Climate Change in Western Australian Agriculture: a Bioeconomic and Policy Analysis

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Summary

Western Australia's Wheatbelt is one of Australia's major agricultural regions. However, its climate is experiencing a warming and drying trend that is projected to continue. The first aim of this thesis is to investigate the possible bioeconomic impacts of these climatic changes on the region's broadacre agriculture. In recent times Australian politicians have expressed interest in, and enthusiasm about, using agriculture for mitigating climate change, based on a belief that sequestration of carbon in soil and reforestation of farmland could provide cost-effective abatement. The second aim of this thesis is to investigate these claims by examining the potential effectiveness of policies to mitigate climate change in the agricultural sector, using the Wheatbelt region as a case-study.

To achieve the thesis aims, biophysical models were used to simulate, firstly, the impact of different climate scenarios on crop yields, pasture growth and tree growth, and secondly, the impact of different land-management practices on carbon sequestration. The results of these simulations were then incorporated into a whole-farm bioeconomic optimisation model of a mixed cropping-livestock farming system, and a number of analyses conducted. The simultaneous considerations of both the impacts and policy aspects of climate change at the farm-level, as is done in this thesis, is a relatively unique approach.

The impact of climate change on farm profit across a range of scenarios varied between -103% to +56% of current profitability in 2030, and -181% to +76% for 2050; in the majority of scenarios, profitability decreased. If the warming and drying trend predicted for the region translates into either large temperature increases and/or rainfall reductions, then results suggest substantial reductions in profitability. Despite agriculture being a larger emitter of greenhouse gases, a price on farm emissions of greenhouse gases had less effect on profit than (even relatively moderate) climate-change scenarios. Adaptive changes to farm management under more severe climate scenarios included reductions in crop inputs and animal numbers and, to a lesser extent, land-use change. Whilst the benefits of this adaptation were substantial (the financial impact of climate change was 15% to 35% greater without it), profit reductions were still large under adverse climate scenarios even following optimal adaptation. Compared to profit margins, production (e.g., crop yield) was much less sensitive to climate

change. The consequence of this is that relatively minor increases in yields or prices would be sufficient to maintain profitability. However, if these price and/or productivity increases would have occurred regardless of climate change, then the actual cost of climate change may still be high.

The potential for agricultural land in the Wheatbelt region to act as a low-cost carbon sink seem limited, particularly for soil carbon. To incentivise large-scale land-use change to sequester carbon would appear to require a relatively high carbon price (higher than featured in any contemporary policies). Compounding this, from a policy perspective, the characteristics of sequestration make it inherently difficult to cost-effectively deploy as a mitigation option. Even where the profitability of agricultural production was substantially reduced due to the impact of climate change, the financial attractiveness of reforesting farmland did not necessarily increase, because climate change also reduced tree growth, and therefore the income from sequestration.

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Statement of Original Contribution

This thesis contains material published/or prepared for publication. The bibliographical details of this material, its location in the thesis, and the candidate's contribution to this material relative to that of the co-authors are as follows:

Thesis Chapter 2

Thamo, T. and Pannell, D.J. (2016). Challenges in developing effective policy for soil carbon sequestration: perspectives on additionality, leakage, and permanence, *Climate Policy*, 16, 973-992. DOI:10.1080/14693062.2015.1075372

Candidate's contribution: 80%

Thesis Chapter 3

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Thamo, T., Pannell, D.J., Kragt, M.E., Robertson, M.J. and Polyakov, M. (In Press). Dynamics and the economics of carbon sequestration: common oversights and their implications, *Mitigation and Adaptation Strategies for Global Change*.

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Thesis Chapter 7

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Candidate's contribution: 75%

Thesis Appendix: Chapter 9

Barton, L., Thamo, T., Engelbrecht, D. and Biswas, W.K. (2014). Does growing grain legumes or applying lime cost effectively lower greenhouse gas emissions from wheat production in a semi-arid climate?, *Journal of Cleaner Production* 83, 194-203.

