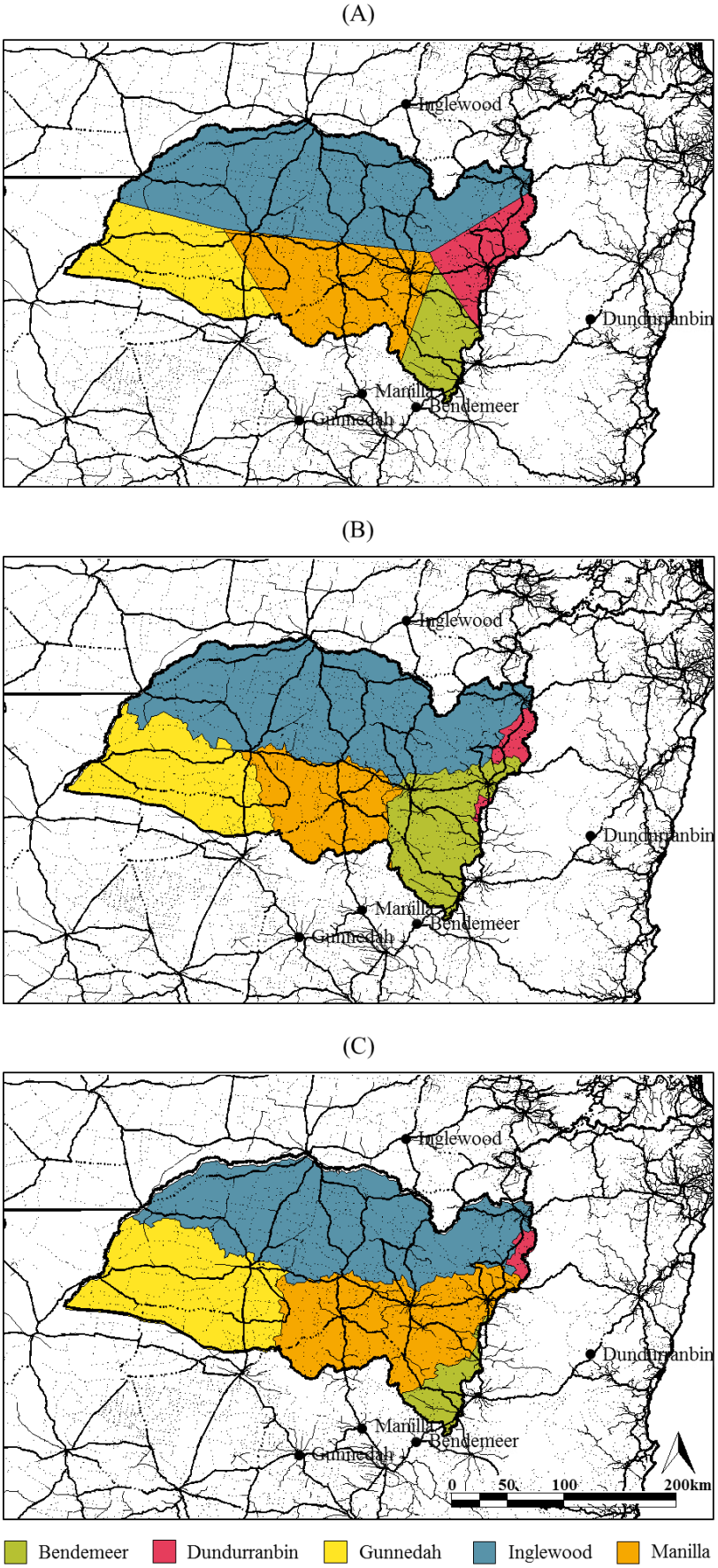

Figure 3.5: Optimal mill determined by shortest distance using the (A) Orthodromic algorithm; (B) Dijkstra algorithm; and (C) least-cost algorithm.

Table 3.4: Percentage of total harvested biomass shipped to optimal mills for the 'extreme case' using the three travel distance estimation methods.

	Optimal mill					
Estimation method	**Bendemeer**	**Dundurranbin**	**Gunnedah**	**Inglewood**	**Manilla**	**Total**
Orthodromic algorithm	9.44	12.70	14.64	41.52	21.70	100
Dijkstra algorithm	21.66	4.02	16.03	42.92	15.37	100
Least-cost algorithm	5.44	1.74	21.31	38.80	32.71	100

There are significant differences between the results presented for the three algorithms in Table 3.4. For example, the Bendemeer mill would each receive 21.66% of the total harvested biomass from the catchment for the Dijkstra algorithm, but only 5.44% for the least-cost algorithm. This occurs because the roads that lead from the catchment to this mill extend across areas of pronounced topographical variance, which is taken into account with the least-cost algorithm.

A more 'realistic case' would be for the cheapest 10% of locations, in terms of opportunity costs as estimated in Chapter 2, to be converted to forest. These locations all have an opportunity cost lower than \$18 t CO_2-e^{-1}. They would be converted to forest if incentive payments were higher than this opportunity cost and were sufficient to cover transaction costs and required rents. Assume that a policy is introduced to encourage these landholders to enter into a carbon sequestration project at the same time and they were to harvest at the same time. Results for the total harvested biomass that would be transported from these locations to the optimal mills are presented in Table 3.5. It is evident that the travel distance estimation method has a significant impact on the biomass that a mill could expect. When road gradient, class and formation characteristics are ignored as in the Orthodromic and Dijkstra methods, the Bendemeer mill would receive 236,815 m^3 and 29,328 m^3 of biomass, respectively. This quantity would drop to just over 32,082 m^3 when the physical road characteristics are taken into account (least-cost method).

Table 3.5: Total biomass (m^3) shipped to optimal mills for the 'realistic case' using the three travel distance estimation methods.

	Optimal mill				
Estimation method	**Bendemeer**	**Dundurranbin**	**Gunnedah**	**Inglewood**	**Manilla**
Orthodromic algorithm	29,328	165,587	366	163,191	25,620
Dijkstra algorithm	236,815	59,271	647	223,252	26,735
Least-cost algorithm	32,082	24,213	1,893	240,033	275,052

The methods used to select optimal mills in this chapter do not take into account the processing capacity of the mills. If the biomass shipped to a particular mill exceeds the processing capacity of that mill, then the biomass will need to be shipped elsewhere and additional transportation costs incurred or the processing capacity of the mill will need to be increased. The travel distance estimation methods considered here could be combined with a model developed to address the classical 'capacitated facility location problem' (Nauss, 1978; Melkote & Daskin, 2001). This would allow for the feasibility of increasing the operational capacity of mills to be determined.

3.3.3 Differences in total abatement costs

The total abatement costs of four land-use options for carbon sequestration projects were determined for each travel distance estimation method using the approach and assumptions described in Chapter 2. These land uses were the (1) planting of *P. radiata* for commercial softwood production; (2) planting of a generic eucalypt species for commercial hardwood production, use as a pulpwood and biofuel source; (3) planting *E. cladocalyx* for commercial hardwood production; and (4) planting a native species mix for environmental purposes with no commercial value. The cost of planting native species for environmental purposes is not affected by transportation costs as there are no harvestable products associated with this land use, so no results are presented for this land-use option.

The impacts of each travel distance estimation method on the total costs of the three remaining land-use options are presented in Table 3.6 and Figure 3.6.

Table 3.6: Percentage increase in total costs for each land-use option with harvestable biomass using the three travel distance estimation methods.

	Estimation method		
Proposed land use	**Δ LC:DA***	**Δ LC:OD***	**Δ DA:OD***
P. radiata plantations	2.17	9.77	7.42
Generic eucalypt plantations	1.55	6.78	5.14
E. cladocalyx plantations	2.10	9.33	7.07

*** Where Δ LC:DA, Δ LC:OD and Δ DA:OD are as defined in Table 3.3.**

An average increase in total costs of 2.17%, 1.55% and 2.10% for land-use changes to *P. radiata*, generic eucalypt plantations and *E. cladocalyx* plantations, respectively, occurs when the least-cost algorithm is compared with the Dijkstra algorithm.

While the average increases in total costs may appear small, the ranges shown in Figure 3.6 demonstrate that for some locations the total costs increase by almost 16% when the least-cost algorithm is used rather than the Dijkstra algorithm. The increase in total costs when the least-cost algorithm is used rather than the Orthodromic algorithm is more significant. For example, a land-use change to a *P. radiata* plantation has an average increase in total costs of 9.77% when the travel distance is estimated using the least-cost algorithm instead of the Orthodromic algorithm, up to a maximum increase of 55.91% (not shown).

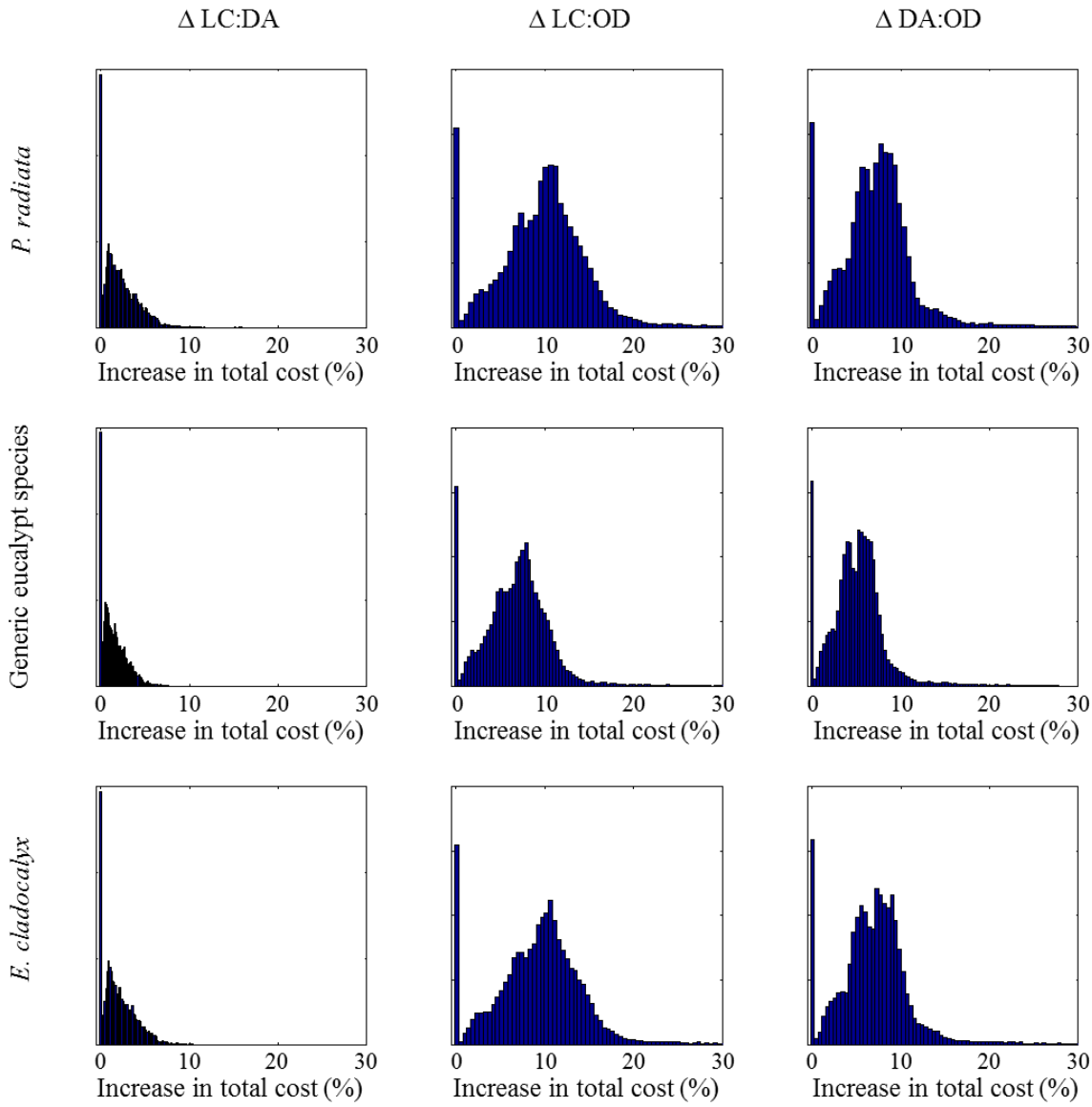

Figure 3.6: Distribution of percentage change in total costs for each land-use option with harvestable biomass using the three travel distance estimation methods.

3.3.4 Additional transport emissions

The impact of the three travel distance estimation methods on the emissions from the trucks used for transporting biomass from source location to mills was also considered. An emissions factor of 214 g CO_2-e t^{-1} km^{-1} (Lindholm & Berg, 2005; Healey *et al.*, 2009) was used to convert biomass and estimated distance to nearest mill into transportation emissions. Figure 3.7 illustrates the increase in CO_2-e emissions derived

for the three travel distance estimation methods[i]. Average emissions using the least-cost algorithm were 7.65% higher than when using the Dijkstra algorithm and 41.29% higher than when using the Orthodromic algorithm. Average emissions using the Dijkstra algorithm were 31.41% higher than when using the Orthodromic algorithm. On average, if the Dijkstra algorithm is used transportation emissions are underestimated by 0.23 t CO_2-e ha^{-1} annually. This further highlights the importance of capturing information about road class, formation and gradient in the form of a cost penalty as in the least-cost algorithm.

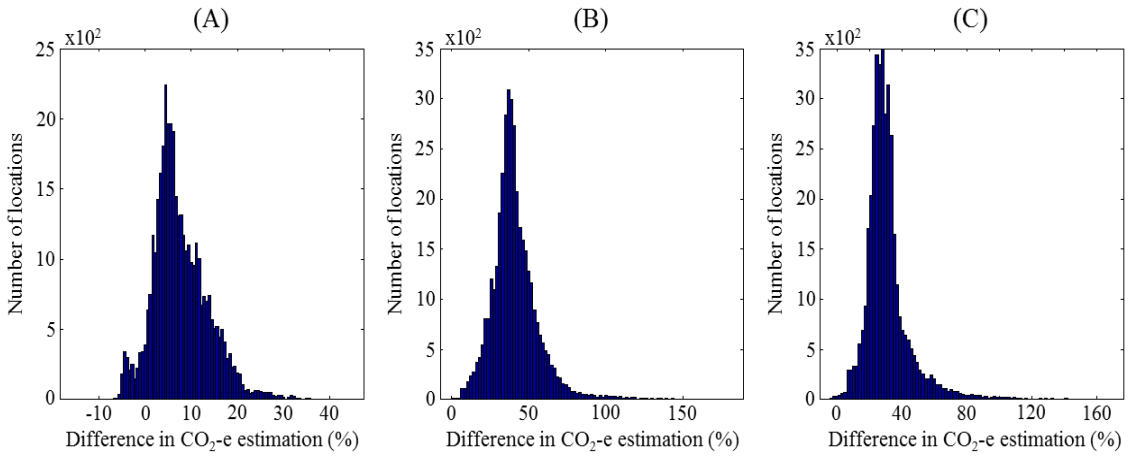

Figure 3.7: Percentage increase in CO_2-e emissions from using the (A) least-cost algorithm rather than the Dijkstra algorithm; (B) least-cost algorithm rather than the Orthodromic algorithm; and (C) Dijkstra algorithm rather than the Orthodromic algorithm.

While annual transport emissions underestimated seem small (0.23 t CO_2-e ha^{-1}), if the locations that comprise the cheapest 10% in terms of opportunity costs were converted to forest, using the Dijkstra algorithm rather than the least-cost method would result in CO_2-e emissions from trucks transporting biomass to mills being underestimated by just over 240 t CO_2-e annually.

[i] This analysis implicitly assumes that any trucks used to transport harvested biomass in the study region would otherwise not have been utilised and would therefore produce no emissions.

3.4 Discussion and conclusions

This study has evaluated three travel distance estimation methods of differing complexity as part of a model to assess incentives for biomass production and carbon mitigation in a catchment in northern New South Wales. Given that transportation costs are often a driving factor in the feasibility of forestry production, especially with biomass transported for the production of biofuels (Hellmann & Verburg, 2011), the study highlights the importance of selecting an appropriate travel distance estimation method.

To implement the Orthodromic algorithm for an entire catchment requires minimal data and can be run on a desktop computer in a matter of seconds, while the Dijkstra and least-cost algorithms are both data intensive and in this study had an execution time of approximately 28 days each to determine the optimal distance, route and mill[j]. Despite the ease with which the Orthodromic algorithm can be implemented, the findings suggest that caution should be exercised if this method is used to determine the feasibility of carbon mitigation projects that require the transportation of forestry products. The travel distances and transportation costs estimated with this method are substantially lower than those estimated with the other methods, and could result in trucks being sent to sub-optimal mills, costing more and resulting in additional carbon emissions.

The least-cost algorithm provides the most accurate estimation of travel distance as it accounts for several of the physical constraints that would be encountered when selecting an optimal route. The results suggest that favouring a method which requires less data will underestimate the cost of landholder participation which may over inflate the expected feasibility of the policy. When compared to the least-cost algorithm, projects assessed using the Orthodromic algorithm underestimated the total costs by an average

[j] Simulations were undertaken on a quad-core workstation with 2.93 gigahertz processors, 32 gigabytes of installed memory and a 64-bit operating system.

of 9.77%, 6.78% and 9.33% for *P. radiata* plantations, generic eucalypt plantations and *E. cladocalyx* plantations, respectively.

The findings support the assertion by Dubuc (2007) that the use of geographical data needs to become more common place when assessing transportation costs. Despite the additional data required, the execution time for the least-cost algorithm is comparable to that of the Dijkstra algorithm. The adoption of the least-cost algorithm is therefore recommended.

The strategic placement of biofuel processing plants is gaining considerable attention in the literature in order to reduce transportation costs (Dunnett *et al.*, 2008; Schmidt *et al.*, 2009; Hellmann & Verburg, 2011; Kim *et al.*, 2011). To ensure the efficient placement of new biofuel processing plants, the travel cost algorithm used to estimate the optimal mill needs to be accurate. The only algorithm considered here that accounts for road class, formation and gradient is the least-cost method. Information derived using this algorithm could also help government to make decisions about where to invest in road infrastructure.

As the purpose of the land-use options considered in this chapter is for climate mitigation, it is important to consider the GHG emissions from the transport of forestry and biomass products (Cherubini *et al.*, 2009). It was found that if a simple algorithm which lacks roads and topographical features is used to estimate travel distance, the CO_2-e emissions from transportation would be grossly underestimated. Features of the road network and of the topography need to be considered when determining optimal travel distances to ensure that an accurate estimation of the carbon balance is achieved.

There are two limitations to this study. First, less than 2% of the catchment is currently under forestry (BRS, 2009), so insufficient data is available to validate the modelled

travel distances against actual individual tracked routes from trucks hauling forestry products across the catchment using the method described by Healey *et al.* (2009). Second, the analysis does not consider the feasibility of establishing a new mill within the catchment and this offers an opportunity for future research. The mills considered are existing mills, which are located outside the catchment. No mills currently exist within the catchment. The strategic establishment of a new mill within the catchment could influence the feasibility of undertaking land-use changes for climate mitigation.

CHAPTER 4: FARM-SCALE ANALYSIS OF THE POTENTIAL UPTAKE OF CARBON OFFSET ACTIVITIES

4.1 Introduction

It is expected that Australia's domestic emissions will be well above its national emission target without the introduction of further policies and incentives to offset emissions from current activities or reduce greenhouse gas (GHG) emitting activities and enterprises (Jotzo, 2012b). Agricultural landholders have the potential to contribute to these emissions targets in several ways: (a) avoidance of land clearing activities (Henry *et al.*, 2002; Skutsch *et al.*, 2007); (b) provision of carbon offsets through carbon sequestration in biomass and soils (Turner *et al.*, 2005; Eady *et al.*, 2009; Paul *et al.*, 2013b); (c) reduction in GHG emitting inputs (Meisterling *et al.*, 2009; Mohammadi *et al.*, 2013); (d) reduction of methane emissions (Hongmin *et al.*, 1996; Shin *et al.*, 1996); and (e) production of biofuels to displace fossil fuels (Fung *et al.*, 2002; Sims *et al.*, 2010).

Kember *et al.* (2013) noted that there is a risk that Australia will rely too heavily on the importation of international permits and thus suggested that domestic mitigation policies need strengthening. Of all the domestic policy options, the sequestration of carbon in forestry biomass has been ranked as the simplest and most cost-effective to implement (Eady *et al.*, 2009). In Australia, previous studies have focused on the forest sequestration potential at the national, state or broad catchment level (Kirschbaum, 2000; Polglase *et al.*, 2008; Eady *et al.*, 2009). Such broad-scale analysis does not allow for local heterogeneity that exists across a region, or even across an individual farm, to be determined. Studies have shown that significant variations occur in farm productivity, management, input and output levels and carbon sequestration potential, even across local regions (Tschakert, 2004; Kwon *et al.*, 2006; Paul *et al.*, 2013b). Hence, broad

region estimations of costs associated with climate mitigation may not only be inadequate but may provide misleading information to decision makers (Tschakert, 2004). Paul *et al.* (2013b) found that the failure to account for local variations in site quality, management and different planting configurations can influence the estimates of sequestration potential by between 15% and 53%.

Planting trees for carbon sequestration purposes may produce additional benefits, known as co-benefits, including enhanced biodiversity conservation, salinity reduction and improved soil and water quality (Plantinga & Wu, 2003; Harper *et al.*, 2007; Shaikh *et al.*, 2007; Mattsson *et al.*, 2009; Townsend *et al.*, 2012). However, paddocks planted to timber plantations for carbon sequestration can potentially impose both positive and negative externalities on adjoining paddocks. In a socio-economic study of returns from farm forestry and agriculture in south-east Australia, Stewart *et al.* (2011) found that an 'edge effect' from timber belts caused a loss in pastures from adjoining paddocks immediately along the paddock edge. This was mainly due to strong competition from tree roots. This impact is dynamic and increases as trees mature. Carberry *et al.* (2007) also found a similar impact on cropping paddocks from adjacent trees. They found that the width of significant decrease due to this edge effect had a linear relationship with tree height. There are, however, also many studies which have shown that outside this immediate 'edge', trees will have a positive impact on pastures and crops in adjoining paddocks (Shelton *et al.*, 1987; Moreno *et al.*, 2007; Gea-Izquierdo *et al.*, 2009; Donaghy *et al.*, 2010; Moustakas *et al.*, 2013).

An Australian carbon offset scheme, the Carbon Farming Initiative (CFI), was introduced in July 2012 with the aim of enticing landholders to reduce GHG emissions through the adoption of approved activities (Macintosh & Waugh, 2012). The CFI has been introduced with the objective of reducing the costs associated with meeting

Australia's mitigation targets (Macintosh, 2013) and allows for emissions from four sectors: stationary energy, industrial processes, fugitive emissions from mines and non-legacy waste (Macintosh & Waugh, 2012) to be offset.