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Candidate's contribution: 20%

Co-authors and my co-ordinating supervisor have provided their consent for these works to be included in this thesis.

Candidate's signature

Coordinating supervisor's signature (D.J. Pannell)

Lead-author's signature (M.E. Kragt)

Lead-author's signature (L. Barton)

Chapter 1. Introduction

1.1 Background—the Western Australian Wheatbelt

1.1.1 A major agricultural region of Australia

Situated in the south-western corner of the Australian continental landmass, to the east of the provincial capital Perth (Figure 1.1), is Western Australia's broadacre agricultural region. In common parlance, this region is known as the 'Wheatbelt', as about 40% of the wheat exported by Australia or around 5% of the wheat traded internationally is produced there (ABARES, 2013).

However, wheat growing in fact only accounts for slightly more than 60% of the area cropped in this Wheatbelt. (Planfarm, 2015). Other common crops are canola, barley and lupin. Whilst some farm businesses in the region are purely devoted to crop production, many farms include phases of pasture in their land use, running mixed cropping-livestock farming systems. Overall, approximately three quarters of the region's farm area is cropped (Planfarm, 2015), with remaining pastures being mostly grazed by sheep. The region produces 11% of the wool exported by Australia or 7% of all internationally-traded wool (ABARES, 2013). The total economic value of the region's agricultural production varies with the vagaries of season and market, but typically is around $5 billion in gross value[1] (ABARES, 2015).

The Wheatbelt has a semi-arid Mediterranean climate. As the rainfall isohyets in Figure 1.1 show, annual rainfall decreases from west to east and from south to north across the region, ranging from 280 to 550 mm. From south to north, average temperatures also increase. Consistent with its Mediterranean classification, the climate is distinctly dichotomous: winter is cool and moist, summer is very hot and dry. This is demonstrated in Figure 1.2, which shows the temporal distribution of rainfall and temperature for the township of Cunderdin. The analyses reported in the main chapters of this thesis are based on farms near to Cunderdin in the central area of the Wheatbelt region because, as Figure 1.1 shows, it represents an approximate 'mid-point' in the range of rainfall and temperature experienced across the region.

Its reliance on rainfall, combined with the semi-arid nature of the climate, makes the region's agriculture potentially vulnerable to climatic change, especially to increases in temperature and/or decreases in rainfall.

[1] Unless otherwise indicated all financial values in this thesis are expressed in Australian Dollars

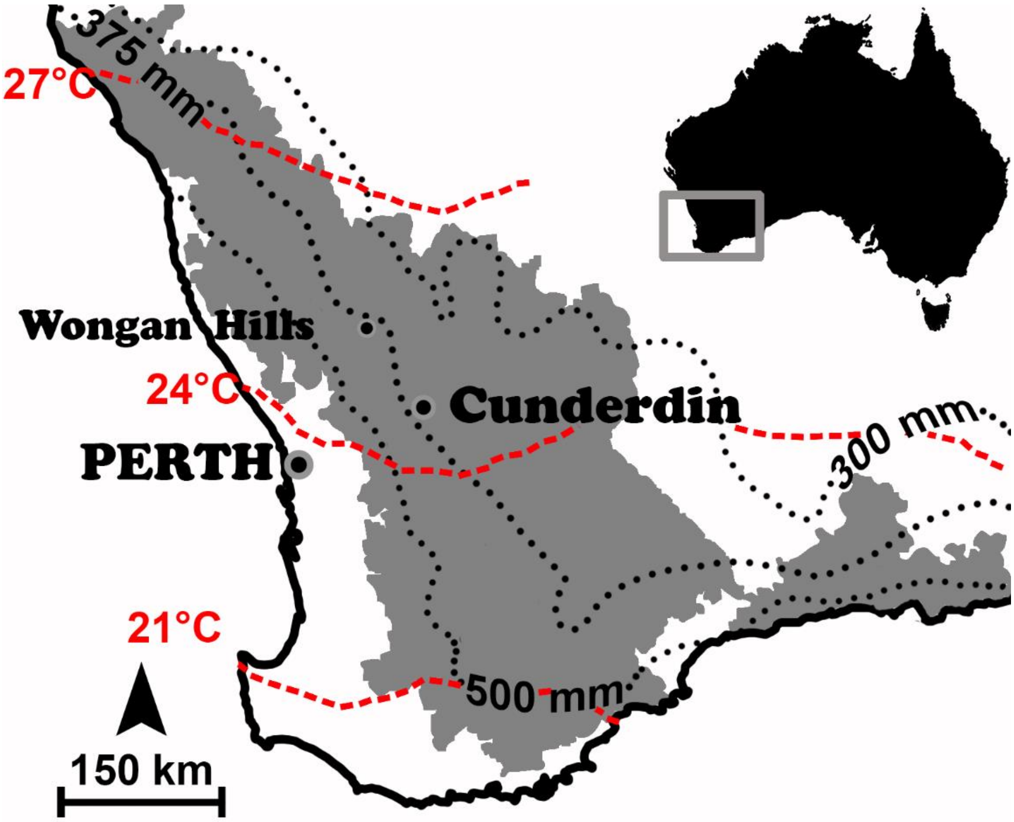

Figure 1.1. The Wheatbelt region of Western Australia (grey shading). Black-dotted isohyets show average rainfall while red-dashed isotherms show the annual average of maximum temperatures (based on data for the years 1961 to 1990).

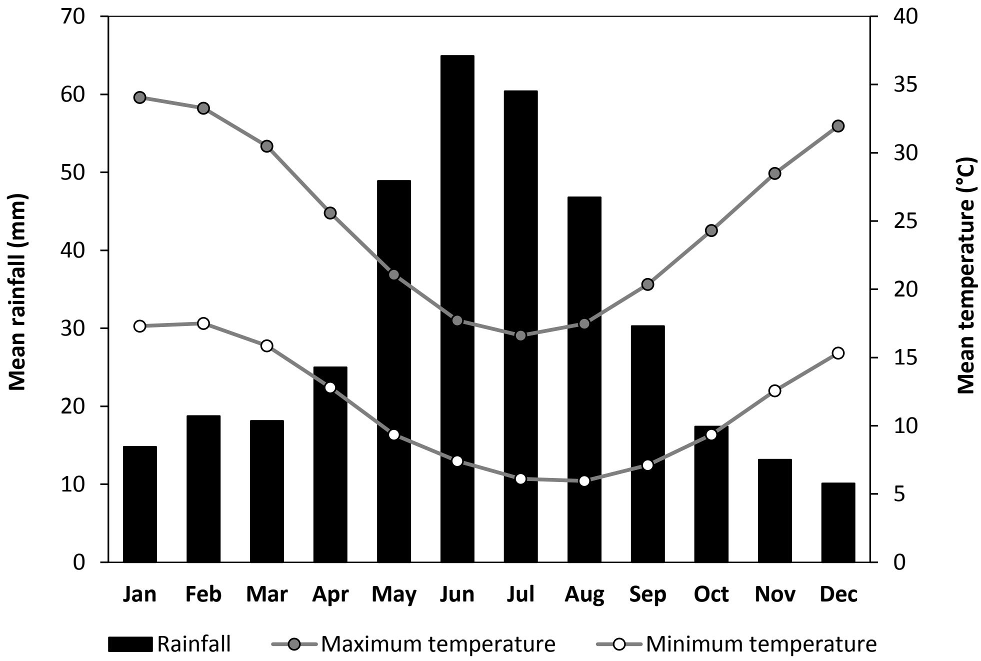

Figure 1.2. Average monthly rainfall and temperature at Cunderdin in Western Australia's Wheatbelt (for the years 1951 to 2006). Data source: Australian Bureau of Meteorology