While the existence and importance of transaction costs is widely acknowledged in environmental policies (Cacho *et al.*, 2005; McCann *et al.*, 2005; Coggan *et al.*, 2010; McCann, 2013), these are rarely quantified, or where they are included, they are usually presented as a simple static value for all landholders involved in a scheme. Given the complexity of project approval, reporting, crediting and compliance of carbon offset schemes such as the CFI, transaction costs can be inhibiting to individual landholders (Fitchner *et al.*, 2003). Therefore, individual landholders may not be able to directly interact with these schemes. However, it is possible that a project developer can manage a pool of individual landholder contracts to gain economies of scale (Henry *et al.*, 2009; Mattsson *et al.*, 2009; Cacho *et al.*, 2013). This aggregation of a large number of landholders can also help reduce project failure risks.

Welsch *et al.* (2014) posited that the spatial distribution of current farm features is an important factor to consider with policies for ecosystem improvements. Most spatial studies in the Australian context investigate carbon collected at either a cell or per hectare level without regard for the spatial distribution of current farm features. The current chapter investigates the impact of farm and landholder heterogeneity while taking into account the actual operating units, individual paddocks and farm level data. In order for the CFI to be successfully adopted, properties with both low opportunity cost and low transaction costs must be identified. Where the net benefits are positive, the inclusion of co-benefits to neighbouring paddocks will also increase the feasibility of the CFI. Therefore, the objectives of this chapter are to assess the potential economic viability of carbon sequestration through environmental plantings at the property and paddock level,

assess the significance of heterogeneous landholder transaction costs, determine the influence of co-benefits on the viability of carbon projects and assess the role of aggregators in obtaining larger pools of landholders. Transaction costs for both individual landholders and project aggregators are quantified using the typology described by Cacho and Lipper (2007) and Cacho (2009). A model based on Cacho *et al.* (2013) was adapted to determine the likelihood of a decision to participate in a CFI project in three case study regions in northern New South Wales, Australia. Project feasibility frontiers based on different market prices of carbon are determined. This chapter concludes with a discussion on the feasibility of projects under the current CFI rules and possible improvements to future policy design.

4.2 Model

4.2.1 Carbon trajectories

The carbon sequestration potential of individual farms from land-use change is mapped both spatially and temporally. This potential is determined as per hectare trajectories [$Cha(t)$] of additional carbon offsets that can be obtained from a land-use change for each eligible paddock. To avoid overestimating additional carbon sequestration potential, the current level of woodiness (i.e. existing carbon storage) in a paddock was taken into account when calculating the carbon sequestration trajectories of each paddock. Adjustments were made using a woodiness index (ξ) and the following equation:

$$C_i(t) = Cha_i(t) \cdot (1 - \xi) \cdot a_i \tag{4.1}$$

where $Cha_i(t)$ is the trajectory of carbon offsets that can be sequestered per hectare on property i if no trees are currently growing within the paddock over the period T of the project ($t = 1,\ldots,T$) and a_i is the area of the i-th paddock in hectares.

4.2.2 Project feasibility

The project feasibility model of Cacho *et al.* (2013) was adapted to consider the feasibility of carbon sequestration projects using a fixed bundle of heterogeneous farms, modelled down to individual paddock scale. This model considers a single project developer (an aggregator) who will purchase carbon offsets from landholders adopting particular land-use changes. The project developer will purchase these carbon offsets from the individual landholders at a farm-gate price (p_F) and will combine and sell them in carbon markets at price p_C. Obviously, the individual landholders will incur a range of abatement costs, including the opportunity costs of foregone income associated with procuring these offsets. The project developer will also incur costs of designing, acquiring and managing carbon contracts with the individual landholders. A project will only be feasible if both the individual landholders and the project developer perceive their participation to be beneficial. It is assumed that for a project developer to take action, the benefits of selling the carbon offsets in the market must be greater than the abatement and transaction costs of aggregating offsets from individual landholders. For an individual to participate, the benefit of selling carbon offsets to the project developer must be greater than the abatement and transaction costs associated with joining the scheme. These conditions are presented as equations (4.2) and (4.3) and both must be satisfied for a project to be feasible.

$$V_C(a, p_C, C(t), \delta_B) \geq V_A(a, p_F, C(t), \delta_B) + V_T(a, n, p_C, C(t), \mathbf{W}, \delta_B) \tag{4.2}$$

$$v_C(a, p_F, C(t), \delta_S) \geq v_A(a, R(t), \delta_S) + v_T(w, \delta_S) \tag{4.3}$$

where V_C and v_C are the present values of the revenues received by the project developer from selling carbon in the market and by the landholder from selling to the project developer, respectively; V_A and v_A are the present values of the abatement costs to the

project developer and the landholders; V_T and v_T are the present values of the transaction costs for the project developer and the individual landholders; $C(t)$ is the trajectory of carbon offsets which can be produced over the life of the project ($t = 1,...,T$); a is the total area of land converted; n is the number of individual landholders; W and w are cost vectors containing the different classes of transaction cost for the project developer and the individual landholders; and δ_B and δ_S are the discount rates for the project developer and the landholders.

As mentioned in Chapter 2, it is argued that, when assessing sequestration projects, physical carbon needs to be discounted in the same manner as project costs[k] (Richards, 1997; van Kooten *et al.*, 2004; Boyland, 2006). Not adopting a discount rate implies that no time preference exists between removing carbon from the atmosphere now or at a future point in time. Cost estimates are sensitive to the length of project period. Adding a discount rate on both costs and physical carbon places more importance on carbon sequestration in the near future, allowing cost estimates to account for the timing of the sequestration. Therefore, when determining project feasibility in this chapter, both these elements are presented in discounted terms.

4.2.3 The project developer

The discounted sum of payments received by the project developer is derived from collecting carbon offsets from n individual landholders producing carbon offsets in i paddocks, and selling them in the carbon market at price p_C:

$$V_C = \sum_{t=1}^{T}\sum_{n=1}^{N}\sum_{i=1}^{i_n} p_C C_{n,i}(t)(1+\delta_B)^{-t} \tag{4.4}$$

[k] Discount rates assumed for project costs and physical carbon do not necessarily need to be identical.

where i_n is the number of paddocks that landholder n will convert to a carbon-offset-producing land use.

The abatement cost for the project developer is the present value of the farm-gate payments for the carbon offsets paid to the individual landholders:

$$V_A = \sum_{t=1}^{T} \sum_{n=1}^{N} \sum_{i=1}^{i_n} p_F C_{n,i}(t)(1+\delta_B)^{-t} \tag{4.5}$$

In addition to the abatement cost payments incurred by paying the landholders in exchange for carbon offsets, the project developer will also incur a range of transaction costs associated with finding appropriate parcels of land, negotiating with eligible landholders and measuring, certifying and monitoring carbon-offset products before they can be sold in the carbon market[l]. Project developer's transaction costs are estimated as:

$$V_T = W_{S1} + W_A + W_{P1} + nW_{S2} + \sum_{t=1}^{T} \left[W_{P2} + W_{M1} + nW_{E2} + (W_{M2} + W_{E1}) C(t) p_C \right] (1+\delta_B)^{-t} \tag{4.6}$$

where the letter and number subscripts of (W) are adapted from the transaction costs notation used in Cacho *et al.* (2013). The letters represent search and negotiation costs (S), approval costs (A), project management costs (P), monitoring (M) and enforcement and insurance costs (E). Number subscripts refer to costs which are measured using different units within each individual transaction cost category. A list of these transaction costs is presented in section 4.2.4.

4.2.4 The individual landholder

The present value of the revenue received by an individual landholder for joining the carbon offset scheme is the sum of the discounted farm-gate carbon payments:

[l] There is a growing recognition that project-level accounting for carbon offsets gives rise to the difficult and common problems of additionality, permanence, and leakage (Richards and Andersson, 2001). Alternatives such as national-level accounting have been proposed to reduce these problems along with reducing transaction costs associated with the management of individual projects (Andersson and Richards, 2001; Plantinga and Richards, 2010).

$$v_C = \sum_{t=1}^{T} \sum_{i=1}^{i_n} p_F C_i(t)(1+\delta_S)^{-t} \tag{4.7}$$

The cost of abatement for the individual landholder is determined using the opportunity cost of switching land use.

$$v_A = \sum_{t=1}^{T} \sum_{i=1}^{i_n} R_i(t)(1+\delta_S)^{-t} \tag{4.8}$$

where $R_i(t)$ is the flow of differences between the net revenues of the best alternative land use and the carbon-offset scheme in the *i*-th paddock. The opportunity cost of each eligible paddock is determined based on the current land-use type. Two broad categories of current land use were estimated; cropping (consisting of both dryland and irrigated cropping) and livestock (consisting of both native and improved pasture) enterprises. The estimation of the opportunity costs for each eligible paddock is described in Appendix C.

The discounted stream of transaction costs for individual landholders joining a carbon offset scheme is:

$$v_T = \left[w_{S1} p_L + w_{S2} d_{\min} p_{trav} + w_A + \sum_{t=1}^{T} \left(w_{P1} p_L + w_{P2} d_{\min} p_{trav} + w_E \right)(1+\delta_S)^{-t} \right] \tag{4.9}$$

where p_L is the opportunity cost of the landholder time, $d_{\min}$ is the minimum distance to the nearest town (estimated using the least-cost algorithm described in Chapter 3), p_{trav} is the cost of travel (\$ km^{-1}), and the letter transcripts for the individual landholder transaction costs (w) are the same as those used in equation (4.6). The number subscripts are defined in Table 4.1.

Table 4.1: Description of transaction costs and the notation adopted in this study (based on Cacho *et al.*, 2013).

Notation	Description	Incurred by
W_{S1}	Search and negotiation (fixed)	Project developer
W_{S2}	Search and negotiation (variable)	Project developer
W_A	Approval (fixed)	Project developer
W_{P1}	Project management (fixed)	Project developer
W_{P2}	Project management (annual)	Project developer
W_{M1}	Monitoring (annual)	Project developer
W_{M2}	Monitoring (per credit)	Project developer
W_{E1}	Enforcement and insurance (per credit)	Project developer
W_{E2}	Enforcement and insurance (per farm)	Project developer
w_S	Search and negotiation (fixed)	Project developer
w_A	Approval (fixed)	Landholder
w_P	Project management (annual)	Landholder
w_E	Enforcement and insurance (annual)	Landholder
p_{trav}	Travel cost (per km)	Landholder
p_l	Cost of labour (per day)	Landholder

4.2.5 Project feasibility frontiers

The maximum price that the aggregator would be willing to pay individual landholders, when including their transaction costs can be determined by substituting equations (4.4) and (4.5) into equation (4.2) and rearranging to obtain:

$$p_F \leq p_C - \frac{V_T\left(n, a, \mathbf{W}, C(t), \delta_B\right)}{\sum_{t=1}^{T}\sum_{n=1}^{N}\sum_{i=1}^{i_n} C_{n,i}(t)(1+\delta_B)^{-t}} \tag{4.10}$$

The minimum feasible farm price for an individual landholder depends on the sum of the abatement and transaction costs and can be found by substituting equations (4.7) and (4.8) into equation (4.3) and rearranging to obtain the following equation:

$$p_F \geq \frac{v_T + \sum_{t=1}^{T}\sum_{i=1}^{i_n} R(t)(1+\delta_S)^{-t}}{\sum_{t=1}^{T}\sum_{i=1}^{i_n} C(t)(1+\delta_S)^{-t}} \tag{4.11}$$

where the numerator is the total cost to the landholder, which includes the transaction and abatement costs and the denominator is the discounted carbon offsets produced.

Project feasibility is determined by the ability of the aggregator to fully cover the cost of the farm-gate carbon payments (p_F) for any given price of carbon in the market (p_C). The minimum project size can be determined by setting equation (4.10) equal to equation (4.11). As pointed out by Cacho *et al.* (2013), the ability to cover these costs is dependent on the number of individual farms, the total area of land converted to carbon offsets and the amount of discounted carbon. At each level of p_C the value of n (number of participating landholders), a (total area) or discounted carbon that satisfies the minimum project size condition while keeping all other parameters constant can be solved. Cacho *et al.* (2013) called this the Project Feasibility Frontier (PFF) expressed as:

$$x_{\min}\left(p_C \mid a, w, W, C(t), R(t), \delta_B, \delta_S\right) \tag{4.12}$$

This function represents the minimum project size ($x_{\min}$) that is feasible as a function of the market price of carbon for the given value of the other parameters. The variable $x_{\min}$ can represent the minimum number of landholders, minimum total area or minimum total discounted carbon. Once heterogeneity is included, an upper bound ($x_{\max}$) is introduced. This can be expressed as:

$$x_{\max}\left(p_C \mid a, w, W, C(t), R(t), \delta_B, \delta_S\right) \tag{4.13}$$

The area between $x_{\min}$ and $x_{\max}$ represents the feasible project range. This is illustrated graphically in section 4.4.3.

4.2.6 Accounting for additional benefits

Additional benefits may be experienced in paddocks adjoining those paddocks which are planted for carbon offset purposes. These additional benefits to agricultural production

include increased survival and weaning rates of livestock and protection of crops from prevailing winds. Donaghy *et al.* (2010) divided paddocks into different regions based on adjoining tree height and their relative position to tree plantings when estimating impact on yield. This approach has been adopted in the current chapter and is graphically depicted in Figure 4.1.

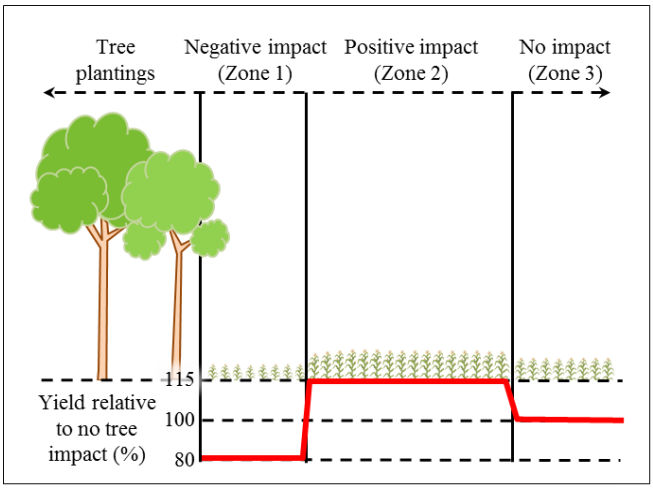

Figure 4.1: Impact of trees on crop or pasture in adjoining paddocks where width of area for the negative and positive impacts are 1x and 4x the height of trees, respectively (adapted from Donaghy *et al.*, 2010, p. 475). Note, figure not to scale.

Each paddock's yield (be it either crop or pasture gain) was determined with the following equation[m]:

$$Yield_{\text{mod}}(t) = Yield \cdot a_1(t) \cdot \tau_1 + Yield \cdot a_2(t) \cdot \tau_2 + Yield \cdot a_3(t) \cdot \tau_3 \tag{4.14}$$

where $Yield_{\text{mod}}(t)$ is the modified pasture or crop yield for a paddock influenced by the planting of trees in adjacent paddocks, a_1, a_2 and a_3 are the area of the paddock with reduced, increased and unaffected production in time period t, respectively and τ_i are the yield-modifying parameters. The length of the 'edge' adjacent to any of the neighbouring paddocks is determined to calculate the different paddock regions. Due to the data-intensive nature of dynamic paddock coordinates, a computational restriction is

[m] Both the positive and negative benefits in terms of increased/decreased agricultural productivity is captured at the localised scale with this equation. While other co-benefits or perverse effects may exist, they were not considered in this study.

applied where a paddock can only be influenced by a maximum of five neighbouring paddocks. Where more than five adjacent paddocks exist, the five paddocks with the longest interface are selected. In addition, a restriction is applied which requires all paddocks to be at least 10 metres wide to provide any additional benefits. This restriction is included to avoid paddocks providing unrealistic benefits.

When determining the optimal paddocks for conversion to a carbon-offset project, the following algorithm was used:

1. Place all eligible paddocks into a set[n].
2. Systematically consider each paddock in the 'eligible pool' set to determine the cost of capturing carbon while taking into account the influence from paddocks in the 'already planted' set and the paddock under current consideration.
3. Find the lowest-cost paddock and remove this paddock from the 'eligible pool' set.
4. Move the last paddock removed from the 'eligible pool' set into the 'already planted' set.
5. If no paddocks remain in the 'eligible pool' set, move to 6, otherwise return to step 2.
6. The set 'already planted' contains the order in which paddocks should be added to achieve additional carbon capture from the individual landholder.

4.3 Case study regions

The Border Rivers-Gwydir catchment covers an area of approximately 5,000,000 hectares. Due to the large scale of the catchment, three case regions were selected based on the results from Chapter 2 using the following criteria:

[n] 'Eligible paddock' in this sense is a paddock where the current land use can be changed to trees for carbon capture in an offset scheme.

- highest carbon sequestration rate potential per hectare (Case region A);
- lowest mean opportunity cost per hectare (Case region B); and
- lowest positive mean opportunity cost per hectare (Case region C).

An algorithm in MATLAB was run to determine the location of these three regions. Coincidently, all three regions occurred along a border of the catchment (see Figure 4.2).

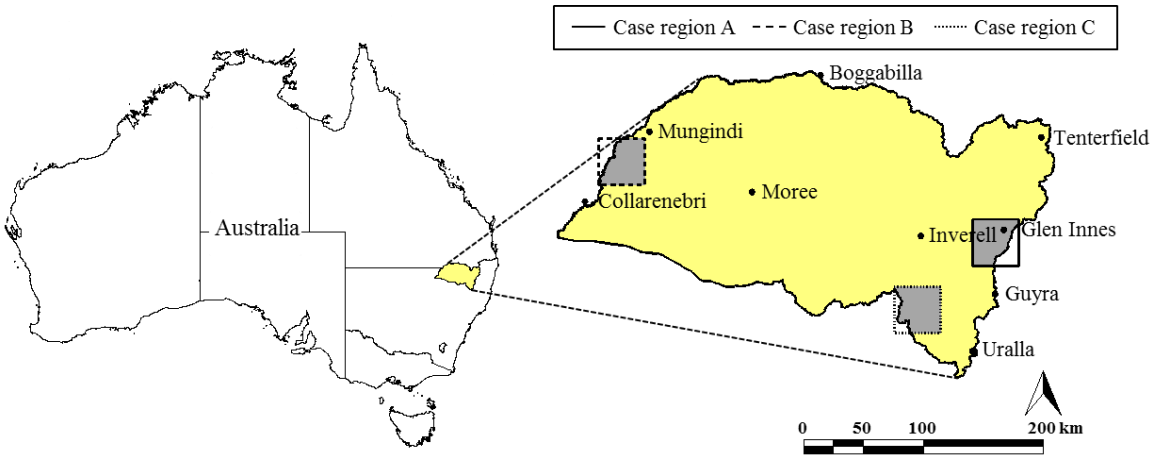

Figure 4.2: Location of the case regions in the Border Rivers-Gwydir catchment.

The general characteristics of the different case regions are shown in Table 4.2.

Table 4.2: General characteristics of the case regions.

	Case study region		
	A	**B**	**C**
Case region size (ha)	128,199	166,446	180,360
Properties (no)	328	46	154
Size of property (ha)[#]	391 ± 544	3,618 ± 7,186	1,171 ± 1,841
Paddocks per property (no)[#]	27 ± 27	107 ± 184	51 ± 80
Average size of paddock (ha)[#]	10 ± 4	45 ± 86	16 ± 5

[#]Values are in terms of mean ± standard deviation.

Case region A has the highest average additional carbon sequestration potential per hectare (determined in Chapter 2), but also the smallest average property size (391 ha). This high carbon sequestration potential is partially attributable to the position on the easterly aspect of the catchment and the relatively high average annual rainfall of

856mm. On the other hand, case region B has a larger average property size (3,618 ha) but a lower carbon sequestration potential per hectare. A summary of the current land uses of the case study regions is shown in Table 4.3.

Table 4.3: Current land uses of the case study regions (as a proportion of the total area).

	Case study region		
Current land use	**A**	**B**	**C**
Nature conservation	-	-	1.20%
Other minimal uses	5.71%	1.50%	12.69%
Grazing of native pastures	-	43.64%	-
Forestry	-	0.05%	-
Plantations	0.40%	0.32%	0.19%
Grazing of modified pastures	86.00%	0.12%	76.95%
Cropping	2.88%	43.58%	0.67%
Irrigated pastures and cropping	-	4.15%	0.44%
Urban intensive uses	2.20%	1.13%	0.36%
Water	1.41%	2.01%	1.29%
Unspecified land uses	1.41%	3.51%	6.21%

In Chapter 2, the carbon sequestration potential over the entire Border Rivers-Gwydir catchment in northern NSW was assessed at a scale of 1.1 km^2 cells. The model used at that scale did not take into account the structure of individual farms, nor were the transaction costs of participating in a carbon offset scheme explicitly estimated. The current chapter models carbon sequestration potential at a paddock level using GIS shapefiles which contain the paddock boundaries, cropping history and level of woodiness for all paddocks in the Border Rivers-Gwydir catchment from summer 1998 to summer 2009 (NSW Government, 2009b). Digital cadastral data containing property boundaries in the study regions was obtained from the NSW Government, Land and Property Information (NSW Government, 2011).

4.3.1 Increased carbon storage

Each paddock across the case study regions was assessed for eligibility in the CFI scheme. If a paddock was not currently wooded and was being used for either dryland or irrigated cropping, grazing of native or improved pasture by livestock, or for other minimal agricultural production, the paddock was deemed eligible for conversion to a mixed-species environmental planting. The conversion of current paddocks to environmental plantings assumes that the current fencing infrastructure will allow the exclusion of stock for at least the first three years of a project, ensuring adherence to the approved methodology (DCCEE, 2011). Carbon sequestration trajectories for each eligible paddock across the three case regions were estimated using the CFI-approved methodology for mixed-species environmental plantings (DCCEE 2011). Practically, this was done through the use of the prescribed Reforestation Modelling Tool for a period of 100 years. In keeping with this approved methodology, baseline emissions were set at zero and not recalculated over the course of the project. Fuel emissions from the establishment and management of the environmental plantings were accounted for using the equations outlined in Schedule 1 of the National Greenhouse and Energy Reporting (Measurement) Determination (Australian Government, 2011b).

4.3.2 Parameter and cost assumptions

Permanent environmental plantings must have the potential to attain a height of at least 2 metres and a crown cover of at least 20% of the project area to be eligible under the approved methodology in the Australian CFI (DCCEE, 2011, pp. 4-5). For the study regions, spatial climate, soil and vegetation parameter sets for mixed-species environmental plantings, were derived from DCCEE (2011). As this methodology has been approved for the generation of carbon offsets in Australia, it was assumed that these parameter sets ensured compliance with the height and crown-cover conditions.

Net revenues for the different paddocks were estimated based on a variety of secondary data (ABS, 2008; BRS, 2009; ABARES, 2012; NSW DPI, 2012a, 2012b, 2012c). A sample of the parameters used is provided in Appendix D, with full details listed on the supplementary material CD-ROM. For the livestock gross margins, individual transportation costs for purchases and sales were determined using the minimum-cost travel distance method described in Chapter 3. Distances were estimated from each of the case study properties to pre-existing saleyards, abattoirs and feedlots (see Figure 4.3). Landholder and project developer discount rates were assumed to be 9% and 7%, respectively. Assumptions for the different project developer and landholder transaction costs were based on estimates by Cacho *et al.* (2013) and are shown in Table 4.4.

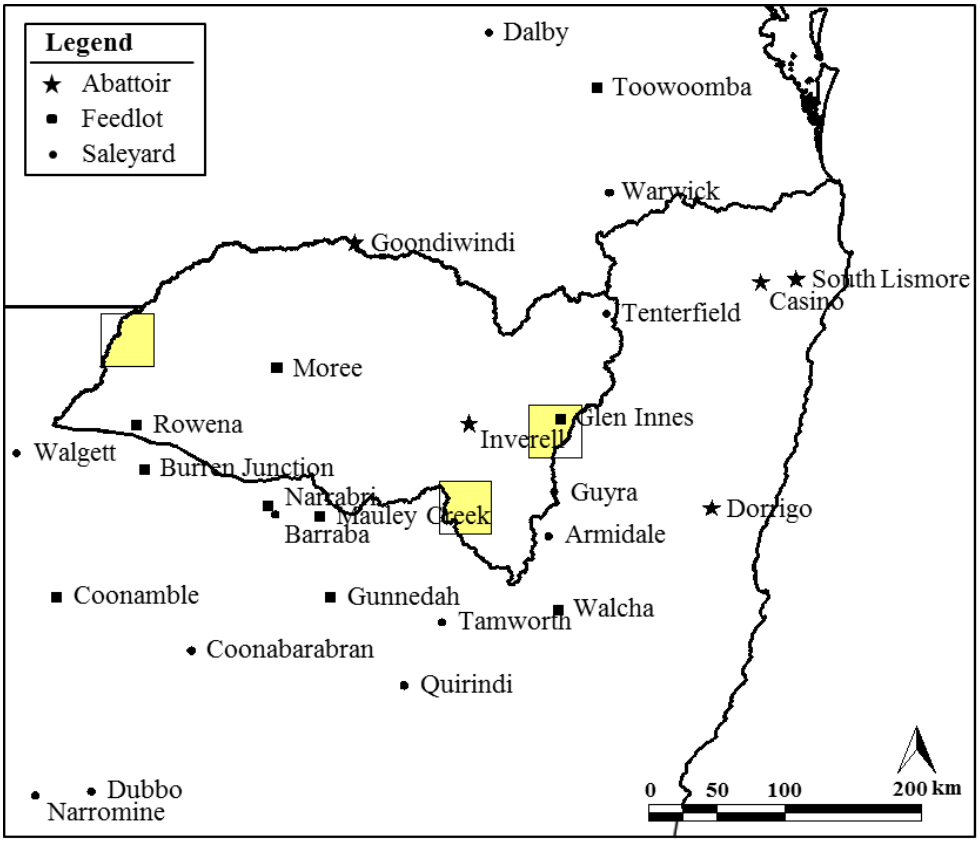

Figure 4.3: Location of currently existing abattoirs, feedlots and saleyards in proximity to the case study regions.

Table 4.4: Assumptions used for transaction costs in the case study regions (based on Cacho *et al.*, 2013).

Notation	Description	Units	Value
W_{S1}	Search and negotiation (fixed)	$	43,500
W_{S2}	Search and negotiation (variable)	$ farm^{-1}	2,500
W_A	Approval (fixed)	$	15,000
W_{P1}	Project management (fixed)	$	9,000
W_{P2}	Project management (annual)	$ yr^{-1}	100,000
W_{M1}	Monitoring (annual)	$ yr^{-1}	5,000
W_{M2}	Monitoring (per credit)	credit yr^{-1}	0.02
W_{E1}	Enforcement and insurance (per credit)	credit yr^{-1}	0.05
W_{E2}	Enforcement and insurance (per farm)	$ farm^{-1} yr^{-1}	500
w_S	Search and negotiation (fixed)	days	3
w_A	Approval (fixed)	$	750
w_P	Project management (annual)	$ yr^{-1}	*
w_E	Enforcement and insurance (annual)	credit yr^{-1}	0.05
p_{trav}	Travel cost (per km)	$ km^{-1}	0.55
p_l	Cost of labour (per day)	$ day^{-1}	184

***Landholder project management fees vary based on geographical location and are calculated as one day of labour plus travel costs to the nearest major town for an annual meeting with the project developer.**

Parameters used to determine the impact of tree plantings on neighbouring paddocks were based on Donaghy *et al.* (2010) and are shown in Table 4.5.

Table 4.5: Parameter values used for determining additional benefits to adjoining paddocks.

Notation	Description	Units	Value
τ_1	Pasture or crop yield, relative to no impact from trees in Zone 1	%	80
τ_2	Pasture or crop yield, relative to no impact from trees in Zone 2	%	115
τ_3	Pasture or crop yield, relative to no impact from trees in Zone 3	%	100
a_1	Area of Zone 1	ha	*
a_2	Area of Zone 2	ha	*
a_3	Area of Zone 3	ha	#

***These parameter values are dynamic in nature and are determined as length of adjoining tree plantations multiplied by 1x and 4x the tree height for a_1 and a_2, respectively. #Area of Zone 3 is calculated as total area of paddock minus the sum of a_1 and a_2.**

Planting, follow-up weed control and annual management cost parameters of $801 ha^{-1}, $80 ha^{-1} and $16 ha^{-1}, respectively were applied to the mixed-species plantings based on Polglase *et al.* (2008).

4.4 Results

4.4.1 Carbon sequestration potential

The detailed paddock simulations undertaken in this chapter found the region on the eastern side of the Border Rivers – Gwydir catchment (case region A) to have the highest carbon sequestration potential. This is consistent with the findings in Chapter 2. Figure 4.4 illustrates that the highest quantity of annual carbon sequestration occurs within the first eight years of changing land use. The average annual additional carbon sequestration in year eight is 4.89 t C ha^{-1}, 2.14 t C ha^{-1} and 2.37 t C ha^{-1} for regions A, B and C, respectively. There is a substantial variance across each region with additional carbon sequestration in year eight ranging between 2.77 t C ha^{-1} and 6.76 t C ha^{-1}, 1.39 t C ha^{-1} and 2.61 t C ha^{-1} and 1.77 t C ha^{-1} and 3.58 t C ha^{-1} for regions A, B and C, respectively.

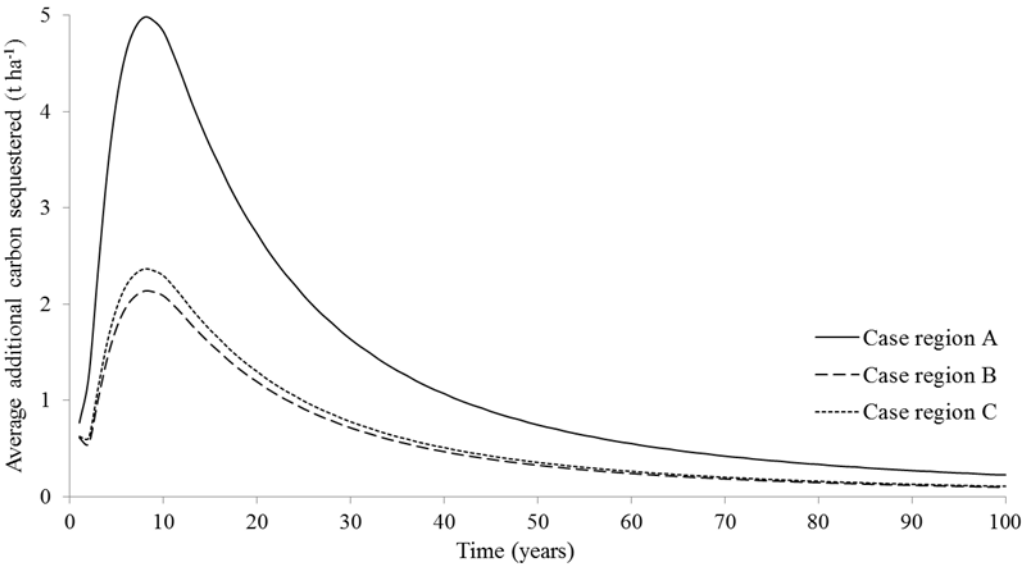

Figure 4.4: Average annual additional carbon sequestration potential across the three case study regions.

The total average additional carbon which can be expected over the 100 years of the project is 18.27 t C ha^{-1} (range 10.16 t C ha^{-1} – 24.78 t C ha^{-1}) in region A, 7.83 t C ha^{-1}

(range 5.10 t C ha^{-1} – 9.57 t C ha^{-1}) in region B and 8.67 t C ha^{-1} (range 6.49 t C ha^{-1} – 13.13 t C ha^{-1}) in region C.

4.4.2 Feasible price to landholders

The minimum feasible farm-gate price for each landholder (P_S) was determined for landholders in the case regions. A significant variance in the minimum feasible farm-gate price was found with prices ranging from \$8.95 t CO_2-e^{-1} to \$295.99 t CO_2-e^{-1} (Table 4.6).

Table 4.6: Minimum feasible farm-gate price to landholders across the case study regions.

	Mean (\$ t CO_2-e^{-1})	**St. dev. (\$ t CO_2-e^{-1})**	**Min (\$ t CO_2-e^{-1})**	**Max (\$ t CO_2-e^{-1})**	**Coefficient of variation (%)**
Case region A	28.06	10.92	8.95	65.30	38.93
Case region B	27.16	8.26	19.07	65.69	30.40
Case region C	52.18	35.73	16.09	295.99	68.48

The average coefficient of variation (CV) for the minimum feasible farm-gate price across the case study regions was 45.94% (determined from the regional CVs in Table 4.6), with a range from 30.40% to 68.48%. Project developers will need to account for this heterogeneity when assessing project feasibility.

4.4.3 Feasible project areas

In order for the available technologies and management strategies which can provide climate mitigation products to be adopted, they must be economically feasible to all stakeholders. For a project to be feasible, the price required by landholders (P_S), which is determined through equation (4.11), must be less than or equal to the maximum amount that the aggregator (the buyer) is willing to pay (P_B); determined with equation (4.10). This is demonstrated, at a market price of \$23 t CO_2-e^{-1}, for region A, in Figure 4.5.

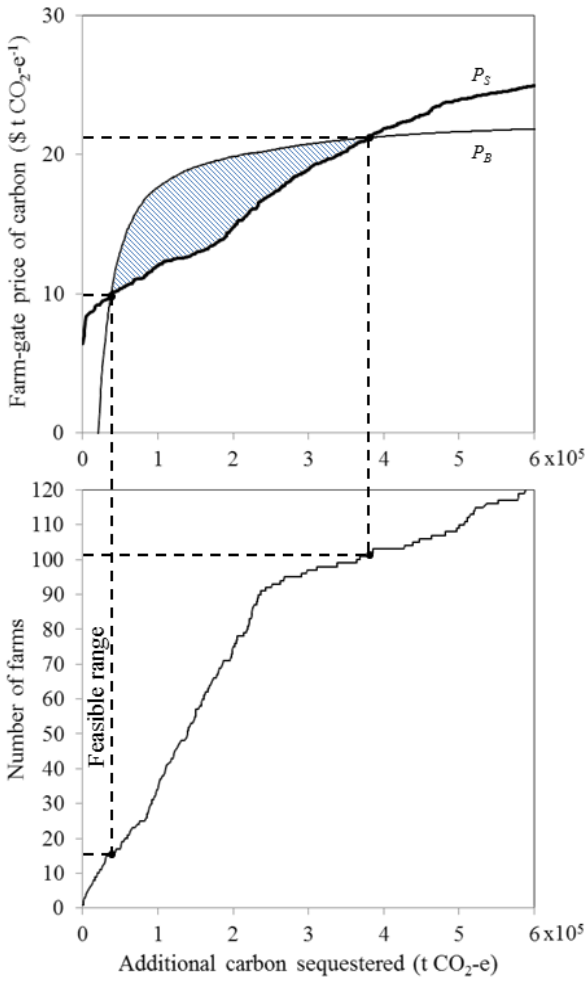

Figure 4.5: Supply of carbon for individual landholders (P_S), the maximum carbon price for the aggregator (P_B) and the feasible number of farms with a market price of \$23 t CO_2-e^{-1}. The feasible range represents the number of contracts in the project.

When the project developer is only able to aggregate a small number of landholders, sufficient economies of scale have not been reached and the minimum farm-gate price acceptable to the landholders (P_S) is greater than the maximum farm-gate price that the aggregator would be willing to pay. As the aggregator encourages more landholders to enter the project, the maximum farm-gate price that they are willing to pay landholders increases. In Figure 4.5, it becomes feasible for the project aggregator and some landholders at a farm-gate price of approximately \$10 t CO_2-e^{-1}, on the condition that the aggregator is able to obtain at least 36,000 t CO_2-e^{-1}[o]. This is the first point of intersection between P_S and P_B. At a quantity of approximately 382,000 t $C0_2$-e^{-1}, P_S and

[o] This value can be determined from the *x*-axis for the minimum feasible bound in Figure 4.5.

P_B again intersect, indicating that no landholders would be willing to add additional paddocks to an offset project as the farm-gate price they would require is greater than the amount the aggregator would be willing to pay (>$21 t CO_2-e^{-1}). Where x_{min} and x_{max} in equations (4.12) and (4.13) are used to represent the viable project size in terms of number of landholders, the feasible range, at this market price for a project is between 16 and 102 landholder contracts.

The project feasibility frontiers (PFF) represent the feasible quantity of carbon that may be sequestered under a project at different carbon prices. In effect, this provides useful sensitivity analyses on the impact of the market carbon price on a range of project characteristics. Unlike the study by Cacho *et al.* (2013) which assumed an unlimited supply of landholders, the current study considers the actual number of landholders available to a project aggregator and accounts for heterogeneity in farm size, land-use types, paddock configurations and productivity. Although this places an arbitrary constraint given by the size of the case study regions chosen for a project, it does demonstrate that an upper limit exists on feasible project size. The feasible project sizes for the three case study regions are shown in Figure 4.6.

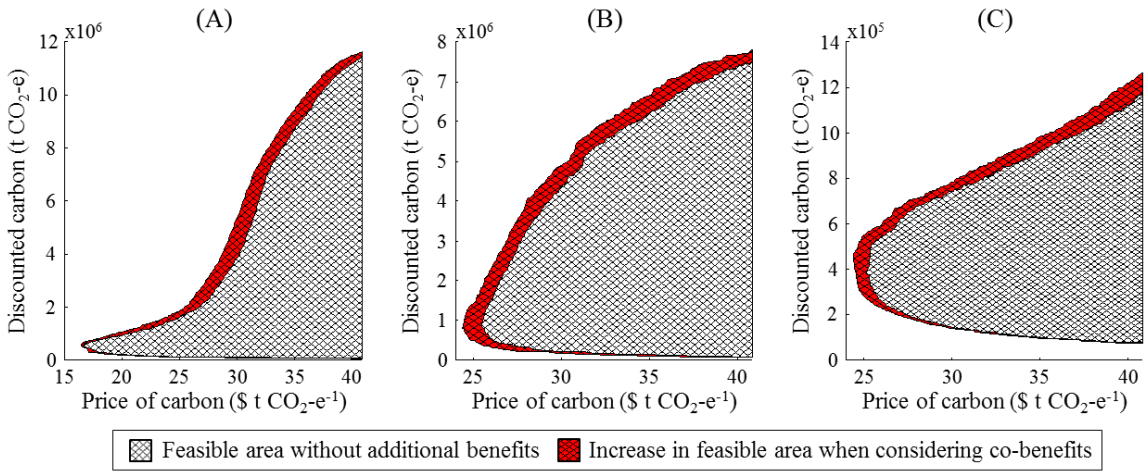

Figure 4.6: Project feasible areas for the three case study regions, showing the feasible project range both with and without the inclusion of additional benefits from the carbon plantings. Feasible range is plotted as the area between x_{min} and x_{max} at each level of p_C, determined with equations (4.12) and (4.13), respectively. Note, different axes scales are used to enhance readability.

Cacho *et al.* (2013) stated that a project is feasible if it falls above or to the right of a PFF curve, but this may not be the case where heterogeneity and availability of suitable land are considered. Ignoring this fact may cause sequestration potential to be overstated.

As discussed in section 4.2.2, the discounting of physical carbon, along with the project costs, allows weighting of carbon sequestration through time and comparison between projects. Feasible project areas are therefore presented in terms of discounted carbon in the current section. When projects that do not account for the additional co-benefits on neighbouring paddocks are considered, it can be seen that projects will become feasible in case region A when the market price of carbon is \$16.70 t CO_2-e^{-1} or higher (Figure 4.6); on the condition that the project aggregator is able to secure enough land to sequester a discounted carbon quantity of at least 563,915 t CO_2-e. On the other hand, for case regions B and C, projects will only become feasible when the market price of carbon is \$25.50 t CO_2-e^{-1} and \$25.16 t CO_2-e^{-1} or higher, respectively. As outlined in section 4.2.6, planting trees can influence the productivity and profitability of adjoining paddocks. This is discussed in the following section.

4.4.3.1 Accounting for additional benefits

When accounting for the impact of co-benefits[p], it was found that overall there is an additional benefit of tree plantings on neighbouring paddocks which influences the feasibility of projects in each of the case study regions. This effect is shown in the project feasibility diagrams in Figure 4.6. When the additional benefits are included, the minimum feasible carbon price decreases by \$0.27 t CO_2-e^{-1} (1.63%), \$1.09 t CO_2-e^{-1} (4.28%) and \$0.75 t CO_2-e^{-1} (2.98%) for case regions A, B and C, respectively. The

[p] The term 'co-benefits' may refer to a range of different benefits, including increases in biodiversity, salinity reduction or improved agricultural productivity. In this chapter, it is used in the sense of profitability from increased agricultural productivity on neighbouring paddocks.

greater reduction in minimum price in region B compared to the other regions can be attributed to the higher value cropping paddocks in this region, thus resulting in higher dollar benefits from adjoining trees.

Employing a model which accounts for the additional benefits to neighbouring paddocks also increases the amount of area that is feasible at different carbon prices. Across the feasible range of prices up to \$40 t CO_2-e^{-1}, there is an average 23.17% increase in the area of eligible land that would be feasible for both an aggregator and the individual landholders for region A. Similarly, regions B and C display an average 36.06% and 13.45% increase across the same range of prices.

4.4.3.2 Alternative PFF inputs

In addition to the range of carbon that must be sequestered to ensure a project is just feasible at any given price, a number of other factors can be plotted as PFFs, making this technique a useful tool for decision making. These include total project size in hectares and total number of farms (contracts).

The number of hectares provides a convenient, readily understood and measureable unit of feasible project size. It should be noted, however, that this needs to be coupled with localised carbon sequestration estimates to determine the quantity of carbon sequestration that could be expected from such an area. The minimum number of hectares that must be secured depending on carbon price for a project in each of the case study regions, to just be feasible is illustrated in Figure 4.7. This is based on the actual heterogeneous sequestration potential across the properties.

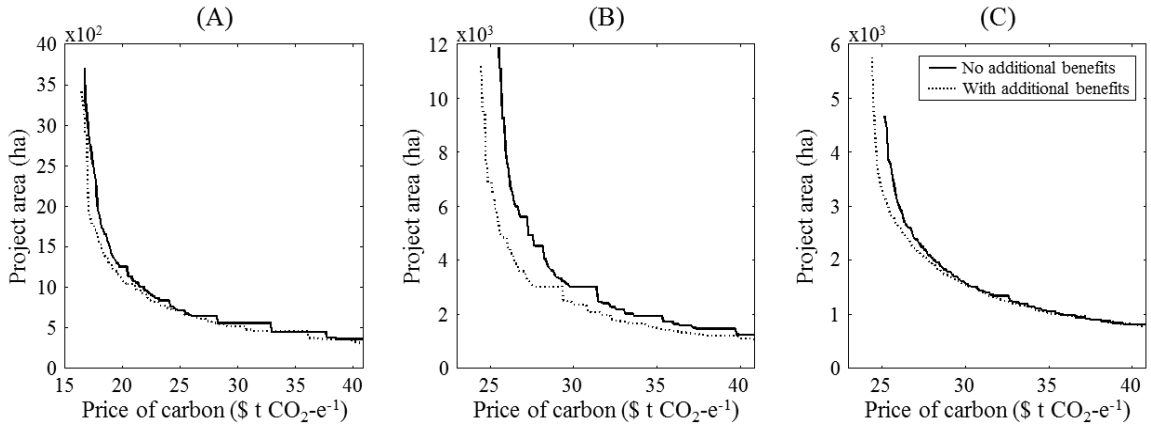

Figure 4.7: Minimum project size in terms of area for each of the case study regions. Note, different axes scales are used to enhance readability.

The minimum number of farm contracts that are required by an aggregator to just break-even is another useful measure which provides a practical business tool (Figure 4.8).

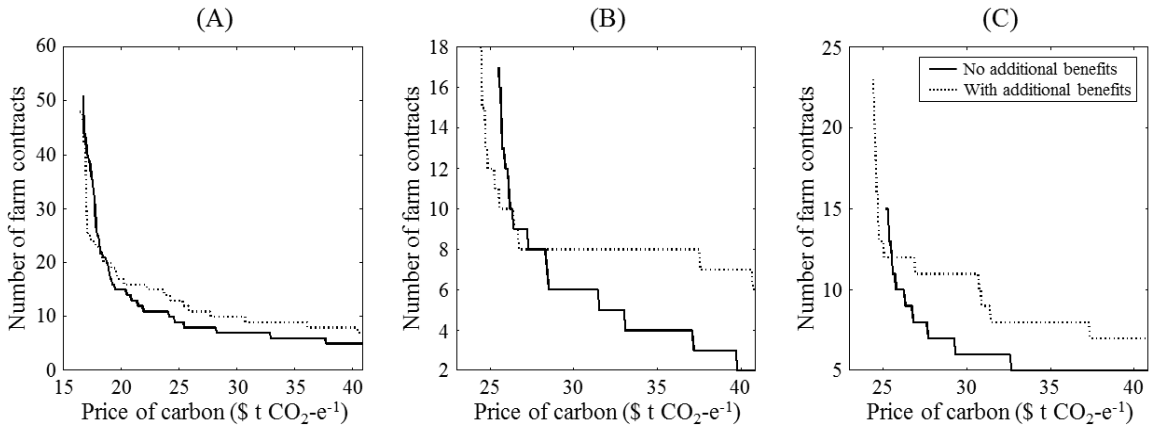

Figure 4.8: Minimum number of farm contracts required for the carbon sequestration project to be feasible. Note, different axes scales are used to enhance readability.