As discussed later, the region is globally unusual as it represents one of the few instances where regional-level changes in observed rainfall have been attributed to anthropogenic climate change (Karoly, 2014). As such, the region represents an opportunity to show how current and projected human-induced climate change impact on a region's farm businesses. Concurrent to the physical impacts of climate change, policies aimed at mitigating greenhouse gases could also impact upon farm businesses. Indeed, lately in Australian politics there has also been much interest in, and enthusiasm about, using agriculture to mitigate climate change, based on a belief that sequestration of carbon in agricultural soil and through reforestation of farmland could deliver cost-effective abatement. This thesis investigates how broadacre agriculture in this region may be affected by these two climate-related forces (physical impacts and the impact of, and opportunity for, mitigation policy). Accordingly, the remainder of the Introduction is devoted to these topics.

1.2 Changing climate in the Western Australian Wheatbelt

1.2.1 Projected changes in climate

If the changes in climate projected for the study region had to be described in two words, those words would be 'hotter' and 'drier'. To understand why, consider the recent climate projections for Australia as a whole by CSIRO and BoM (2015), and the related study by Hope et al. (2015) which focused on the study region for this thesis. In these two comprehensive studies, the results of over 40 Global Climate Models (GCMs)[2] from the CMIP5 (Coupled Model Intercomparison Project Phase 5) ensemble[3] of climate models were collated for a range of emissions scenarios or 'Representative Concentration Pathways'. Statistical downscaling techniques were used to couple the outputs from the GCMs with regional climate models. When communicating their results, CSIRO and BoM (2015) and Hope et al. (2015) emulated the approach of the Intergovernmental Panel on Climate Change (IPCC)'s Fifth Assessment Report by ranking the confidence of their projections based on the quality, amount, type and consistency of evidence. Using this approach, they predicted with *high confidence* that annual rainfall in the study region will decrease. These changes are not necessarily distributed equally throughout the year; in particular predictions of drying are less

[2] 'GCM' is used synonymously to abbreviate 'General Circulation Model' and 'Global Climate Model'; both refer to the same class of model, the latter title is often used when this class of model is employed for climate change projection.

[3] The CMIP5 ensembley underpins the Intergovernmental Panel on Climate Change's Fifth Assessment Report.

categorical for summer months (Figure 1.3). That said, it is worth noting that these summer months are: (a) already very dry, so these relative changes do not translate into large differences in the absolute amount of rain likely to be received and; (b) are outside the current May to October agricultural growing season. Compared to the 1986 –2005 period, June to November (i.e., the majority of the agricultural growing season) rainfall is predicted to change by −15% to +5% by 2030, and −45% to −5% by 2090.