Interestingly, results show that the number of farm contracts required is generally lower when additional benefits are not included. This can be attributed to the fact that lower opportunity costs occur when the additional benefits are included which results in landholders being willing to enter smaller parcels of land into carbon plantations. Therefore, while more farm contracts are required when including these additional benefits, the actual area required is less than that required when these additional benefits are not included.

4.4.4 Heterogeneous nature of regions

A wide variance in the carbon sequestration potential across the different regions, as well as across the different paddocks within an individual landholder's property was evident in the current study. Figures 4.9 – 4.11 depict the spatial variance in average additional carbon that could be expected in eligible paddocks if they were converted to mixed-species carbon plantations. Case region A has a higher sequestration potential in the south-eastern localities and potential in case region B is highest in the eastern side of the sections. In contrast, case region C displays its highest potential in the western extremities due to their proximity to water courses.

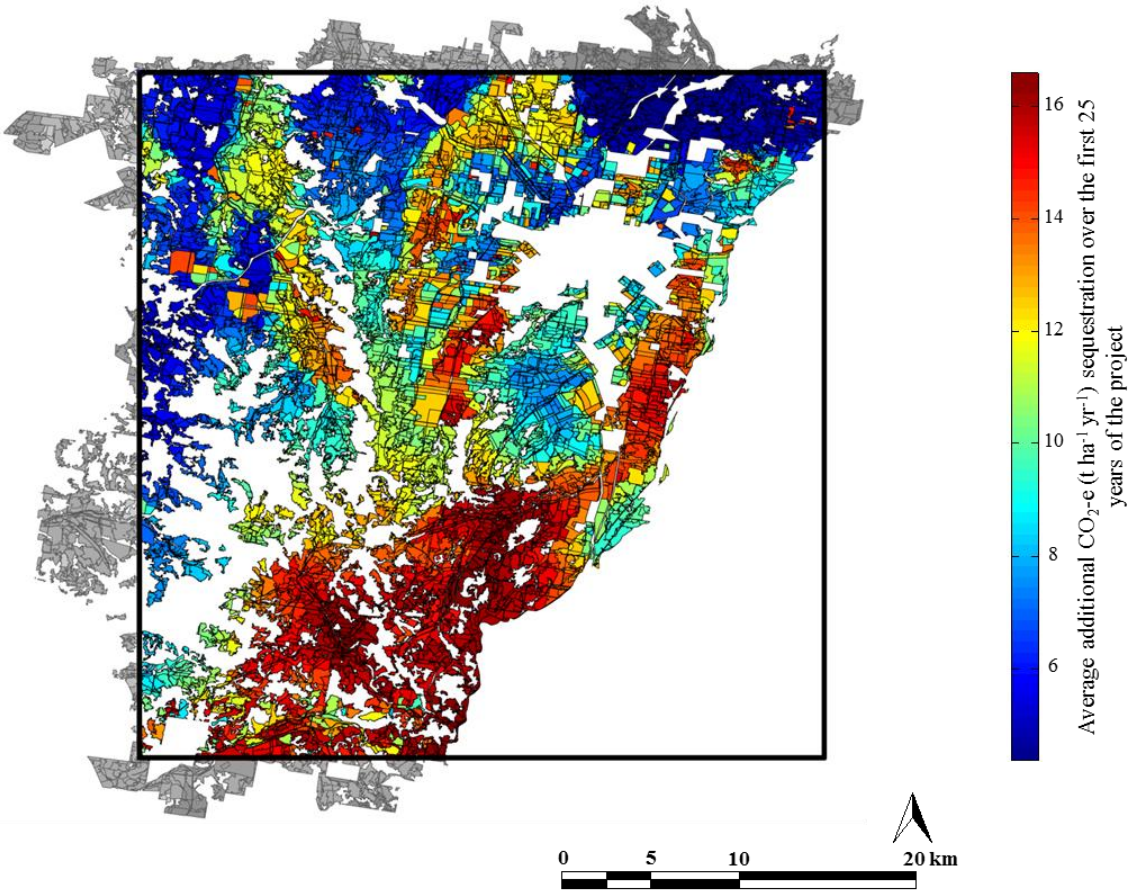

Figure 4.9: Spatial distribution of average annual carbon sequestration potential for eligible paddocks in case region A.

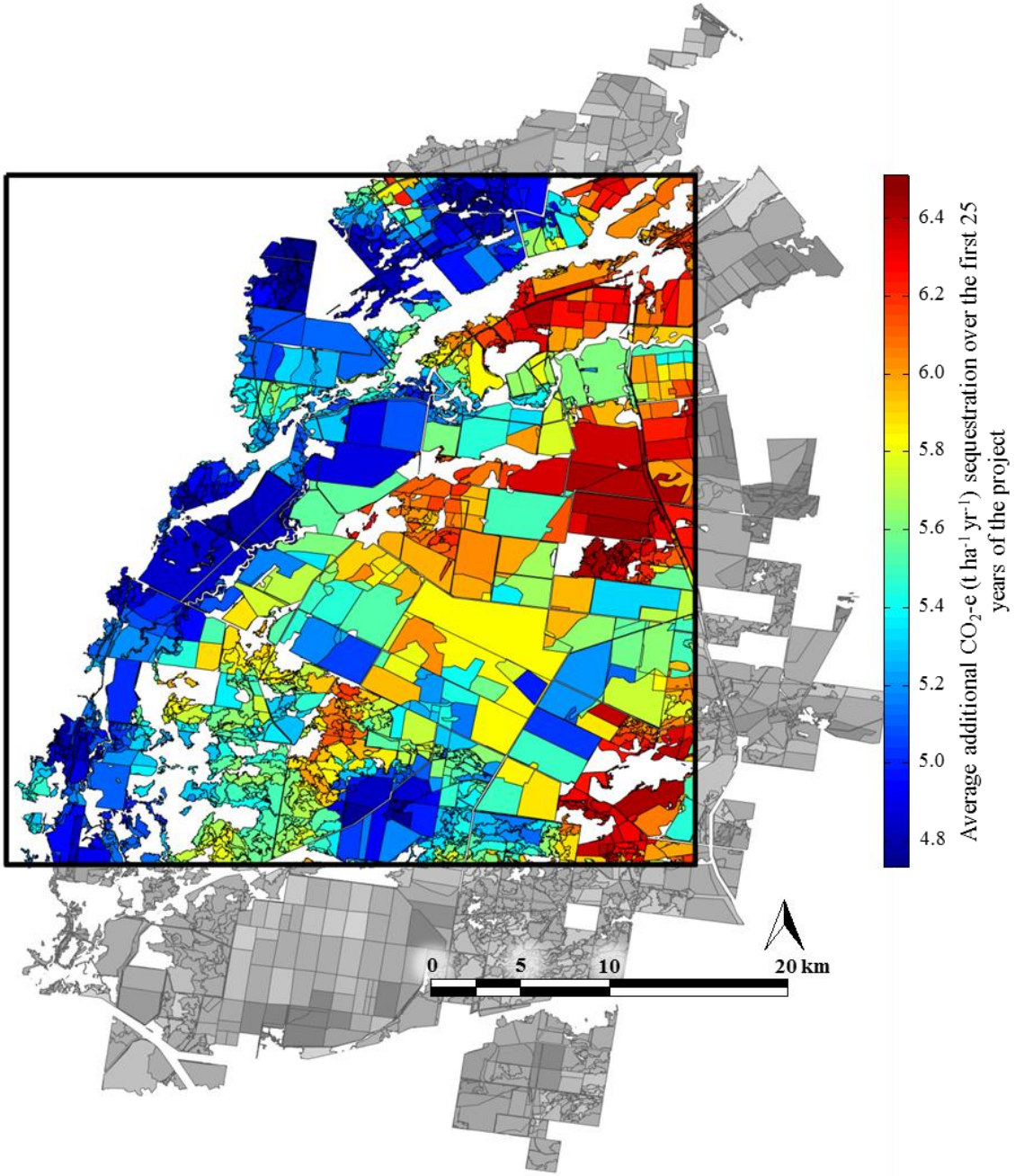

Figure 4.10: Spatial distribution of average annual carbon sequestration potential for eligible paddocks in case region B.

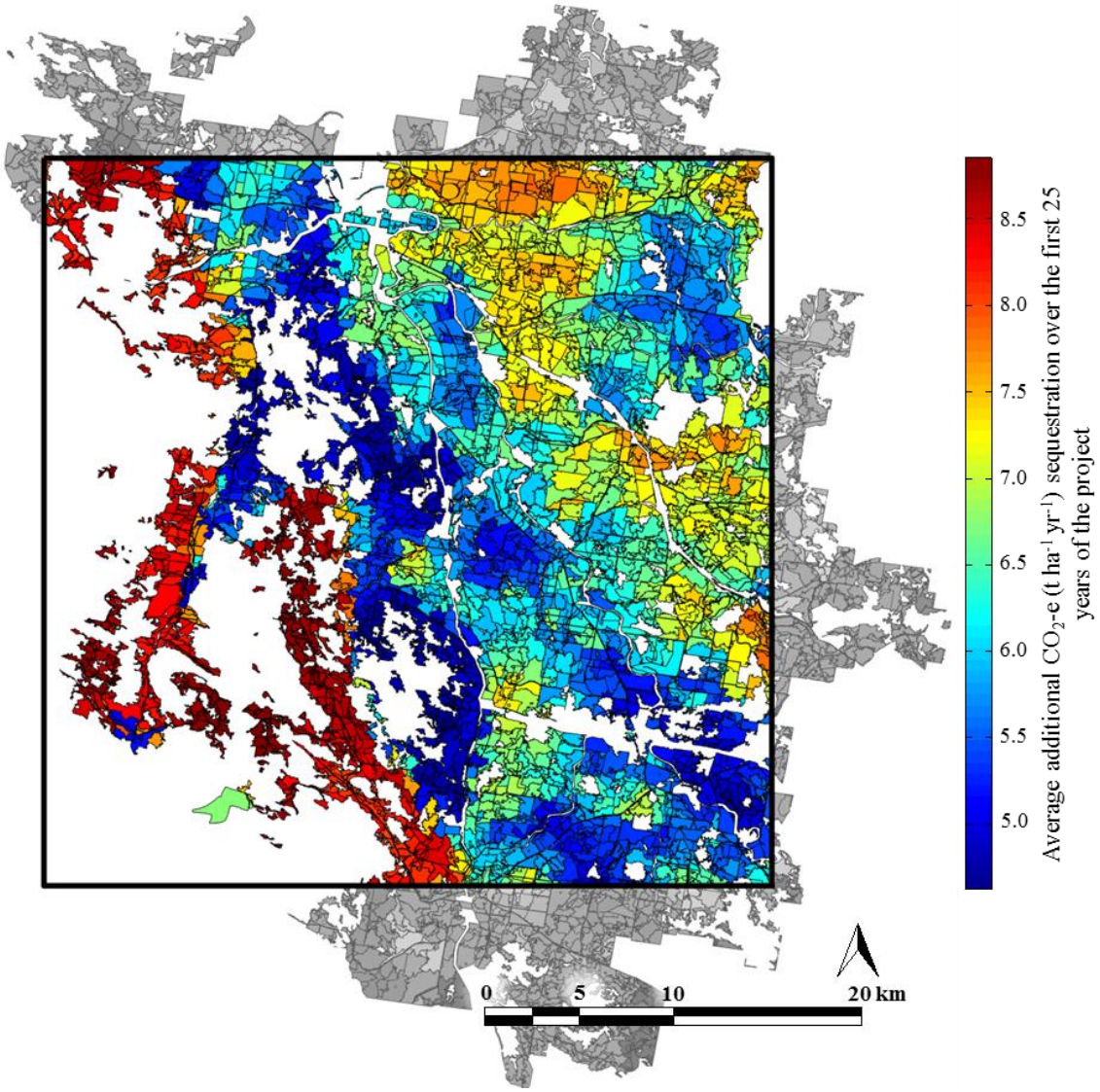

Figure 4.11: Spatial distribution of average annual carbon sequestration potential for eligible paddocks in case region C.

From a temporal point of view, the variance in sequestration potential becomes more pronounced through time, as shown by the carbon trajectories in Figure 4.12. The average coefficient of variation (CV) of the additional carbon sequestration potential across each of the three case study regions was 12.67%, with a range between 6.98% and 17.89%. These CV values are lower than those reported by Paul *et al.* (2013b) who estimated an average CV of 39% for carbon sequestration potential across their study regions in south-eastern Australia. The smaller variances in the present study may be attributable to the smaller case study regions, the use of a single block planting

configuration and a mixed-species carbon plantation strategy with no additional inputs compared to the multiple planting configurations, forestry plantations and the inclusion of nitrogen fertiliser inputs which were included in the Paul *et al.* (2013b) study.

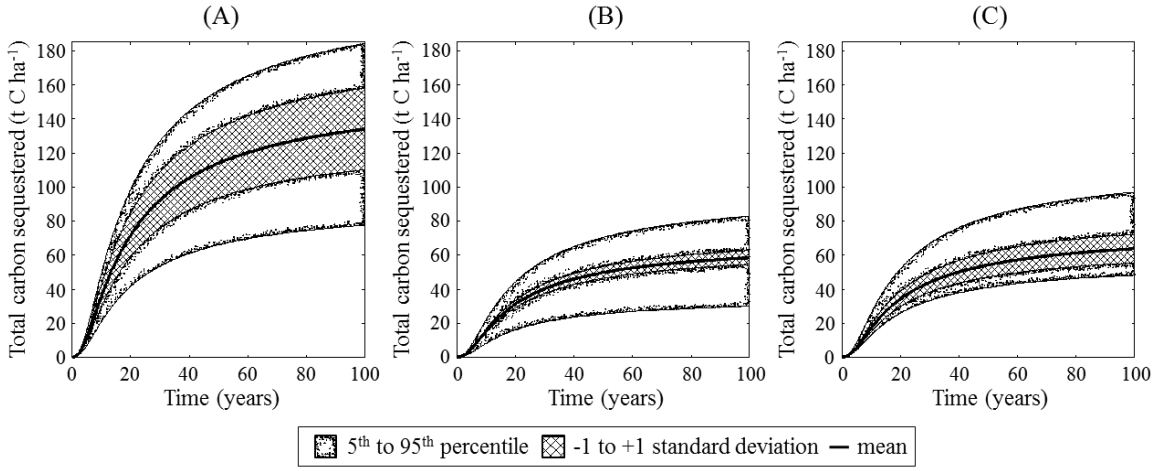

Figure 4.12: Expected carbon trajectories for the three case region areas.

Assuming that the current management practices and the opportunity costs are homogeneous across a study region is obviously a limiting assumption. Therefore, an additional analysis was undertaken to determine the difference between the actual heterogeneous set of landholders in the case study regions and the assumption regarding a set of homogeneous landholders based on the regional average. For case region A, when assuming a homogeneous group of landholders, it was estimated that no areas would be feasible until the market price reaches \$33.62 t CO_2-e^{-1}. This is a 105% increase in market price required for properties in this case study region to participate compared to the heterogeneous landscape that exists in the region (Table 4.7). For case regions B and C, the assumption of homogeneous regions would see the minimum market price required for any area to be included in a carbon plantation scheme, increase by 20% and 22% respectively.

Not surprisingly, the estimated minimum area that must be contracted by an aggregator for the scheme to just be viable is also affected by the assumption of a homogeneous group of landholders. For case region A the minimum area of land required was between 179% and 669% more than that required when accounting for the heterogeneity across the region. For case region B, the difference was between 47% and 275% and for case region C between 139% and 648% (not shown).

Table 4.7: Increase in minimum feasible starting price when assuming a homogenous landholder set.

	Case region		
	A	**B**	**C**
Homogenous ($ t CO_2-e^{-1})	34.23	30.68	30.68
Heterogeneous ($ t CO_2-e^{-1})	16.70	25.50	25.16
Difference (%)	105	20	22

These findings highlight that the failure to account for the heterogeneous nature of the landscape, and the variance in properties across a region, may significantly overestimate both the market price of carbon and the quantity of carbon sequestration required before projects will become feasible.

Antle and Valdivia (2006) described a technique for generating supply curves for ecosystem goods derived from a heterogeneous population of landholders in the US. They derived upward-sloping supply curves by arranging farms in ascending order of opportunity cost. The heterogeneous nature of the landholder populations in the present study allows this technique to be employed. The resulting supply curves, which include transaction costs, are shown in Figure 4.13.

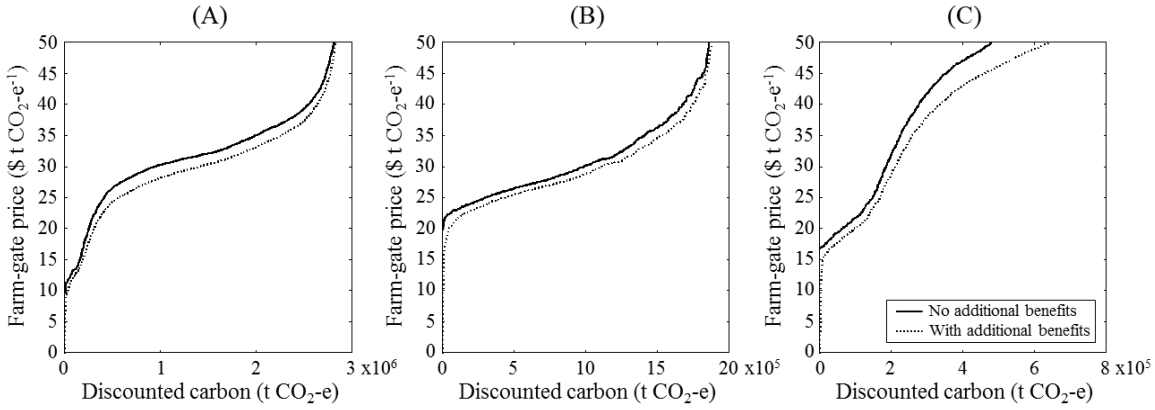

Figure 4.13: Supply curves for each of the case study regions illustrating cost of carbon offsets both with and without accounting for additional benefits.

An aggregator or policy maker can use these figures to determine the quantity of carbon that may be purchased at different prices from landholders in a region. As discussed earlier, accounting for the influence of carbon plantations on neighbouring paddocks will provide an additional benefit which will move the supply curves down and to the right when these benefits are captured by the landholder[q]. This implies that greater quantities of carbon will be obtained at any price if these additional benefits are taken into account.

4.4.5 Sensitivity analysis

The PFF diagrams presented in Figures 4.6 – 4.8 provide a graphical depiction of the sensitivity of a number of important inputs in relation to a change in the market price of carbon. In addition to this, a sensitivity analysis was undertaken on a range of additional variables. These are presented in Table 4.8 as elasticities in terms of percentage change in the required minimum project area.

[q] Additional public benefits from carbon plantations, such as the generation of wildlife corridors, are not captured in this model as they will not influence these supply curves.

Table 4.8: Results from the sensitivity analysis on the three case study regions at a market carbon price of \$29 t CO_2-e^{-1}[r]. Values are in terms of elasticies as percentage change in the minimum feasible project area (ha) in response to a one per cent change in the value of each variable/parameter.

	Case region		
	A	**B**	**C**
Aggregator transaction costs	1.28	1.58	1.55
Landholders discount rate	0.45	2.51	1.23
Cost of agricultural inputs	0.17	-0.45	-0.27
Cost of cattle production inputs	0.10	0.01	-0.15
Cost of sheep production inputs	0.10	-0.06	-0.30
Landholder transaction costs	-0.03	-0.14	0.14
Cost of cropping inputs	-0.07	-0.33	0.00
Income from cattle production	-0.10	-0.01	-0.14
Aggregators discount rate	-0.22	-0.26	-0.12
Crop yield	-0.26	0.47	0.00
Income from sheep production	-0.35	0.13	0.37
Agricultural yields and outputs	-0.36	0.66	0.32
Income from livestock production	-0.45	0.11	0.32
Influence of additional benefit	-0.52	-3.46	-1.06

Interestingly, the signs of the elasticities were not consistent for different assumptions across the regions. For example, an increase of one per cent in the cost of agricultural inputs in case region A would see a 0.17% increase in the minimum quantity of area required for a project to become feasible. On the other hand, the same one per cent increase in the cost of agricultural inputs would see a 0.45% and 0.27% decrease in the minimum feasible project area for case regions B and C, respectively[s].

Despite the landholder transaction costs not having a major influence on the minimum feasible project area, the elasticities for case regions A and B suggest that a one per cent increase in these costs will result in a decrease of 0.03% and 0.14% in the minimum area

[r] A market price of \$29 t CO_2-e^{-1} was assumed to allow feasible areas for each case region to be determined when conducting the sensitivity analysis.

[s] Due to the heterogeneous but fixed nature of eligible paddock areas, the variance of different parameter assumptions can cause a different set of larger paddocks to be available at a lower cost, resulting in a positive elasticity in some cases where a negative elasticity would be expected.

required for a project to be feasible in the region, respectively. This reduction in minimum required area in response to higher landholder transaction costs is due to a different set of properties being selected as optimal. When the sensitivity analysis was conducted on the simulation model which ignored the additional benefits, the elasticity ranged between 0.00% and 0.06% (results not shown).

From the elasticities in Table 4.8, we can see that in addition to the aggregator transaction costs, the additional benefit to neighbouring paddocks and the landholder's discount rate are critical factors. The assumption of the parameter value on the beneficial influence of trees past the initial 'edge' effect (τ_2) is particularly elastic in case region B (elasticity = -3.46). This region has a greater proportion of high value crop paddocks which neighbour eligible carbon plantation paddocks. Therefore, any change to the assumption for the benefit received on these adjacent paddocks will be more pronounced than in case regions A and C, which have lower value enterprises, reflected in elasticities of -0.52 and -1.06, respectively for this parameter.

The landholder's discount rate also had most influence in region B, with an elasticity on the minimum feasible area of 2.51. Again this can be attributed to the influence on the choice of high value crops when they are discounted over a long period of time (100 years). In comparison, the elasticities with respect to landholder discount rate in regions A and B are 0.45 and 1.23. This highlights that the choice of discount rate is an important consideration when modelling carbon sequestration policies. Interestingly, the choice of aggregator discount rate had a significantly lower influence on the minimum feasible project area, with elasticities of -0.22, -0.26 and -0.12 on regions A, B and C respectively.

4.4.6 Proportion of property placed in scheme

Simulations in the current study found that, on average, the minimum proportion of a property required for carbon plantations to be feasible to an individual landholder was 18.35% (± 13.56%) at a market carbon price of \$23 t CO_2-e^{-1}. At this carbon price, there were no feasible projects in either case region B or C. If a market carbon price of \$25 t CO_2-e^{-1} is assumed, the average minimum proportion of a landholder's property required is 18.16% (± 12.52%), 6.52% (± 7.13%) and 9.20% (± 13.50%) for case regions A, B and C, respectively. As the market price of carbon increases, the minimum proportion of a landholder's property requiring conversion to carbon plantations decreases. At a price of \$35 t CO_2-e^{-1} these minimum proportions decrease to 14.38% (± 9.33%), 2.79% (± 2.9%) and 5.72% (± 8.81%) for case regions A, B and C, respectively. While the estimated total proportion of individual landholder's property required to become feasible is higher in case region A, when viewed in terms of actual hectares required, case region A requires an average of only 71 hectares to become feasible, compared to 236 hectares for case region B and 108 hectares for case region C.

4.5 Discussion

Simulations in the current chapter have shown that there is significant technical potential to sequester carbon through mixed-species carbon plantations. The results from this chapter have shown that the minimum feasible price for carbon sequestration projects in northern NSW will vary depending on the location. The lowest market price at which these projects will become feasible is \$16.40 t CO_2-e^{-1} in case region A. For the other two case regions, projects will not become feasible until a market price of approximately \$25 t CO_2-e^{-1} is reached. Given the recent declining global price on carbon (Newell *et al.*, 2013), the findings from this chapter indicate that the current framework of the CFI may be limiting the potential supply of carbon sequestration from private landholders.

Alternative policies and strategies to reduce the cost of this carbon capture, together with methods of encouraging participation by individual landholders, need to be investigated.

4.5.1 Encouraging landholder participation

Currently, there is a low level of interest in providing long-term carbon plantations in Australia without considerable subsidies or incentives (Hunt, 2008; Patrick *et al.*, 2009). The findings in the current chapter suggest that, when negotiating terms of contract with an aggregator, encouraging landholders to include additional benefits of planting trees into their opportunity cost estimations may reduce the level of incentives required. It was found that the inclusion of this allowance reduced the minimum feasible price of carbon plantation projects across the case study regions by between 1.63% and 4.28%. The findings show that the area of eligible land available to an aggregator will also increase if landholders are educated on the additional private benefits of planting trees for climate mitigation purposes. Therefore, policy makers should investigate the impact of extension and education programs as a method of reducing the cost of carbon sequestration by private landholders.

Results were found to be highly sensitive to the assumed influence of additional benefits to paddocks adjacent to carbon plantations. There appears to be a lack of rigorous scientific research on the beneficial impacts for adjacent paddocks as a result of planting blocks of trees. Therefore, given the highly elastic nature of this parameter, further scientific research and empirical data will increase the accuracy of estimates.

Given the permanence requirement and the long-term nature of planting native trees for carbon mitigation, farmers may be reluctant to enter large proportions of their property into such schemes. Patrick *et al.* (2009) conducted a study on landholders' willingness to participate in the production of environmental services from a region bordering the

current study. They found that 86% of the landholders surveyed did not wish to commit to long-term or perpetuity environmental management agreements. A survey of landholders in central and southern NSW by Schirmer and Bull (2011) reported that a 100-year permanence requirement would be a significant barrier to entry for a carbon offset scheme. Likewise, Markowski-Lindsay *et al.* (2011) found that contract length and concerns over early withdrawal penalties influenced landholders participation in carbon markets. In a study of participation in land diversion schemes in the UK, Brotherton (1989) found that even when presented with economically viable incentives, landholders were only willing to place on average seven per cent of their total landholding into a scheme requiring the conversion of productive agricultural land to woodlands. Raymond and Brown (2011), in an Australian study, found that highly engaged landholders were likely to maintain an average area of 19.54% of their farm to native vegetation and they argued that there is limited scope to expand the areas of conservation with these landholders. They did, however, find that moderately engaged landholders, on average, maintain 12.56% of their property to native vegetation but highlighted that this demographic has the most potential to increase the areas of conservation for environmental purposes.

Therefore, given the evidence that landholders will be likely to commit less than 10% to 20% of their property to trees, the results of this study are not unrealistic. As shown in Table 4.3, only 1.2% of case region C is currently conserved in ecosystem supply schemes. Thus, given the appropriate incentives and policies, there is significant potential for increasing carbon plantations for the supply of climate mitigation products.

4.5.2 Heterogeneity and local-scale estimation

Using the local-scale estimation techniques described here, case region A was found to have the highest carbon sequestration potential along with the lowest feasible project costs. This contrasts with the findings in Chapter 2 where the highest carbon sequestration potential did not correspond to the area with the lowest estimated costs of sequestration. This suggests that taking into account the individual property characteristics such as existing property infrastructure, tree cover and individual landholder costs, significantly alters the estimated feasible areas and has implications for the areas which should be targeted in policies. It was also found that assuming a homogeneous set of farms and landholders based on a regional average will overestimate the minimum feasible carbon price by a significant amount. This highlights the importance of undertaking a local-scale analysis when developing climate policies to avoid the misrepresentation of costs which occur in broad-scale analyses.

A problem with accounting for heterogeneity across a region and undertaking local scale analyses is the data intensive requirements in regards to the information on individual farm and landholder characteristics required. The procurement of this additional data may incur significant costs and require substantially more work to correctly simulate. Policy makers and aggregators will need to estimate the trade-offs between the extra time, data and modelling requirements required to simulate more accurate supply curves to allow for more informed policy making.

4.5.3 Reducing project costs

As already mentioned, the results from the simulations in this chapter suggest that the minimum feasible carbon price of projects under the current CFI framework will impose a barrier to the adoption by landholders in the study region, particularly in case regions B

and C. A further sensitivity analysis undertaken on the individual aggregator transaction-cost categories indicated that the aggregator's annual fixed costs were the most influential transaction cost on project feasibility. These findings correspond with those of Cacho *et al.* (2013). This cost category is composed of major project management costs to the aggregator such as the fixed running costs of the local office and staff salaries. Elasticities of this cost category were 0.49 for case region A, 1.53 for case region B and 0.39 for case region C (not shown). The current study assumes that local offices would be set up by private enterprises acting as aggregators in each of the case study regions. The transaction costs of maintaining local offices could be significantly reduced if existing private and government organisations with adequate existing infrastructure were to adopt an aggregator role.

4.5.4 Choice of discount rate

Studies have found that the choice of discount rate has a marked effect on the cost and feasibility of carbon sequestration projects (for example, Stern, 2007; Hunt, 2008; Torres *et al.*, 2010; Yemshanov *et al.*, 2012). The current study has also found this to be the case, particularly the discount rates assumed for landholders when determining their opportunity costs. In a survey of landholders with private forests in southern USA, it was estimated that landholders required discount rates of 13% for forestry projects lasting 25 years or more (Bullard *et al.*, 2002). The current chapter highlights that the higher the discount rate of landholders, the less feasible area will be available to project aggregators. This contradicts the findings in Chapter 2 which found that a higher discount rate would result in higher sequestration potential. In that chapter, physical carbon was not discounted, explaining the different direction of impact.

4.5.5 Policy design of landholder carbon plantations

The current study only considered the land-use change option of converting agricultural land to a mixed-species environmental planting, in accordance with the current CFI rules. A couple of recent studies have found that monoculture farm forestry plantations in south-eastern Australia have the potential to sequester approximately 22% more carbon per year when compared to mixed-species carbon plantations (Crossman *et al.*, 2011; Paul *et al.*, 2013b)[t]. Due to this higher sequestration potential, the uptake of monoculture farm-forestry plantations in the CFI scheme could further reduce the cost of carbon sequestration. In addition, if methodologies for the CFI were introduced to allow harvesting of timber strands, in exchange for carbon offset payments, landholders might be willing to accept lower payments per tonne of carbon sequestered as they would have additional economic benefits from the harvesting of timber[u]. This would also encourage a quicker rate of carbon sequestration. As the current chapter, along with Chapter 2 has shown, the highest yearly additional sequestration occurs within the first 25 years of planting. Often the carbon in harvested wood products remains locked-up for several decades (Stockmann *et al.*, 2012) and does not return directly to the atmosphere at the time of harvest. Given the modelling complexities of harvesting and carbon stocks, the current CFIs accounting rules for mixed-species native trees excludes commercial harvest (DCCEE, 2011).

With the current advancements being made in the carbon life-cycle accounting of harvested timber products (Newell & Vos, 2012; Stockmann *et al.*, 2012), it may be possible to more accurately account for a higher rate of permanent/semi-permanent

[t] This may not always be the case. In rainforest regions of north-eastern Australia, it has been found that mixed-species plantations sequestered higher quantities of carbon compared to monoculture plantations (Kanowski & Catterall, 2010).

[u] This chapter is set in the policy context of the original CFI rules where no methodologies allowing the harvest of plantations were approved. Given the dynamic nature of the Australian climate regulations, a recent amendment to the legislation now allows farm forestry activities to claim carbon-offset credits (Australian Government, 2013).

carbon sequestration in harvested wood products. When assessing the life-cycle of wood products, Ingerson (2011) estimated that approximately 14% of carbon captured in timber products across America will remain stored 100 years post-harvest. Currently, there are a handful of voluntary carbon offset markets which have explicitly recognised carbon in long-lived harvested wood products as a forest offset pool (Chicago Climate Exchange, 2009; Winrock International, 2010; Climate Action Reserve, 2012). A shift from short-term to long-term wood products from harvested products will reduce the loss of carbon captured in trees back to the atmosphere. It should be noted that the emissions involved with harvesting, processing and transporting wood products will also need to be included in any accounting policy (Ingerson, 2011).

Finally, the findings of Paul *et al.* (2013b) suggested that different planting configurations, such as belt plantings, will be more viable than the simple block plantations which have been assumed in the current chapter. Results from Chapter 2 also highlighted that no singular land-use option will provide the lowest-cost option across a region. Therefore, the influences of different planting configurations on project viability are assessed further in Chapter 5.

4.6 Conclusion

The spatial and productive heterogeneity at not only a regional scale, but also across a landholder's property, is often lost in the broader scale analyses which are frequently used by climate mitigation policy makers. The current chapter has highlighted the vast divergence both between and within the different case study regions. Ignoring this variance may significantly overestimate both the market price of carbon and the quantity of carbon sequestration required before projects will become feasible.

Results in this chapter have shown that the current methodology of planting mixed-species environmental plantings to produce carbon offsets will not be feasible until a carbon price of approximately \$25 t CO_2-e^{-1} in two of the three case study regions. Given the current trend in global carbon markets, this mixed-species, non-harvestable land use may not be a viable option. Planting monoculture plantations may increase the viability of future projects and is investigated in Chapter 5.

CHAPTER 5: PLANTING AND POLICY DESIGNS TO INCREASE LANDHOLDER UPTAKE OF AGROFORESTRY

5.1 Introduction

Planting of trees by landholders to offset carbon emissions has received considerable attention as a relatively cheap and simple approach for countries to meet greenhouse gas (GHG) emission targets (Plantinga *et al.*, 1999; Cacho *et al.*, 2008b; Polglase *et al.*, 2008; Eady *et al.*, 2009; Garnaut, 2011; Nijnik *et al.*, 2013), especially over the next 50 or so years while longer term solutions or adaptation plans are developed (Gowdy & Juliá, 2010). While this planting of trees for carbon sequestration in developing countries provides a cheap method of climate mitigation (Sathaye *et al.*, 2001; Makundi & Sathaye, 2004; Sheeran, 2006), and it has been recognised that Australia has similar potential (Harper *et al.*, 2007; Polglase *et al.*, 2008; Burns *et al.*, 2009; Crossman *et al.*, 2011; Garnaut, 2011; Mitchell *et al.*, 2012), the current rules in the Australian carbon offset mechanism may be providing barriers to this realization (Macintosh, 2013).

The introduction of the Australian Carbon Farming Initiative (CFI) legislation in 2011 (Macintosh & Waugh, 2012) provided a mechanism from July 2012 for landholders to sell carbon credits for GHG emission reductions resulting from approved[v] changes to farming practices. This chapter is set in the context of the original CFI policy (Australian Government, 2011a) where only the establishment of permanent plantings existed as an approved methodology[w] (DCCEE, 2011) and the planting of trees in an area with an average annual rainfall in excess of 600mm was an excluded offset project unless it was a mixed-species permanent planting.

[v] For a project to be eligible under the CFI legislation a project must (1) be specified in the so called 'positive list' of activities in the regulations; (2) not be required under any law at the Commonwealth, State or Territory level; and (3) not be an excluded offset project under the current legislation (Australian Government, 2011a).

[w] Given the dynamic nature of Australian climate regulations this has now changed with farm forestry recently being added as an approved methodology (Australian Government, 2013).

As the findings from Chapter 4 demonstrated, projects in Australia may not become viable under this methodology of the CFI until market prices of carbon reach approximately AUD$16 to $25 t CO_2-e^{-1}. Although an Australian price on carbon was designed to have a floor price of AUD$15 t CO_2-e^{-1} (Jotzo, 2012a), this feature is due to be abolished in July 2015 with plans to link the Australian carbon pricing mechanism (CPM) to the EU Emissions Trading Scheme (Peters-Stanley *et al.*, 2012).

Given the recent decline of the global carbon price since the global financial crisis (Newell *et al.*, 2013) and the current average global forest carbon price of US$7.80 t CO_2-e^{-1} (Peters-Stanley *et al.*, 2013), it is unlikely that significant landholder adoption of the mixed-species environmental plantings under the current CFI rules will occur. To date, reforestation and afforestation sequestration has only accounted for 0.68% (11,962 units) of the total Australian Carbon Credit Units (ACCUs) generated under the CFI (Carbon Market Institute, 2013). Current CFI rules require projects ensure permanence of carbon offsets produced. This stipulation covers the risk of carbon releases from project biomass as a result of future events such as wild fires or deliberate clearing of previously credited biomass.

The CFI legislation has three inbuilt mechanisms to address the permanence issue. These include (i) a risk of reversal buffer, accounting for 5% of a project's credits, to be deducted from all sequestration projects (Australian Government, 2011a, pp. 32-35), (ii) a requirement that all sequestration projects maintain the carbon stores for 100 years (Australian Government, 2011a, p. 164) and (iii) that carbon sequestration estimates be conservative and account for significant cyclical variations in the amount of carbon sequestered (Australian Government, 2011a, pp. 120-123). Legislation also imposes strict conditions limiting the eligibility of harvested plantations.

It has been found that monoculture harvested forests will sequester higher levels of carbon at lower cost per tonne when compared to biodiversity plantings such as the mixed-species methodology currently approved under the CFI (Hunt, 2008; Crossman *et al.*, 2011; Paul *et al.*, 2013b). These findings also suggest that the current restrictions in the legislation will have an influence on both the quantity of carbon that can be sequestered and the viability of carbon farming to private landholders in Australia. Macintosh (2013) identified this stringent 100-year permanence requirement of carbon sequestration projects under the CFI as a major barrier to its success. In a review of agroforestry and carbon sequestration potential, Nair *et al.* (2009) suggested that agroforestry systems have significant global potential for low-cost carbon sequestration as long as institutional and carbon accounting difficulties such as permanence can be overcome.

Over the past decade, carbon accounting techniques have been developed to avoid this permanence issue. These include the use of a so called 'tonne-year' approach where the quantity of carbon removed from the atmosphere for a given number of years is given an equivalency to permanence factor (Marland *et al.*, 2001), a rental market to trade temporary carbon credits (Marland *et al.*, 2001; Galinato *et al.*, 2011; Cacho *et al.*, 2013) based on a discount-based approach where different permanence periods are selected (such as the period until commercial harvest) and a permanence deduction calculation based on the project length (Kim *et al.*, 2008).

While the integration of agroforestry planting configurations within current agricultural activities is not a new concept, recent research has focussed on the additional carbon sequestration benefits produced from these designs (for example, Eady *et al.*, 2009; Jose, 2009; Nair *et al.*, 2009; Donaghy *et al.*, 2010; Aertsens *et al.*, 2013; Paul *et al.*, 2013b). Planting trees in a row configuration according to typical agroforestry design is

estimated to have a 19-39% higher rate of carbon sequestration compared to block planting configurations (Paul *et al.*, 2009; Paul *et al.*, 2013b) due to decreased intra-specific competition for light, water and nutrients (Binkley *et al.*, 2010; Ryan *et al.*, 2010; Paul *et al.*, 2013a).

Hodgman *et al.* (2012) noted that site and species selection, along with the forest management practices implemented, such as fertilisation, thinning and harvesting are the key success factors for carbon sequestration projects through afforestation and reforestation. Paul *et al.* (2013b) estimated a 39% coefficient of variation in the economic viability of farm forestry projects across and within their case study of regions in southern Australia, mainly attributed to spatial heterogeneity, the choice of tree species and management regimes. Therefore, if the CFI rules for planting permanence and non-harvesting are relaxed, there is a possibility that adopting different planting configurations and allowing monoculture plantations will provide additional economic benefits to landholders; thus reducing the level of incentives required to encourage participation.

Therefore, the objectives of this chapter are to assess the viability of carbon offset projects under different planting configurations, species selection and management regimes and to determine the impact that relaxing the rigid permanence rule in the CFI legislation will have on landholder participation. This chapter builds on the model developed in Chapter 4. The limitation of a single non-harvested mixed-species regime in a simple block planting configuration is relaxed to include harvested monoculture plantations under a number of management regimes and planting configurations. Project feasibility frontiers are used to assess project viability under these different regimes and configurations. This chapter concludes with a discussion on the current carbon

accounting rules and suggests areas for improvement to reduce the level of incentives required to encourage landholders to participate in carbon offset markets.

5.2 Method

Landholders may be unable to directly interact with carbon offset schemes such as the CFI due to high transaction costs and complexities involved with project approval, reporting, crediting and compliance (Fitchner *et al.*, 2003; Peters-Stanley *et al.*, 2013, p. 57). Project developers, however, can play a role in aggregating and managing a pool of individual landholder contracts to gain economies of scale (Henry *et al.*, 2009; Mattsson *et al.*, 2009; Cacho *et al.*, 2013). This aggregation of a large number of landholders can also help reduce risk of project failure. The model adopted in the current study is based on the mathematical model developed in Chapter 4 and is centred on carbon offsets from individual landholders being purchased at a farm-gate price (p_F) and aggregated by a project developer to sell in the market at a market price (p_C).

5.2.1 Carbon trajectories

The production and carbon sequestration potential for four different tree species were determined both spatially and temporally across a range of common management scenarios (Table 5.1). These tree species include (a) locally indigenous mixed-tree and shrub-species mix with no commercial harvesting potential, (b) monoculture *Eucalyptus cladocalyx* plantations for hardwood sawlogs and pulpwood (c) generic monoculture eucalypt species grown for pulpwood and hardwood production and (d) monoculture plantations of *Pinus radiata* for softwood sawlogs and pulpwood.

Table 5.1: Management regimes assessed for each of the possible farm forestry and carbon offset tree species.

	Management regime						
Tree species	**Code**	**Planting density**	**Fertilisation (years)**	**Weed control (years)**	**Thinned (years)**	**Pruned (years)**	**Clear-fell harvest (year)**
Mixed species							
	a	direct seeded	-	1,2	-	-	-
	b	low	-	1,2	-	-	-
	c	medium	-	1,2	-	-	-
	d	high	-	1,2	-	-	-
E. cladocalyx							
	e	high	1	1,2	9*, 22	-	30
	f	low	1	1,2	12*, 29	-	41
	g	medium	1	1,2	10*, 25	-	35
	h	high	1	1,2	-	-	22
	i	low	1	1,2	-	-	29
	j	medium	1	1,2	-	-	25
Generic eucalypt							
	k	low	1	1,2	13*	6	33
	l	medium	1	1,2	10*	4	25
	m	high	1	1,2	8*	3	19
	n	high	1	1,2	-	-	12
	o	low	1	1,2	-	-	20
	p	medium	1	1,2	-	-	15
P. radiata							
	q	high	1	1,2	13*, 22, 26, 30	-	34
	r	low	1	1,2	18*, 29, 35, 41	-	46
	s	medium	1	1,2	15*, 25, 30, 35	-	40
	t	high	1	1,2	13*, 22, 26, 30	6	34
	u	low	1	1,2	18*, 29, 35, 41	9	46
	v	medium	1	1,2	15*, 25, 30, 35	7	40

*** Non-commercial tree thinning.**

Four different planting configurations were also assessed for these plant species and regimes. These planting configurations were based on Cubbage *et al.* (2012) and Paul *et al.* (2013b) and consisted of: (i) 3.66m wide rows (or belts) of trees with 12.2m wide rows containing pastures or crops, (ii) 3.66m wide rows (or belts) of trees with 24.4m

wide rows containing pastures or crops, (iii) 3.66m wide boundary plantings running around the perimeter of an existing paddock and (iv) a block planting configuration where the entire existing paddock is planted to trees. The row planting configurations were assumed to follow a north-south orientation to maximise sunlight captured by trees. These planting configurations are depicted in Figure 5.1.