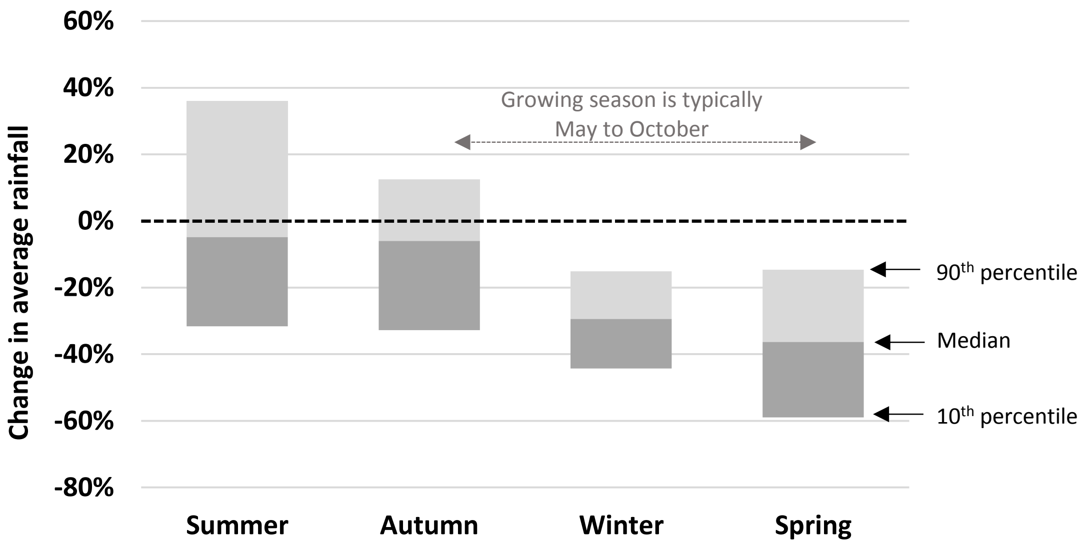

Figure 1.3. The distribution of changes to seasonal* rainfall in 2080 –2099 (relative to 1986 – 2005) for the south-west of Western Australia, including the Wheatbelt region, predicted by an ensemble of 39 CMIP5 GCMs. Results are shown for Representative Concentration Pathway 8.5 (a high emissions trajectory) so as to more clearly highlight the differences between seasons. Figure adapted from CSIRO and BoM (2015)

*Summer: Dec, Jan and Feb. Autumn: Mar, Apr and May. Winter: Jun, Jul and Aug. Spring: Sep, Oct and Nov.

In regards to temperature, Hope et al. (2015) predicted with *very high confidence* that, compared to 1986 –2005, average temperatures in the study region will increase by 0.5 to 1.1°C by 2030, and by 2090, in the range of 1.2 to 4.0°C. The temporal distribution of these temperature changes was projected to be relatively uniform across seasons.

The changes in temperature and rainfall predicted by Hope et al. (2015) and CSIRO and BoM (2015) are relatively consistent with earlier projections by others (e.g., CSIRO and BoM, 2007; Suppiah et al., 2007; Moise and Hudson, 2008; Delworth and Zeng, 2014). To demonstrate, Figure 1.4 shows not only the temperature changes predicted by CSIRO and BoM (2015), but also how they are relatively consistent with previous predictions by CSIRO and BoM (2007).

As a result of these changes, 'exceptionally dry years' (that is, years whose rainfall is low enough that they would qualify as being in the lowest 5th percentile of all years 1900 –2007) are predicted to occur every 6.5 years between 2010 to 2040; and 'exceptionally hot years' (years with temperatures above the 95th percentile of years from 1910 –2007) are predicted to occur every 1.2 years between 2010 to 2040 (Hennessy et al., 2008). Consequently, the occurrence of exceptionally low soil moisture levels in the south-west of Australia (including the Wheatbelt region) is predicted to increase by a factor of about 2.5 in 2030 compared to 1957 –2006 (Hennessy et al., 2008).