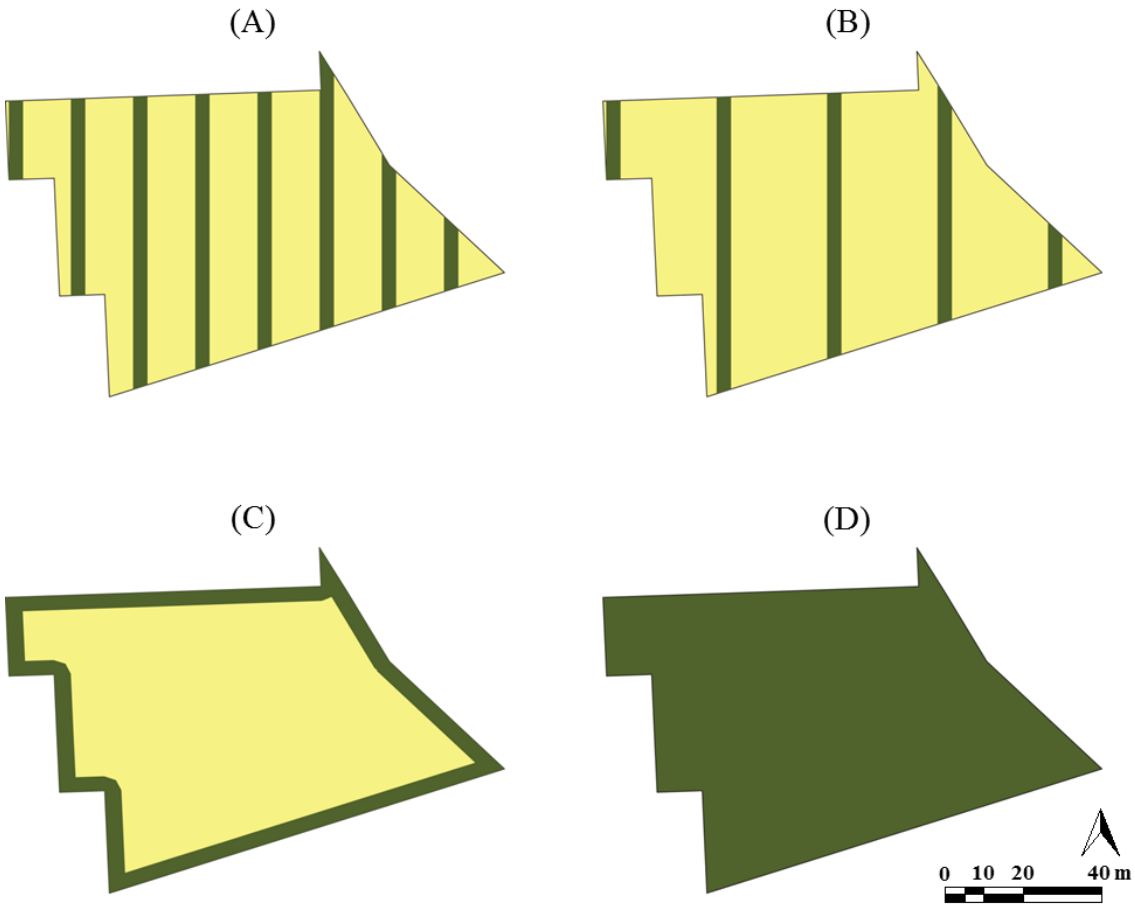

Figure 5.1: Different planting configurations assessed within existing paddock framework including: (A) 3.66m wide alley plantings with 12.2m wide inter-row cropping/pasture, (B) 3.66m wide alley plantings with 24.4m wide inter-row cropping/pasture, (C) 3.66m boundary planting around perimeter of property and (D) block planting of entire paddock.

Carbon trajectories for each of the scenarios were determined on a monthly simulation step for a 100 year period using the CFI prescribed Reforestation Modelling Tool (DCCEE, 2011). For all clear-fell harvested scenarios, it was assumed that the land would be replanted to the same species the year after clear-felling. In accordance with the current CFI rules (Macintosh & Waugh, 2012), a 5% risk of reversal buffer was

removed from the trajectory of carbon that could be claimed from any sequestration project.

Carbon trajectories [$C_{ha}(t)$] for each eligible paddock were presented in terms of tonnes of additional carbon offsets from one hectare of the land-use change scenario. As discussed in Chapter 4, the current level of woodiness in a paddock needs to be considered. The planting configuration will also affect the quantity of carbon biomass sequestered. Paul *et al.* (2013b) assumed that the sequestration rate would be between 19% and 24% greater in row plantings when compared to the same scenario under a block planting configuration. Therefore, equation (4.1) in Chapter 4 was modified to take into account this factor. The carbon sequestration trajectories of each paddock were modified using a woodiness index (ξ) and a growth rate modifier (λ), with the following equation:

$$C_i(t) = Cha_i(t) \cdot \lambda \cdot (1-\xi) \cdot a_i \tag{5.1}$$

where $Cha_i(t)$ is the trajectory of carbon offsets that can be sequestered per hectare on property i if no trees are currently growing within the paddock over the period T of the project (t = 1,…,T) and a_i is the area of the i-th paddock in hectares. The growth rate modifier parameter was assumed to have a value of 1.00 for block planting configurations and 1.19 for the row and boundary planting configurations. The value of 1.19 ensures that the benefits estimated with row or boundary plantings err on the conservative side of the estimates from Paul *et al.* (2013b).

5.2.2 Case study areas

Given the large number of scenarios, computational limitations restricted the detailed simulation of each scenario to three individual case study properties. This detailed analysis was conducted on the property from each of the case study regions which

exhibited the lowest cost of sequestering carbon in Chapter 4 (Figure 5.2). Following the detailed analysis on the feasibility of each scenario, the lowest-cost species and management regime from this analysis was simulated on each of the three case study regions assessed for a mixed-species carbon offset project in Chapter 4. This allowed the impact of the stringent permanence rules under the CFI on project feasibility to be examined. The impact of different planting configuration design when using a mixed-species planting approved under the CFI was also assessed across each of the three case study regions.

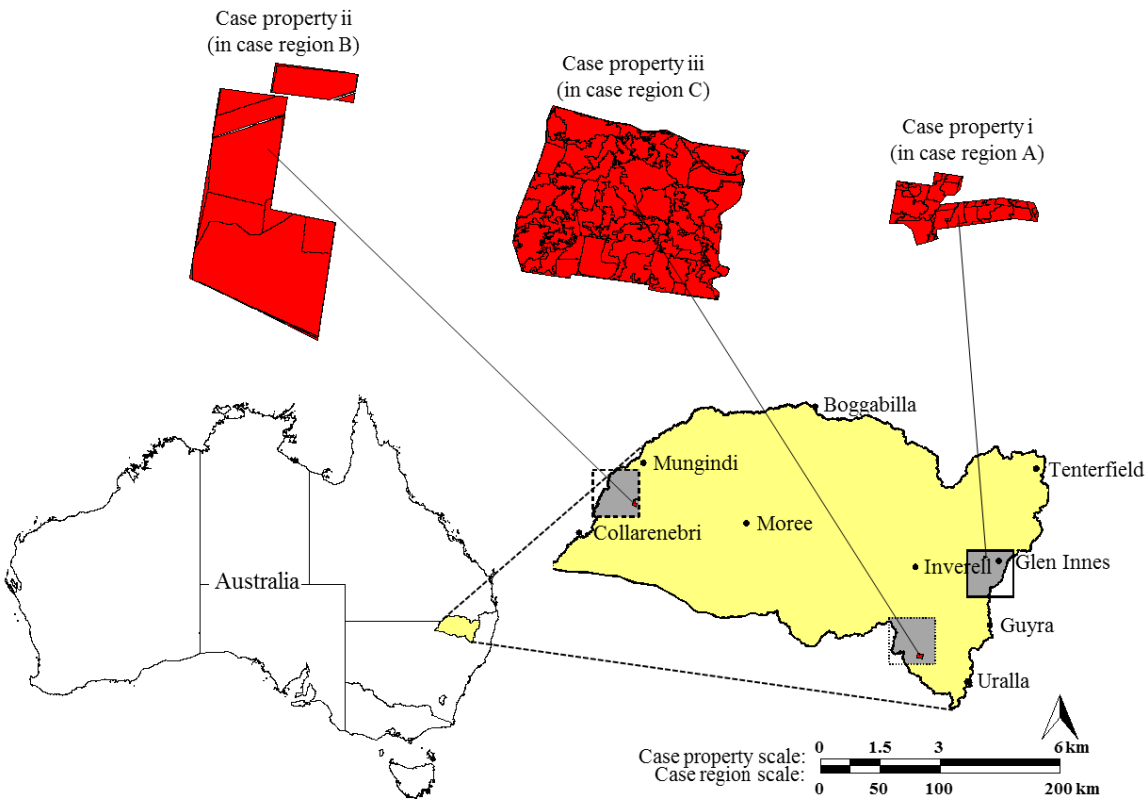

Figure 5.2: Location of the case regions and case study properties in the Border Rivers-Gwydir catchment.

General characteristics of the different case properties for the detailed analysis and the case regions to determine the feasibility of carbon offset projects are shown in Table 5.2.

Table 5.2: General characteristics of the case study properties and regions.

	Case study property			Case study region		
	i	**ii**	**iii**	**A**	**B**	**C**
Case region size (ha)	-	-	-	128,199	166,446	180,360
Properties (no)	-	-	-	328	46	154
Size of property (ha)	320	1,471	1,949	391 ± 544#	3,618 ± 7,186#	1,171 ± 1,841#
Paddocks per property (no)	25	15	90	27 ± 27#	107 ± 184#	51 ± 80#
Eligible paddocks per property (no)	24	15	75	10 ± 4#	45 ± 86#	16 ± 5#
Average size of paddock (ha)	13 ± 10#	98 ± 172#	22 ± 20#	14 ± 15#	58 ± 121#	25 ± 25#

#Values are in terms of mean ± standard deviation.

5.2.3 Feasibility to the individual landholder

The individual landholders will incur a range of abatement costs, including the opportunity costs of foregone income associated with procuring carbon offsets through the proposed carbon sequestration scenarios. Unless one of the proposed tree-planting scenarios provides a higher benefit than the next best land use[x], incentives will be required to encourage landholders to change land use. The present value of the revenue received by an individual landholder for joining a carbon offset scheme is the sum of the discounted farm-gate carbon payments:

$$v_C = \sum_{t=1}^{T} \sum_{i=1}^{i_n} p_F C_i(t)(1+\delta_S)^{-t} \tag{5.2}$$

where v_C is the present value of revenue received from selling carbon offsets to the project developer, i_n is the number of paddocks that landholder n will convert to a carbon-offset producing land use and δ_S is the discount rate for the landholders.

The cost of abatement for the individual landholder is determined using the opportunity cost of switching land use.

[x] If, in the absence of additional incentives, a proposed land use is more beneficial than the current land use, this would not generally meet the additionality requirement of carbon offset projects and would not usually be eligible for crediting (Macintosh & Waugh, 2012).

$$v_A = \sum_{t=1}^{T} \sum_{i=1}^{i_n} R_i(t)(1+\delta_S)^{-t} \tag{5.3}$$

where $R_i(t)$ is the flow of differences between the net revenues of the best alternative land use and the carbon-offset scheme in the *i*-th paddock. The opportunity cost of each eligible paddock is determined based on the current land-use type and the technique and parameter values described in Chapter 4. A sample gross margin for a sheep enterprise along with the cropping parameters used to determine the opportunity cost is shown in Appendix D. Planting cost assumptions necessary for estimation of v_A, along with 17 livestock enterprise gross margins, are included in the supplementary material CD-ROM.

When an agroforestry planting configuration is assumed (i.e. row or boundary plantings) benefits from agricultural enterprises are still received in the area between the rows or inside the boundary plantings. While the planting of trees can have both positive and negative influence on neighbouring pastures and crops (Donaghy *et al.*, 2010), it was assumed that the positives such as increased livestock survival rates and increased crop yields would outweigh any negative impacts. As such, the current model assumed no impact of tree belts on adjacent pastures and crops.

A requirement of the CFI is that paddocks planted to trees for carbon offset production be excluded from stock for the first three years. In block planting configurations it is assumed that the current paddock configuration will have adequate stock proof fences. For the agroforestry options, two different options are assessed: (1) stock are excluded from the entire paddock for the first three years following the planting of trees and (2) the perimeters of the rows are fenced with temporary fencing to allow livestock grazing in the inter-row alleys during the first three years after planting.

When assuming a permanent accounting rule with the flexibility to harvest trees before the 100-year requirement, two different carbon accounting techniques were tested: (1) the purchase market where carbon offsets are sold in year t for the additional amount (minus the risk reversal buffer) of carbon sequestered that year [$C(t)$], and (2) the rental market where the price of carbon is adjusted relative to the permanent carbon offset price based on a discount rate. In the purchase case, the landholder must purchase credits from the carbon market to offset the CO_2 released at the time of harvest. In the rental market, the carbon trajectory is measured as the total stock of biomass attributable to the carbon offset project at the end of year t. If we assume the aggregators discount rate (δ_B) is equal to the discount rate of the participants in the carbon offset market, the carbon rental price is simply the market price of carbon multiplied by this discount rate.

The discounted sum of the annual stream of transaction costs for individual landholders joining a carbon offset scheme is determined using equation (5.4):

$$v_T = \left[w_{S1} p_L + w_{S2} d_{\min} p_{trav} + w_A + \sum_{t=1}^{T} \left(w_{P1} p_L + w_{P2} d_{\min} p_{trav} + w_E \right) \left(1 + \delta_S\right)^{-t} \right] \qquad (5.4)$$

where p_L is the opportunity cost of the landholder time, $d_{\min}$ is the minimum distance to the nearest town (estimated using the least-cost algorithm described in Chapter 3), p_{trav} is the cost of travel per kilometre, and the letter subscripts for the individual landholder transaction costs (w) are as follows: (S) is search and negotiation costs, (P) is project management costs, (E) is enforcement and insurance costs and (A) is approval costs. The number subscripts and the parameter values used are provided in Chapter 4 and are also presented in Appendix E.

5.2.4 Feasibility of projects

The study of the land-use scenario which minimises carbon sequestration potential to the case study properties was extended to the case-region scale used in Chapter 4 to determine the feasibility of carbon offset projects to not only the individual landholder but also to the project developer. The project developer will incur both abatement and transaction costs in the procurement and aggregation of contracts from eligible landholders. Abatement costs to the project developer (V_A) are the sum of the farm-gate carbon payments that must be paid to individual landholders and can be determined as:

$$V_A = \sum_{t=1}^{T}\sum_{n=1}^{N}\sum_{i=1}^{i_n} p_F C_{n,i}(t)(1+\delta_B)^{-t} \tag{5.5}$$

where n is the number of landholders and δ_B is the aggregators discount rate. The discounted aggregator transaction costs are estimated as:

$$V_T = W_{S1} + W_A + W_{P1} + nW_{S2} + \sum_{t=1}^{T}\left[W_{P2} + W_{M1} + nW_{E2} + (W_{M2} + W_{E1})C(t)p_C\right](1+\delta_B)^{-t} \tag{5.6}$$

where the letter and number subscripts of (W) are adapted from the transaction costs notation used in Cacho *et al.* (2013) and Chapter 4. The letters represent the different cost categories and number subscripts refer to costs which are measured using different units within each individual transaction cost category. A list of these transaction costs is provided in Appendix E.

When accounting for their abatement and transaction costs, a project developer would only be willing to pay individual landholders values up to p_F:

$$p_F \le p_C - \frac{V_T(n, a, \mathbf{W}, C(t), \delta_B)}{\sum_{t=1}^{T}\sum_{n=1}^{N}\sum_{i=1}^{i_n} C_{n,i}(t)(1+\delta_B)^{-t}} \tag{5.7}$$

where V_T is a function of the number of landholders, the area obtained, the different transaction costs to the project developer ($\boldsymbol{W}$), quantity of carbon sequestered and the

project developers discount rate. The minimum feasible farm price for an individual landholder is determined by the following equation[y]:

$$p_F \geq \frac{v_T + \sum_{t=1}^{T}\sum_{i=1}^{i_n} R(t)(1+\delta_S)^{-t}}{\sum_{t=1}^{T}\sum_{i=1}^{i_n} C(t)(1+\delta_S)^{-t}} \tag{5.8}$$

where the numerator is the total cost to the landholder, which includes the transaction and abatement costs, and the denominator is the discounted carbon offsets produced.

Project feasibility is determined by the ability of the aggregator to fully cover the cost of the farm-gate carbon payments (p_F) for any given price of carbon in the market (p_C). The minimum project size can be determined by setting equation (5.7) equal to (5.8). This produces the project feasibility frontiers introduced in Chapter 4. The PFF can be expressed as:

$$x_{\min}\left(p_C \mid a, w, W, C(t), R(t), \delta_B, \delta_S\right) \tag{5.9}$$

where this function represents the minimum project size ($x_{\min}$) that is feasible as a function of the market price of carbon for the given values of the other parameters. When assessing a finite area with heterogeneous landholders, an upper bound ($x_{\max}$) can also be determined. This concept was introduced in Chapter 4 and can be expressed as:

$$x_{\max}\left(p_C \mid a, w, W, C(t), R(t), \delta_B, \delta_S\right) \tag{5.10}$$

The area between $x_{\min}$ and $x_{\max}$ represents the feasible project range and these variables can represent the minimum number of landholders, minimum total area or minimum total discounted carbon.

[y] The irreversibility of conversion to forests may discourage landholders from undertaking a land-use change that is an otherwise profitable decision. This is not explicitly captured in this equation.

5.2.5 Sensitivity analysis

A great deal of uncertainty surrounds the future market price of carbon. The use of PFFs allows assessment of project viability at a range of different market prices. Further sensitivity analyses were conducted to determine the sensitivity of the final case region results to changes in several key assumptions. The elasticities in terms of percentage change of the minimum feasible project area were estimated for a range of variables and are reported in section 5.3.4.

5.3 Results

5.3.1 Carbon sequestration potential

Detailed simulations of the case study properties found the average quantity of carbon sequestered over the life of a project to be highly dependent on both the choice of species and management regime employed (Figure 5.3). While the carbon sequestration potential for the mixed-species forests remains relatively unchanged across the four different planting densities, all other species assessed exhibit (often large) fluctuations in average carbon sequestration potential with different management regimes.

With the exception of regime (n), which is the planting of a generic eucalypt species for hardwood and pulpwood production under a very short (12 year) planting to clear-fell harvesting cycle, all other regimes provide a higher average carbon sequestration potential when compared to the mixed-species planting which is eligible for crediting under the current CFI rules. These increases in average carbon sequestration potential range between 5.99% and 120.12%. Overall, *E. cladocalyx* plantations for hardwood production provide the highest carbon sequestration potential, followed by the generic eucalypt plantations. Plantations of *P. radiata* for softwood and pulpwood production sequester only marginally higher levels of average carbon (an increase of between 5.99%

and 21.69%) compared to the mixed-species regimes. This is attributable to the frequent thinnings during the plantation cycle. Not surprisingly, carbon trajectories also vary temporally. This is particularly marked when combined with harvesting regimes. Appendix F contains a sample of the carbon trajectories through time for each case study property, with all carbon trajectories provided on the supplementary material CD-ROM.

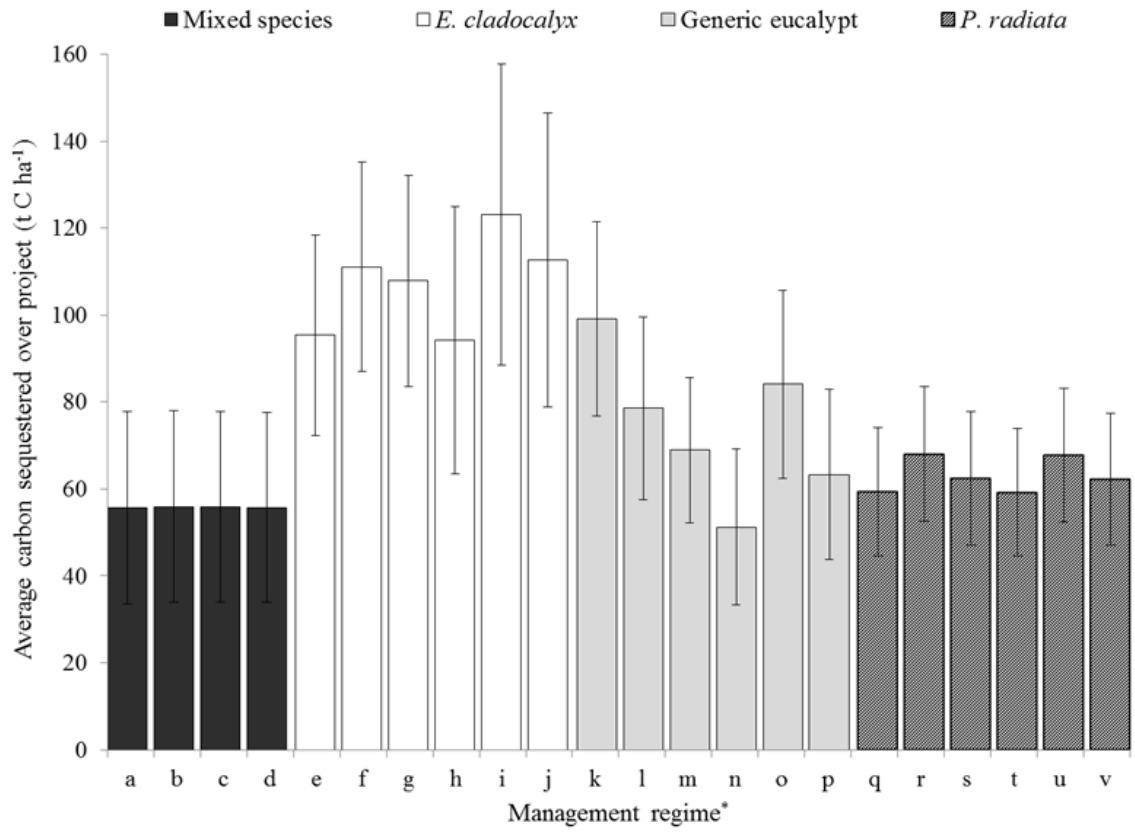

Figure 5.3: Comparison of average carbon sequestration potential of each species and management regime over the 100-year project life. Error bars indicate one standard deviation from mean. *Management regime letters correspond to the regimes presented in Table 5.1.

These findings differ from the sequestration potential estimated in Chapter 2, which estimated that *E. cladocalyx* would have the lowest carbon sequestration potential of the four farm-forestry land uses assessed. The FullCAM model used in that chapter estimated this species would only sequester 10% of the carbon compared to that of an identical area of mixed-species plantings. In contrast, the Reforestation Modelling Tool used in this chapter estimated *E. cladocalyx* to sequester an average of between 69% and

121% higher carbon relative to a mixed-species plantation, with differences caused by management regime. This highlights the importance of using accurate modelling techniques to estimate sequestered carbon, ensuring that the credits claimed are actually produced.

5.3.2 Optimal landholder strategies

The minimum farm-gate price required to encourage landholders to adopt different types of plantations was found to vary significantly depending on the tree species and management regime scenario implemented (Figure 5.4). With the exception of two *P. radiata* regimes, all farm forestry scenarios with harvestable products have a lower average cost per tonne of carbon sequestration compared to the mixed-species forests.