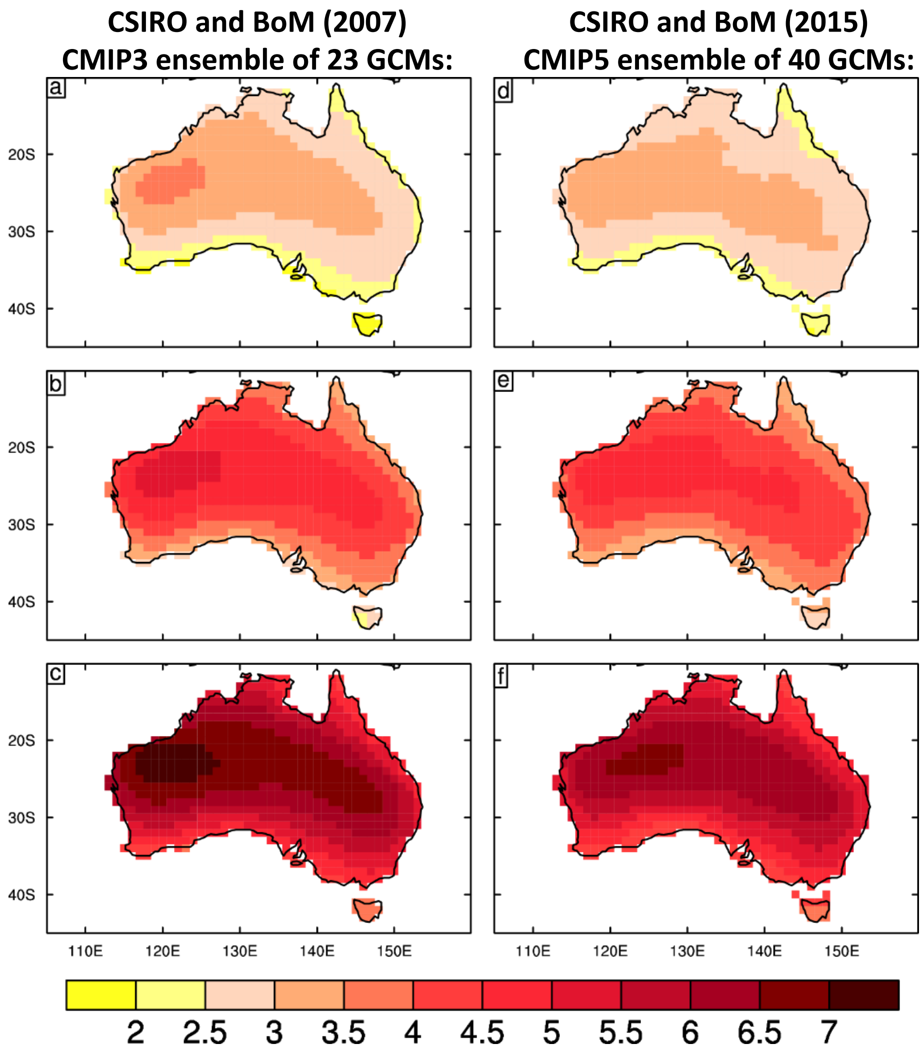

Figure 1.4. Annual changes to average temperatures (°C) by 2080 –2099 relative to 1986 –2005 as predicted by CSIRO and BoM (2007) using the CMIP3 ensemble of 23 GCMs (left panels) and by CSIRO and BoM (2015) using the CMIP5 ensemble of 40 GCMs (right panels). Middle maps (**b**, **e**) show the median prediction; top (**a**, **d**) and bottom (**c**, **f**) rows of maps show the 10th and 90th percentile, respectively. Figure source: CSIRO and BoM (2015).

The projections are also broadly consistent with changes in temperature (Asseng and Pannell, 2013) and rainfall (Timbal et al., 2006; Delworth and Zeng, 2014) that have been experienced in the region in recent decades. For instance, since 1975 June-July rainfall has declined, on average, by 20% (Ludwig et al., 2009). Figure 1.5 shows how growing season rainfall declined in the Wheatbelt region in the last century. This decline in rainfall is one of the only instances globally where regional-level changes in precipitation have been attributed to anthropogenic climate change (Karoly, 2014)[4].