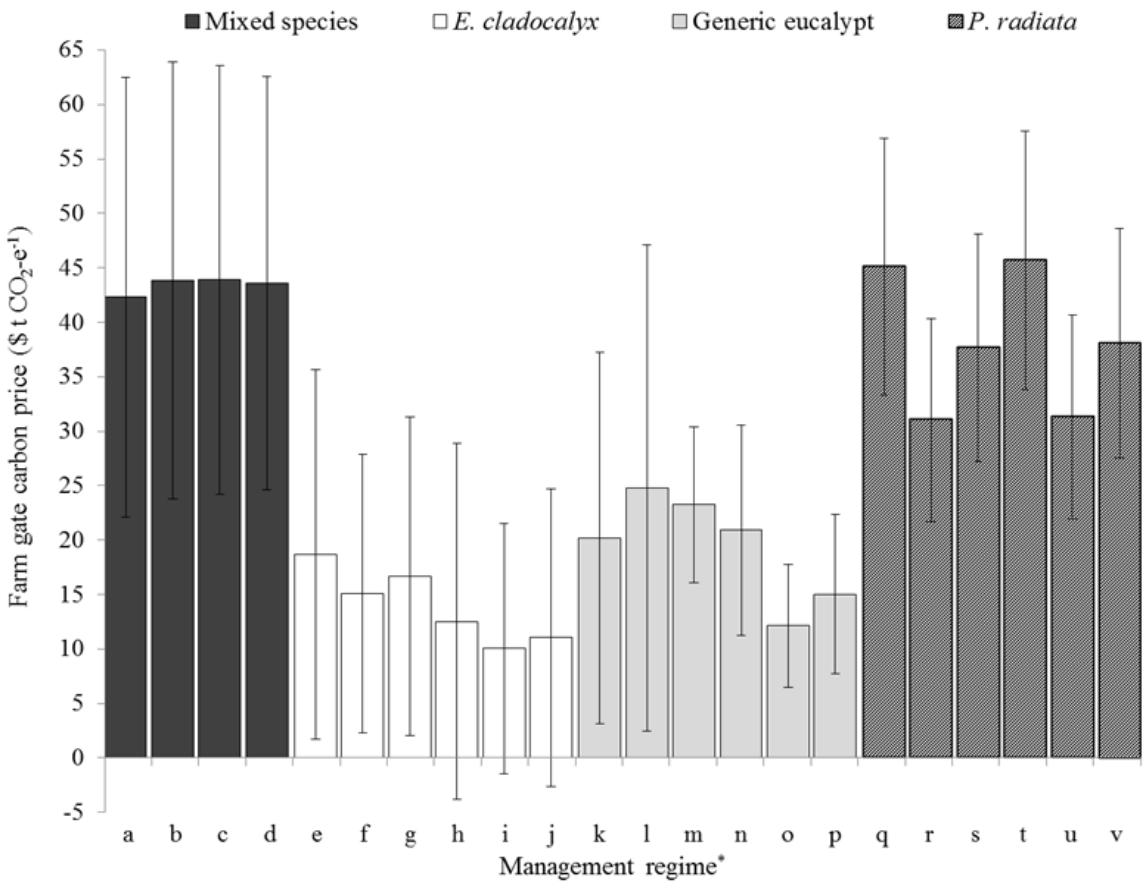

Figure 5.4: Average farm-gate price required for project to be feasible to the individual landholder (before considering transaction costs). Error bars indicate one standard deviation from mean. *Management regime letters correspond to the regimes presented in Table 5.1.

Table 5.3 lists the optimal tree species, management strategy and planting configurations for the case study properties. Interestingly, across these three properties, no individual management strategy or planting configuration exclusively provided the minimum-cost strategy. There is a trend, however, for the monoculture *E. cladocalyx* plantations to provide the lowest-cost option. These findings match those of recent studies in southern Australia (Paul *et al.*, 2013b) and northern Australia (Hunt, 2008) which found monoculture plantations provide the lowest-cost carbon sequestration strategy when compared to native species plantations for biodiversity purposes.

Table 5.3: Optimal scenarios for each of the case study properties.

Tree species	Planting density	Thinned (yr/s)	Pruned (yr/s)	Harvest (yr)	Planting configuration	Fenced	Number of paddocks
Property i							
E. cladocalyx	low	-	-	29	block	NA	13
E. cladocalyx	high	-	-	22	12.2m inter-row	fenced	7
E. cladocalyx	medium	-	-	25	block	NA	2
E. cladocalyx	medium	-	-	25	12.2m inter-row	stock excluded	1
E. cladocalyx	low	-	-	29	24.4m inter-row	fenced	1
Property ii							
E. cladocalyx	low	-	-	29	12.2m inter-row	stock excluded	8
E. cladocalyx	low	-	-	29	24.4m inter-row	stock excluded	7
Property iii							
E. cladocalyx	low	-	-	29	block	NA	60
E. cladocalyx	high	-	-	22	12.2m inter-row	stock excluded	7
Generic eucalypt	medium	-	-	15	24.4m inter-row	stock excluded	7
Generic eucalypt	low	-	-	20	block	NA	1

The majority of paddocks in case study properties i and iii produce the lowest-cost carbon offsets when planted in block geometrics. In contrast, the optimal planting configuration for case study property ii is a row planting system. This trend toward block configurations in properties i and iii can be attributed to the smaller paddock sizes and higher proportion of paddocks containing livestock which must be either completely

excluded for three years following planting of rows or costly temporary fencing installed. In property ii, the major land use is cropping so income is not excluded or fencing costs not incurred during the first three years of row plantings.

By arranging a landholder's paddocks in ascending order of carbon sequestration cost, aggregating the carbon quantities and applying their individual transactions costs according to equation (5.4), it is possible to generate individual carbon supply curves for each landholder. Figure 5.5 provides a comparison of the supply curves for a mixed-species planting under the CFI rules and the optimal carbon sequestration scenarios in each of the case study properties.

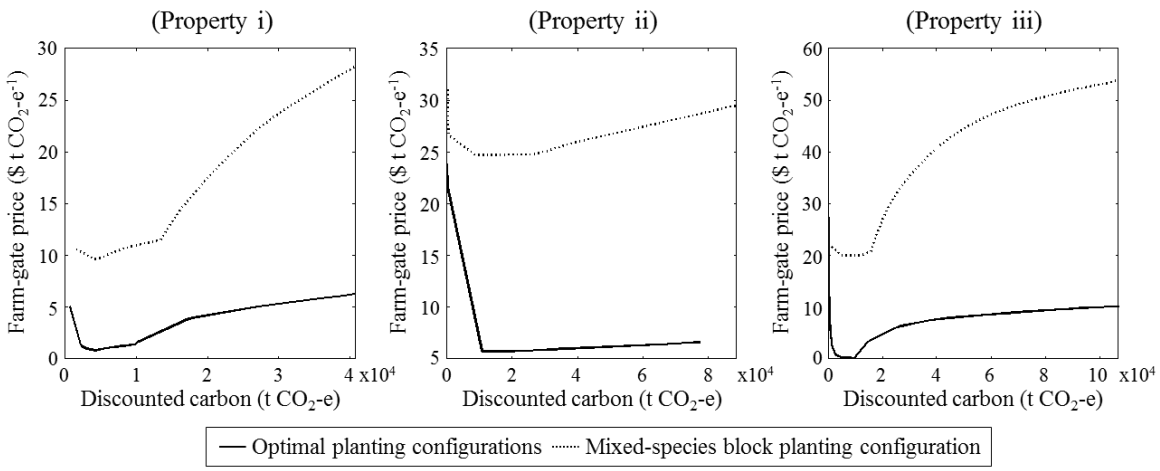

Figure 5.5: Individual landholder supply curves for carbon under the optimal combination of tree species, management regime and planting configuration compared to a mixed-species environmental planting using a block configuration[z] for each case study property.

It can be seen that there is a significant difference between the cost of acquiring carbon offsets from individual landholders under a permanent mixed-species planting and the optimal harvested monoculture scenario. Carbon offsets from property i would cost between \$8.20 and \$19.41 t CO_2-e^{-1} less under the optimal species and management

[z] Mixed-species environmental plantings adopting a block configuration were utilised for all simulations in Chapter 5.

scenario[aa]. Likewise, property ii would be between \$6.77 and \$22.89 t CO_2-e^{-1} cheaper and property iii would be between \$17.66 and \$41.61 t CO_2-e^{-1} lower than the mixed-species regime.

5.3.3 Project feasibility

While the individual landholder supply curves from the three case study properties were demonstrated to be substantially lower under an optimal tree species, management regime and planting configuration compared to the native mixed-species plantations currently approved under the CFI, it is unlikely that these landholders would be able to directly interact with the scheme. As previously mentioned, an aggregator may be required to increase landholder participation. To determine project feasibility, the optimal tree species, management regime and configuration with the average lowest carbon sequestration cost for case properties i, ii and iii were simulated for each property across case regions A, B and C, respectively. In each region, *E. cladocalyx* with a low planting density, no pruning or thinning and a clear-fell harvest 29 years after planting was selected. For case regions A and C, a block planting configuration was assumed and for case region B, a row planting configuration with 12.2m inter-row spacing was adopted.

The project feasibility frontiers for these monoculture *E. cladocalyx* plantations compared with a project comprised of native mixed-species plantations are depicted in Figure 5.6.

[aa] This can be determined from the differences between the required prices at each quantity of carbon in Figure 5.5. For example, to obtain 10,000 t CO_2-e^{-1} from property i, it would cost approximately \$1.50 t CO_2-e^{-1} if the optimal species, management regime and planting configuration is adopted. To obtain the same quantity, however, from a mixed-species under a block planting configuration, would cost approximately \$11.00 t CO_2-e^{-1}; an increase in cost of \$9.50 t CO_2-e^{-1}.

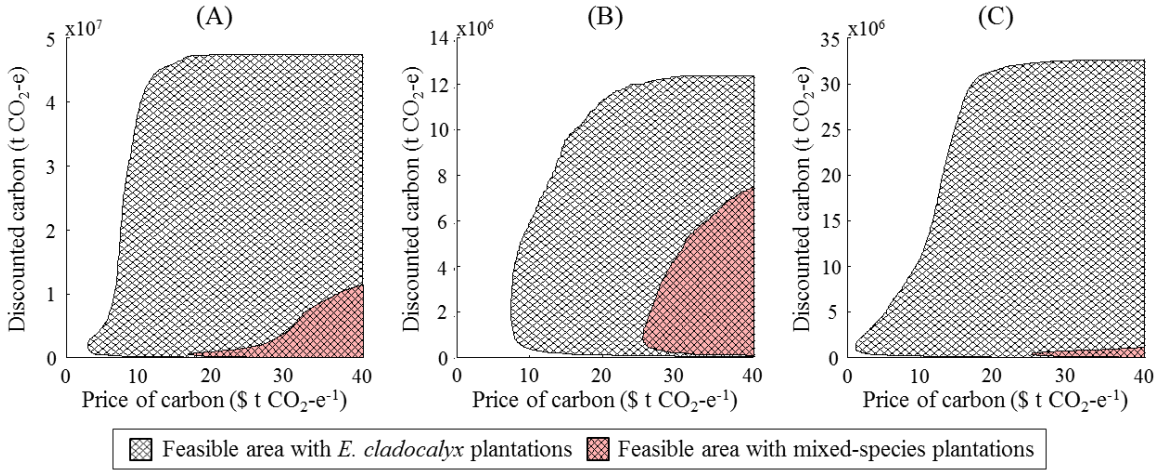

Figure 5.6: Project feasibility areas for the three case study regions, comparing the project feasibility frontiers of a project comprised of *E. cladocalyx* plantations and a project consisting of mixed-species plantings.

The impact of only accepting carbon offsets from a native mixed-species plantation is evident from this figure. The minimum market price required for a project to become feasible decreases from \$16.70 t CO_2-e^{-1} down to \$2.93 t CO_2-e^{-1} under *E. cladocalyx* plantation projects in case region A, an 82.45% decrease. For case regions B and C, the minimum market price decreases by 70.86% and 95.95% when compared to the mixed-species carbon offset plantations. Not only is the market price of carbon at which projects become feasible significantly lower when a monoculture plantation is assumed, the quantity of carbon that can be sequestered at all carbon prices is substantially higher under this monoculture configuration, even when taking into account the carbon assumed to be re-emitted at the time of clear-fell harvesting.

5.3.3.1 Rental versus purchase of carbon offsets

Given the problem of permanence when considering harvested biomass crops, project feasibility under two different carbon accounting techniques was assessed; the use of rental market contracts and the purchase of carbon offsets, where credits for the release of carbon already credited must be purchased when harvesting. Cacho *et al.* (2013) found the use of rental contracts required significantly larger project sizes to become

feasible. Conversely, in the current chapter we found the use of rental contracts reduced the minimum project size and the minimum price required for project feasibility (Figure 5.7). Under a rental scheme, the minimum feasible market price of carbon was between \$4.77 t CO_2-e^{-1} and \$5.45 t CO_2-e^{-1} cheaper than the option of purchasing permanent carbon offsets. In regions A and C, the minimum required price was less than zero, implying that no incentives would be required for some landholders to participate.

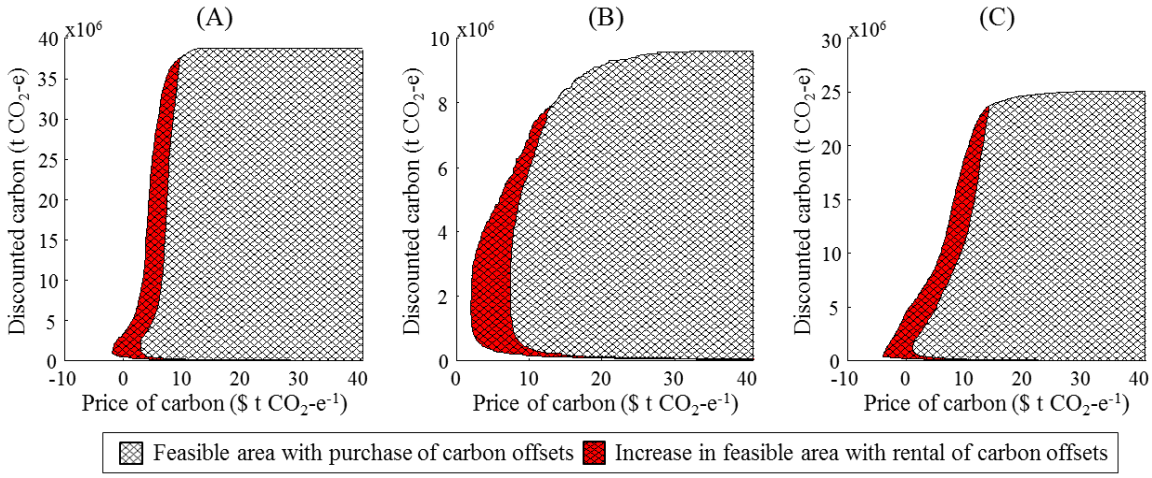

Figure 5.7: Project feasibility for carbon offset projects using *E. cladocalyx* plantations or a purchase versus a rental contract.

The lower cost of carbon under rental contracts can be attributed to the high cost incurred by landholders for purchasing carbon credits in the market under a purchase scheme to offset carbon released at the time of harvest. This effect was not captured in the study by Cacho *et al.* (2013), as it did not include harvested tree plantations.

5.3.3.2 Block versus row planting configurations

The impact of different planting geometries on the feasibility of projects under mixed-species plantings was also assessed. The different ranges of market prices at which carbon offset projects become feasible in each of the case study regions under both block and row planting configurations are shown in Figure 5.8. It can be seen that the total feasible quantity of carbon that can be captured solely using a row planting configuration

of mixed native species is significantly less than that possible if block planting configurations are adopted.

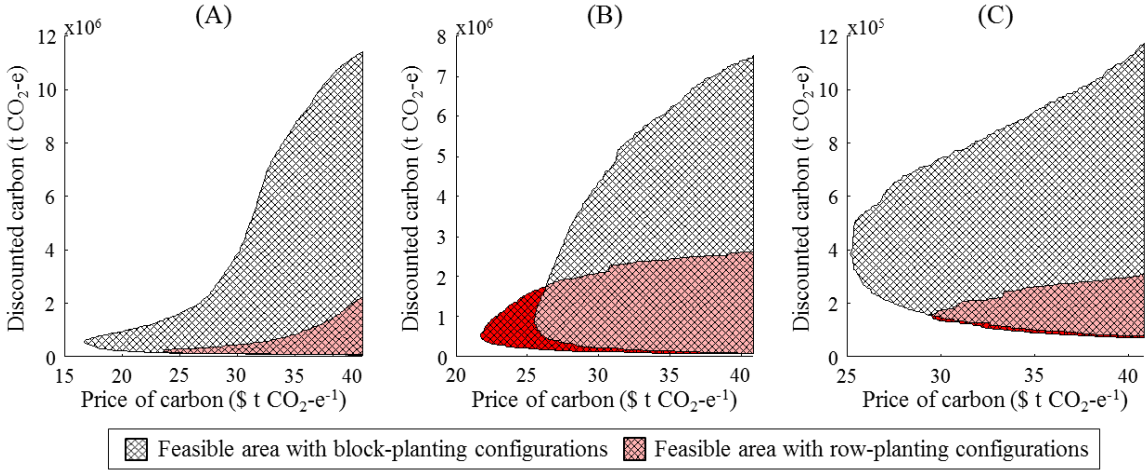

Figure 5.8: Project feasibility areas for the three case study regions, showing the project feasibility frontiers for a mixed-species planting under both a block and row configuration.

Interestingly, the minimum price required for a project to become feasible when adopting a row planting configuration increases in case regions A and C. Under this planting configuration, projects will not become feasible until the market price of carbon is \$23.66 t CO_2-e^{-1} (an increase of 41.6%) and \$29.32 t CO_2-e^{-1} (an increase of 16.5%), respectively. In contrast, the adoption of a row planting configuration reduces the minimum feasible market price to \$21.75 t CO_2-e^{-1} in case region B, a 14.7% decrease. This can be attributed to the smaller average property sizes in case regions A and C not producing enough carbon credits under a row planting configuration to cover the cost of foregone income. Also, the majority of the paddocks in these two regions currently run livestock and would incur the cost of fencing or complete exclusion of stock from existing paddocks in the first three years of the project. On the other hand, case region B has a higher proportion of cropping paddocks which will not require fencing or stock exclusion, allowing additional profit to be made from crops in the first three years of the project.

5.3.4 Sensitivity analysis

As already discussed, the PFF diagrams in Figures 5.6 – 5.8 provide a form of sensitivity analysis in regards to the feasibility of carbon offset projects in relation to the market price of carbon. Sensitivity analyses were also undertaken on a range of additional variables. These are presented as elasticities in terms of percentage change in the minimum project area required in Table 5.4.

Table 5.4: Results from the sensitivity analysis on the three case study regions at a market carbon price of \$4.50 t CO_2-e^{-1}, \$9.00 t CO_2-e^{-1} and \$3.59 t CO_2-e^{-1} for case regions A, B and C, respectively[bb]. Values are in terms of elasticies as per cent change in the minimum feasible project area (ha) in response to a percentage change in each variable/parameter assumption.

	Case region		
	A	**B**	**C**
Landholders discount rate	2.23	1.75	3.04
Market price of offsets when harvesting	1.24	2.77	0.44
Aggregator transaction costs	0.90	0.62	1.40
Cost of planting trees	0.90	2.55	1.14
Harvesting costs	0.55	0.60	0.83
Income from sheep production	0.53	0.67	0.51
Cost of transporting harvested logs	0.45	2.86	0.51
Landholder transaction costs	0.18	0.13	0.00
Aggregators discount rate	0.00	0.00	0.03
Row planting growth modifier (λ)	0.00	-3.58	0.00
Crop yield	0.00	1.23	0.00
Cost of cropping inputs	0.00	-1.44	0.00
Cost of cattle production inputs	0.00	0.01	0.00
Income from cattle production	0.00	-0.01	0.00
Cost of sheep production inputs	-0.27	0.06	0.00
Income from harvested timber	-3.90	-8.23	-4.92

Given that interest lies in the minimum price required for projects to just become feasible, sensitivity analyses were undertaken on the left-hand side of the PFF curves in Figure 5.6. As the minimum feasible price varies between the regions, the price used for

[bb] Different market prices of carbon were assumed for each case region to allow feasible areas to be determined when conducting the sensitivity analysis at a consistent price relative to the minimum feasible price when original parameters are used.

the sensitivity analyses were also undertaken at differing prices. A carbon market price of \$1.57 t CO_2-e^{-1} higher[cc] than the minimum-feasible price was adopted to allow a feasible solution to be found while running the sensitivity analyses. Therefore, the sensitivity analyses were undertaken at carbon prices of \$4.50 t CO_2-e^{-1}, \$9.00 t CO_2-e^{-1} and \$3.59 t CO_2-e^{-1} for case regions A, B and C, respectively.

The additional revenue received from the sale of harvested biomass is a critical assumption for the feasibility of the project. Elasticities ranged from -3.90 and -4.92 in case regions A and C, through to -8.23 for case region B. This implies that the future price of timber and other harvest biomass is of critical significance when considering the viability of projects with harvestable products. Also, the distance to the nearest mill was an important consideration. Leduc *et al.* (2009) pointed out the importance of selecting climate mitigation projects with harvestable biomass in locations where transportation distance is minimised. Landholders in region B have significantly further travel distances to the nearest existing mill compared with regions A and C (refer to Chapter 3). The sensitivity analysis in this chapter reflected this with a high elasticity on the cost of transporting timber from region B (2.86) compared to the lower elasticities in regions A and C (Table 4.8).

As with Chapter 4, the landholders' discount rate was critical to the simulation. Elasticities for this parameter were between 1.75 and 3.04. Interestingly, the discount rate assumption for the aggregator has a negligible impact on the simulations. The choice of discount rate has been discussed in Chapter 4.

[cc] The value of \$1.57 t CO_2-e^{-1} greater than the minimum feasible price for each region was arbitrarily chosen as the price to run the sensitivity analyses because this was high enough for elasticities to be determined on each of the assumptions and resulted in a round figure (\$4.50 t CO_2-e^{-1} and \$9.00 t CO_2-e^{-1}) for two of the regions.

The tree growth multiplier used for the row plantations had a significant impact on the feasibility of a carbon offset project in region B[dd]. The elasticity of -3.58 implies that an increase in this parameter value will result in a decrease in the minimum area required for a feasible project. Paul *et al.* (2013b) reported increased biomass production from row planting configurations of up to twice the value assumed in the current chapter. This implies that the minimum-cost estimates here are conservative, with project feasibility at lower costs possible.

5.4 Discussion and conclusions

5.4.1 Species selection, management regime and planting configuration

The choice of tree species and management regime was found to have a significant impact on the quantity of carbon which may be sequestered and its cost. Generally, *E. cladocalyx* plantations grown for hardwood saw-log production and a generic eucalypt species plantation grown for hardwood sawlog and pulpwood production provided the lowest-cost options for carbon sequestration. The most frequent optimal management regime was to avoid pruning and thinning of plantations until clear-fell harvesting. These findings support those by Daigneault *et al.* (2010) who found that the inclusion of a carbon price encouraged landholders to delay the thinning of forest plantations.

Planting configuration was also found to influence the feasibility of carbon sequestration projects. Paul *et al.* (2013b) found row-planting configurations to be between \$1 and \$21 t CO_2-e^{-1} cheaper when compared to block planting systems. Despite assuming an increased biomasss production of 19% under a row or boundary planting configuration, block planting configurations were found to be more feasible in two of the three case study regions. The choice of planting configuration needs to take into account the size of

[dd] Monoculture *E. cladocalyx* row planting configurations were only simulated in case region B.

the property as a smaller property may not be able to produce enough carbon credits from row plantings to offset the transaction costs. The land use applied between the tree rows is also an important consideration, especially in the initial three years of a project. With livestock enterprises, a block planting configuration often provides the lowest cost option by avoiding the expense of either excluding stock or installing temporary fencing for the first three years after planting. Even in cases where cost of sequestering carbon is lower under row or boundary planting, compared to block planting, the total quantity of carbon obtainable from each landholder will decrease which means that the number of landholders required to capture a given amount of carbon will be significantly higher. This will increase the transaction costs of the project developer.

5.4.2 Issues of permanence

Carbon sequestration projects under the CFI are currently limited by the 100-year permanence rule (Macintosh, 2013). Different mechanisms exist which allow for the trading of carbon credits for projects which do not meet this permanence requirement (Kim *et al.*, 2008). Mechanisms already exist in the Kyoto Protocol's Clean Development Mechanism (CDM) to allow for the trading of temporary carbon credits using a rental system (Galinato *et al.*, 2011). Macintosh (2013) comprehensively argued against the inclusion of the current permanence rule. Results in this chapter show that a rental system under the CFI would increase the adoption of carbon sequestration above what could be expected under the rigid 100-year rule on permanence. There is also a significant decrease in the required market price for projects to become feasible. Given the current trend for a decrease in the global price of carbon, the perceived benefits of including this restriction in the legislation should be reassessed by policy makers.

5.4.3 Generation of co-benefits

The production of offsets may provide several by-products. In particular, carbon forests may produce increased conservation of biodiversity, improvements to soil, hydrological and environmental systems through reduced run-off and increased regional employment (Shaikh *et al.*, 2007; Venter *et al.*, 2009). These products are known as co-benefits (Plantinga & Wu, 2003) and may result in additional benefits or costs with the production of offsets. While projects generating positive co-benefits have the potential to increase the appeal of these schemes (Plantinga & Wu, 2003; Shaikh *et al.*, 2007; Bollen *et al.*, 2009), poor design of offset policies can lead to the generation of negative impacts, known as perverse effects (Jackson *et al.*, 2005; Macintosh & Waugh, 2012; Bradshaw *et al.*, 2013).

5.4.3.1 Perverse effects

There was considerable concern expressed over the potentially adverse impacts from forestry offset projects during the consultation period of the CFI. Issues raised included the potential increased fertiliser use, altered fire regimes and increased hydrological pressures from plantations (Macintosh & Waugh, 2012). A major concern was the impact of plantations on water flows which have been reported to direct yields away from agriculture, especially in watershed or groundwater areas (Jackson *et al.*, 2005). However, in a review of plantations, Vanclay (2009) argued that several of the water use implications raised in the literature may be controlled through design and management techniques. A study on the potential for carbon plantations on farmland in the Murray-Darling Basin, Australia, found that the supplementary irrigation and run-off reductions per catchment would use less water than most conventional irrigation activities (Schrobback *et al.*, 2011). Although these authors pointed out that under drought conditions, the expansion of forestry would result in increased water use. Townsend *et*

al. (2012) demonstrated that plantations motivated by carbon sequestration payments may provide improved water quality benefits in areas with salinity issues. Therefore, the impact on water flows is an important consideration for policy makers considering project design and should be addressed in terms of both potential benefits and perverse effects.

5.4.3.2 Biodiversity co-benefits

The capture of biodiversity conservation products as a by-product of carbon offset projects has received considerable attention in the climate mitigation policy sphere (Hunt, 2008; Venter *et al.*, 2009; Harvey *et al.*, 2010; van Oosterzee, 2012; Bradshaw *et al.*, 2013). This desire to acquire biodiversity co-benefits from carbon sequestration is explicitly stated in the CFI legislation. The current framework used to assess potential new methodologies in the CFI actively encourages the provision of co-benefits, such as biodiversity increase, through the allowance in the CFI Act for these attributes to be noted on the Register of Offset Projects (Australian Government, 2011a, p. 194). While there is little doubt that significant biodiversity benefits may be achieved through mixed native-species plantings for carbon sequestration, there is a consensus that an extra premium will be required for these projects (Hunt, 2008; Kanowski & Catterall, 2010; Crossman *et al.*, 2011; Paul *et al.*, 2013b). Crossman *et al.* (2011), in a study of ecological and monoculture plantings in southern Australia, estimated that this additional premium would be relatively small if the monoculture plantations were not harvested. The findings in the current chapter suggest that the additional incentives required for a non-harvested mixed native species planting compared to harvested monoculture plantations in northern New South Wales can be substantial. The carbon price required for carbon sequestration could be reduced by between 70.86% and 95.95% across the case regions if harvestable biomass that increases the average stock of carbon is allowed.

The PFF diagrams derived in this chapter show the level of incentives required to capture biodiversity co-benefits as part of carbon projects may be prohibitive. Therefore, policy makers need to weigh up the trade-offs between the priority given to additional biodiversity benefits received from conservation plantings and the higher quantity and cheaper carbon sequestration potential of monoculture plantations. The findings in this chapter suggest that there may be occasions when it is beneficial to uncouple carbon sequestration and biodiversity conservation products when trying to reach Australia's climate mitigation goals. Alternatively, biodiversity payments may be introduced to cover the additional opportunity cost experienced by landholders (Polglase *et al.*, 2011). In a review of agroforestry and the ability to provide carbon sequestration, secondary salinisation abatement and biodiversity conservation in an Australian context, George *et al.* (2012) concluded that placing a title on the biodiversity benefits generated would be beneficial. Schemes offering payments for capture of biodiversity benefits already exist (for example, Stoneham *et al.*, 2003; Moss *et al.*, 2012).

In conclusion, this chapter has shown that there are several different strategies available to increase the viability of carbon sequestration plantings. The legislation of the CFI, however, has restricted most of these beneficial strategies. Therefore, to encourage increased uptake of the CFI, policy makers should continue to assess the feasibility of some of their restrictions[ee].

[ee] Findings in this chapter lends support to the recently introduced allowance for farm forestry under the CFI (Australian Government, 2013) as a low-cost carbon mitigation strategy.

CHAPTER 6: CONCLUSIONS

6.1 Introduction

This chapter presents the main findings of the study in light of the objectives stated in Chapter 1. Limitations and challenges identified during the research are discussed and areas for potential future research are recommended. The chapter concludes with a summary of the main conclusions.

The aims of the study were to determine the:

1. potential supply of carbon offsets from land-use change by Australian farmers;
2. abatement and transaction costs for the supply of carbon offsets and their spatial variability;
3. impact of different techniques for calculating transportation costs for harvested biomass from some land-use options;
4. minimum feasible size of a carbon-offset project that would entice aggregators to act as intermediaries in carbon markets; and
5. optimal species selection, management regime and planting configuration of farm forestry in the presence of carbon markets.

The key findings of the study are summarised below with reference to these aims.

6.2 Overview of the results

6.2.1 Potential supply of carbon offsets

The potential supply of carbon sequestration from a technical perspective was estimated using two carbon accounting models, FullCAM and the Reforestation Modelling Tool, with parameter sets calibrated to the case study catchment. The technical potential of several land-use options was found to exhibit significant spatial variability. Across the 5,000,000 ha catchment, a coefficient of variation of between 35% and 93% was found

for the range of land-use options considered (Chapter 2). The study identified regions within the catchment and those land uses which have the highest technical potential to sequester carbon.

Choice of plantation management regime was found to have a significant influence on the carbon sequestration potential of all tree species considered (Chapter 5). There were large fluctuations in the carbon sequestered by plantations of the same species to which different management practices had been applied. The management practice that maximised profit from carbon offsets did not involve pruning and thinning of plantations until clear-fell harvesting. This result was expected.

The carbon sequestration potential estimated for the *Eucalyptus cladocalyx* plantation differed substantially between the FullCAM model and the Reforestation Modelling Tool, even though both models used parameters specific to the case study catchment. The FullCAM model (Chapter 2) estimated that *E. cladocalyx* had the lowest carbon sequestration potential of the four land-use options considered, capturing only 10% of the carbon sequestered by a mixed-species planting. In comparison, the Reforestation Modelling Tool (Chapter 5) estimated that *E. cladocalyx* would sequester on average 69% to 121% more carbon than the mixed-species plantings, with the difference attributable to management regime. The difference in the carbon sequestration potential estimated by the two models demonstrates the importance of using accurate modelling techniques and parameter sets derived from empirical data of the appropriate scale to ensure that the carbon credits claimed reflect what is actually produced.

Optimal land-use systems to maximise carbon sequestration were found to vary across the catchment (Chapters 2 and 5). No single land-use option provided the highest sequestration potential for the entire catchment; instead a range of land uses comprised

the optimal solution. This highlights the importance of flexible policies for carbon sequestration that allow for a range of land-use options to be adopted.

6.2.2 Abatement and transaction costs for the supply of carbon offsets

Carbon sequestration costs were determined using two techniques of increasing complexity. Initially, supply (abatement) costs were estimated in terms of opportunity costs on a 1.1 km^2 grid using regional statistical data, current land uses and satellite data on vegetation greenness status (Chapter 2). This technique allowed for the lowest abatement cost areas to be identified. Results indicated that the land-use system and geographical location with the highest carbon sequestration potential does not necessarily provide the lowest abatement costs per unit of CO_2-e. As with potential carbon sequestration, the spatial distribution of supply costs was highly variable. The average abatement costs were found to vary from -\$283 to \$349 t CO_2-e^{-1} across the catchment. However this technique was implemented at a level of resolution (1.1 km^2) at which landholders would be unlikely to base their land-use decisions. The technique also ignored the time preference for sequestering carbon.

The second technique for estimating abatement costs was applied at the farm scale (Chapter 4). While numerous studies have reported the cost of carbon sequestration from land-use change activities, most Australian studies report broad-scale estimates with generalised management regimes, without accounting for heterogeneity at a local or farm level. The current farm structure and the extent of existing trees on a farm are commonly overlooked when estimating abatement costs. A key contribution of this thesis is the estimation of localised abatement costs using this second (farm-scale) technique. Cadastral information and satellite data on historical cropping and current paddock woodiness were combined with regional agricultural production and gross

margin averages to estimate carbon sequestration costs at a paddock scale. Transaction costs were also included at a farm level, allowing for variance in the different cost categories. The time preference of carbon sequestration was captured with this technique. It was possible to determine both the farm-gate price and the quantity of offsets required for an offset scheme to appeal to landholders. As with the first simpler technique (Chapter 2), significant spatial variability was found in the cost of carbon offsets, at a scale that has not been captured in most previous studies.

This study developed a model to estimate project feasibility at the farm-scale and the results highlight that ignoring this level of detail can lead to a significant overestimation of carbon-sequestration costs. The coefficient of variation for the minimum feasible farm-gate carbon price to landholders across and within the three regions of the catchment that were assessed was 46%.

Numerous studies have suggested that aggregators have a role to play in reducing transaction costs and other barriers that inhibit landholders' participation in carbon markets. In Australia to date, the number of project developers has been low partly due to a lack of interest by landholders to participate in carbon markets. Results from this study suggest that when project developers act as intermediaries to aggregate offsets from landholders, the transaction costs incurred by landholders are not barriers to them undertaking land-use changes for financial reward. Transaction costs incurred by project developers critically determine whether they will engage in the carbon market.

6.2.3 Estimation of transportation costs

Cost associated with the transportation of biomass to processing plants was critical to the viability of land-use changes involving harvested biomass. Transportation costs are determined by the distance from the biomass source to the processing destination and the

physical aspects of the road. The techniques available to estimate travel distances and transportation costs vary considerably in terms of complexity, accuracy and processing time. These techniques were evaluated in Chapter 3.

Travel distances from source locations to the nearest mill were determined across the case study catchment using three travel distance estimation methods, the Orthodromic ('as the crow flies'), Dijkstra and least-cost algorithms. The Orthodromic algorithm provides a straightforward calculation technique requiring minimal data; only the geographical coordinates of the source and destination were required. It was found that there was a significant trade-off between accuracy and complexity and caution needs to be taken when applying this technique instead of the more data intensive Dijkstra and least-cost algorithms. The Orthodromic algorithm underestimated transportation distance by an average of 41.32% ($\pm$ 16.40%) when compared to estimates using road network data. This translated into an average underestimate of total project costs of 6.78% to 9.77% for the different land-use systems analysed.

Travel distance can also influence the net quantity of creditable project offsets. Production of biomass locks up carbon, reducing atmospheric carbon, but transporting harvested biomass increases atmospheric carbon by burning fossil fuels. This means that when designing policies and methodologies to determine the carbon sequestration by a project, it is imperative that all emissions are captured by the accounting technique. By including geographical data, the current road network, road characteristics and topography, a more accurate estimate of travel distance and the resulting fuel emissions will be achieved. The findings in Chapter 3 demonstrate that ignoring these features will grossly underestimate transport emission by as much as 41.29% on average.

6.2.4 Minimum feasible size of offset projects

This study improves an existing project feasibility frontier model by introducing landholder heterogeneity to estimate the feasible range of offset projects. The project feasibility frontier model requires the estimation of both the minimum farm-gate price that landholders would be willing to accept to produce offsets and the maximum farm-gate price that project developers would be willing to pay for these offsets.

This method to determine project feasibility frontiers for offset projects from a heterogeneous group of landholders provides a practical tool to evaluate project viability and help design policies. The minimum feasible project size was found to be dependent on the characteristics of the individual landholders and therefore varied between the regions considered. In the three case study regions, a market price of \$16.70 t CO_2-e^{-1} to \$25.50 t CO_2-e^{-1} was required for a mixed-species planting project to be established. When accounting for the additional benefits on neighbouring paddocks, the minimum feasible price decreased by 1.63% to 4.28%. The required market price was even lower when different species, management regimes and planting geometries were considered.

In Chapter 5, the accounting rules of a carbon market had significant implications for the minimum project size and minimum price required for a project to be feasible. Offset projects based on rental prices became feasible at costs between \$4.77 t CO_2-e^{-1} and \$5.45 t CO_2-e^{-1} less than for projects based on permanent purchase prices. This suggests that rental markets should be considered when designing future Australian carbon markets.

6.2.5 Optimal species selection, management regime and planting configuration

Minimum project size was sensitive to the choice of land-use system, management regime and carbon accounting rules employed (Chapter 5). The lowest-cost strategy in

each region comprised different monoculture systems, involving two species, and several management regimes. Monoculture plantations returned the most offsets at the lowest cost compared to mixed-species plantings, which did not feature in these solutions.

A stated objective of current Australian policy is to achieve biodiversity co-benefits from carbon sequestration land uses. This objective was found to add a significant cost to the acquisition of carbon offsets in the case study catchment. Simulation results indicate that the price required for landholders and project developers to engage in carbon offset projects would be reduced by between 71% and 96% with harvested monoculture plantations compared to permanent native plantings for environmental purposes. This finding suggests that current policy to achieve environmental co-benefits inhibits the adoption of carbon mitigation activities. This raises an argument in favour of separating biodiversity from carbon payments.

Planting configuration was also found to influence the viability of carbon offset projects. In contrast to other recent studies, which have found that row planting configurations produce carbon offsets from \$1 to \$21 t CO_2-e^{-1} cheaper than block plantings (for example, Paul *et al.*, 2013b), this study found that the difference was not so clear. Benefits from different planting configurations are influenced by several factors. Transaction costs, especially fixed costs, reduced the benefits of row plantings, particularly on smaller farms with lower sequestration potential. If the current land use is a livestock system then the financial viability of row plantings is reduced because of the need to exclude livestock from young trees.

6.3 Limitations of the study

The results obtained from the bioeconomic models developed in this study are reliant on the accuracy of the parameter sets used in the biophysical models, the assumptions

behind those models, and the input and output prices used in the economic models. The biophysical models, FullCAM and the Reforestation Modelling Tool, used parameter values from the literature. As mentioned in section 6.2.1, there was a significant divergence in the estimated sequestration potential of *E. cladocalyx* plantations between these two models. This highlights the need to evaluate the reliability of these models using empirical data at a localised level.

Given the data intensive spatio-temporal nature of this study, a deterministic modelling approach was used. Uncertainty about carbon prices, resource requirements, crop and pasture yields and input and output prices would influence the results of this study. The use of expected values does not capture the full realm of possible future outcomes, but it does provide valuable approximations of project feasibility. The project feasibility frontier methodology encompasses the impact of carbon prices and sensitivity analysis identifies critical variables and parameters.

Carbon supply (abatement cost) curves were derived by assuming that landholders only require payments to cover their opportunity and transaction costs, but this is minimum bound. In reality, landholders may require additional rent for assuming the risk of a carbon project. In addition, abatement cost curves assume that landholders are motivated only by profit. However, landholders do have objectives other than maximising profit. In a review of cost estimation methodologies for carbon sequestration from afforestation, Dempsey *et al*. (2010) found that, due to their ability to capture unobservable factors such as irreversibility, cost of knowledge and non-market benefits, cost estimates derived from econometric models are generally higher than bottom-up approaches, such as the techniques used in this study, and sectorial optimisation models. Despite this finding, in a study of American farmers by Lubowski *et al*. (2006), the marginal costs of management estimated using an econometric model were lower than those estimated

with sectoral optimisation models (Sohngen and Mendelsohn, 2003). Shaikh *et al.* (2007) also found that non-market benefits may reduce the incentives required for farmers to participate in environmental markets below their net returns from current agricultural activities on marginal agricultural land. Using contingent valuation to measure willingness to accept, Shaikh *et al.* (2007) estimated that auction mechanisms can result in cost savings of between one-third and two-thirds for afforestation projects on private landholders' farms.

The analysis of project feasibility in Chapters 4 and 5 assumed that all farmers would join a carbon sequestration project in year zero. Such coordination is unlikely to occur, so the influence of gradual growth in the number of landholders participating in a project should be considered in future work. There has been relatively low uptake of carbon sequestration projects by Australian landholders to date (Mitchell *et al.*, 2012).

The analyses in this study made no allowance for variability in tree, crop and pasture yields in response to a changing climate. Simulations of the carbon sequestration potential of a number of land-use systems were undertaken using the FullCAM model for a range of future climate scenarios (Chapter 2). Changes in output across the entire case study catchment were insignificant. It is not clear whether this result reflects possible deficiencies in the models or the scenarios tested and is worthy of future research.

Howden and Jones (2001) studied wheat crop changes in northern NSW in response to a changing climate and found that it was likely to have a beneficial effect. Sensitivity analysis conducted in Chapter 4 returned an elasticity of 0.47 for minimum project size with respect to crop yield for those regions within the catchment where cropping is currently the main enterprise. This suggests that viable project costs are underestimated by ignoring future climate.

When estimating the impact of transportation costs on the feasibility of carbon offset projects, only the distances from the biomass sources to existing processing mills were considered. With increased demand for biomass for biofuel production, it is possible that additional processing mills may be built, affecting transportation cost estimates. The techniques applied in this study could be modified to identify ideal locations for future processing mills.

A final limitation is that the study assumes that a property boundary in the farm-scale models delineates a unique farm, and that each farmer sells their carbon offsets to the project aggregator. In reality, several farms may be owned by the one entity (company or farmer), which may reduce total transaction costs through economies of scale achieved by an individual farmer selling offsets from multiple farms.

6.4 Suggestions for future research

This study has identified a number of gaps in knowledge and several limitations have been discussed above, which would be worthy of future research.

It would be possible to incorporate stochasticity in the model developed in Chapters 4 and 5, through probability distributions that represent the inherent variability of both the biophysical and economic parameters. This would require the optimisation approach to be revised, because in its current form, the model would require many months to solve a single scenario under stochastic conditions.

Research into the impact of a changing climate on the biophysical and agricultural features of the catchment is required. Several models already exist to estimate crop and pasture growth under different climate scenarios. For example, APSIM (Keating *et al.*, 2003) could be used to model crops in the catchment under different future climate

scenarios. This would require full down-scaled parameter sets from global climate models for the main cropping enterprises in the catchment, preferably validated by localised empirical data. The AussieGRASS or GrassGro models could also be adopted to provide estimates of future pasture growth and livestock performance.

There is scope for further studies to assess the feasibility of different climate mitigation policies if new processing mills were to be strategically built in target locations. The use of the least-cost travel distance estimation method described in Chapter 3 would allow the least-cost distance from biomass sources to mills to be estimated while accounting for the existing road network, road classes and formation, and topographical features. Travel distance is a critical factor in the feasibility of carbon-offset projects and the least-cost method used in this study could be extended to determine the optimal location of new mills.

The single-objective profit-maximisation function could be replaced in the models developed in this thesis by a utility function for landholders. This would require risk attitude and other behavioural characteristics of landholders to be considered.

Additional research into the impact of trees on neighbouring paddocks is required to improve the reliability of these estimates. Additional benefits to neighbouring paddocks reduce the cost of procuring offsets from a landholder, provided the paddocks are owned by the landholder. Assumptions regarding the additional benefits are critical to the assessment of project feasibility. The relationship between different tree species and adjacent crops or pastures is specific to a number of factors including geographical location, species composition and age of tree. More empirical research is required to enhance the reliability of predictions over a wide range of land-use systems.

6.5 Conclusions

This study adds to growing evidence that land-use changes by farmers can provide substantive climate mitigation outcomes, but only under specific conditions regarding carbon price and accounting and eligibility rules. The potential quantity of carbon offsets and their abatement costs were both found to vary at the farm and regional scale due to the heterogeneity of site characteristics, current farming systems and infrastructure and management strategies. Most studies understate this variability due to their broad-scale nature.

Farm forestry configurations that optimise the quantity of carbon offsets procured at a minimal cost were determined. The optimal species and management regimes identified here have the potential to inform government policies. A key message from this study is that monoculture plantations can contribute significant carbon offsets cheaply. Policy makers should weigh up their negative perceptions of monoculture plantations against the estimated cost savings and potential supply of carbon offset associated with these systems.

Transportation costs were found to critically influence the feasibility of land-use changes with harvestable biomass products. While it was tempting to use a technique that requires minimal data to estimate transportation costs, the Orthodromic algorithm was shown to underestimate transportation costs by an average of 41.32% for the case study catchment. This algorithm also results in an underestimate of transport emissions, which would compromise the integrity of carbon sequestration projects. The limitations of this travel distance estimation method must be kept in mind and it is recommended that preferred methods take into account geographical features and road characteristics.

Transaction costs and lack of knowledge of current regulations and reporting requirements inhibit most landholder's ability to directly participate in carbon-offset markets. Project developers who aggregate offsets from individual landholders are often advocated as a solution to increasing the participation of landholder in carbon-offset markets. To date, this type of aggregation activity has been limited in Australia. The models developed in this thesis provide a framework for determining the carbon price, number of individual landholders and quantity of carbon offsets that must be aggregated for a project to become feasible. While these models were tested in a particular catchment setting, they could be applied to other regions and to evaluate other scenarios.