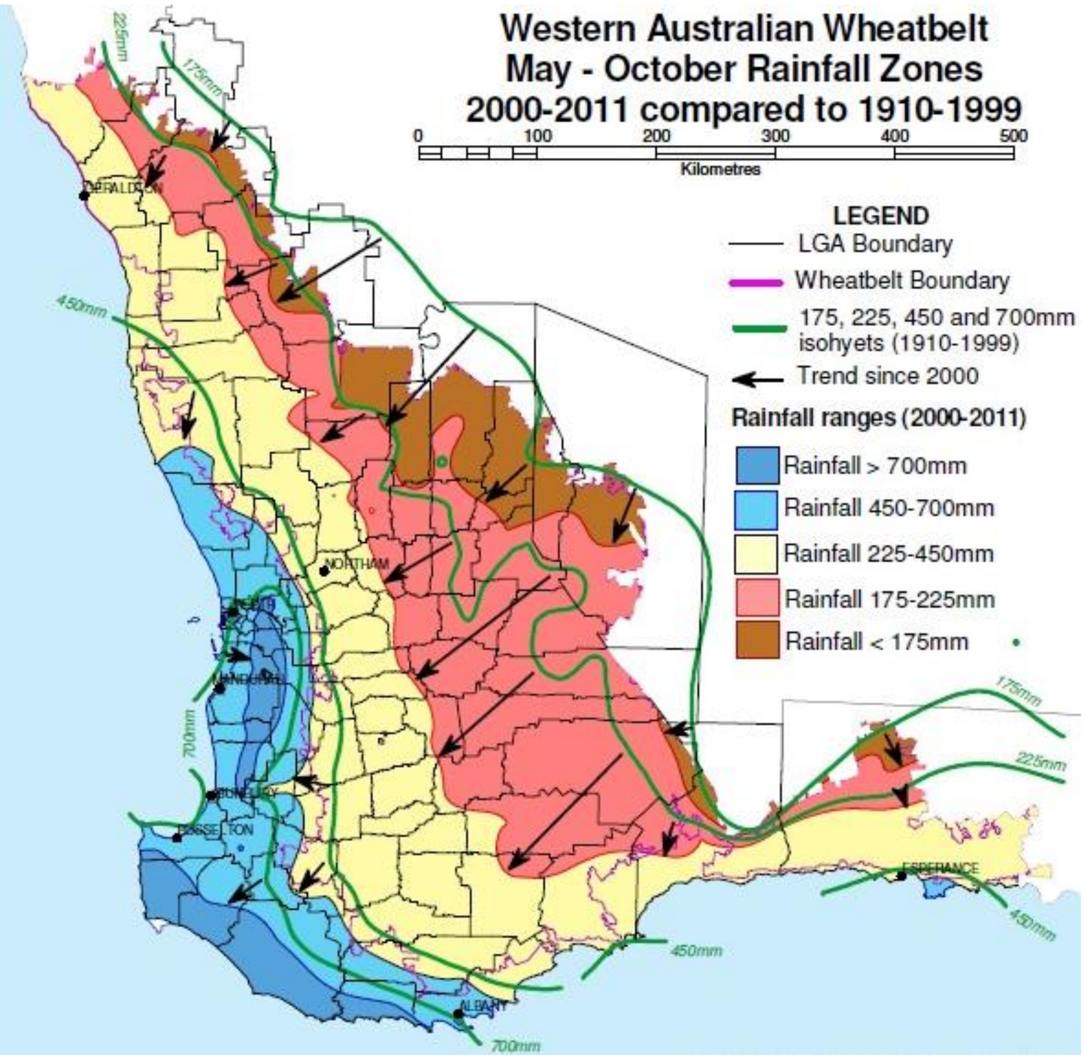

Figure 1.5. The difference in average rainfall received during the May to October growing season between 1910 –1999 and 2000 –2011. Figure source: Dept. of Agriculture and Food, Western Australia https://www.agric.wa.gov.au/drought/evolution-drought-policy-western-australia?page=0%2C3

[4] Compared to temperature changes, rainfall changes are relatively more difficult to conclusively ascribe to human-induced changes to the atmosphere.

1.2.2 Impact of these changes in climate

To investigate the effect on crop yields of this already-observed warming and drying, Ludwig et al. (2009) used a crop model to simulate a time-series of crop yields based on actual weather observations. Their results suggested that yields had been unaffected by these climatic changes. A couple of explanations have been offered for this counter-intuitive result. The first is that rainfall reductions have been concentrated in June and July, which are the two wettest months, during which moisture typically does not limit crop production. The second is that the decline in average rainfall has mainly been caused by a reduction in the frequency of very wet years (which can actually be too wet for crop growth); whilst there has been no increase in the frequency or severity of very dry years (Ludwig et al., 2009; Asseng and Pannell, 2013).

Empirical records of crop yields—which unlike model simulations, cannot control for the effect of improvements in agronomic practices and technology—support Ludwig et al.'s (2009) results, to a point. Even though the climate was getting hotter and drier, between the mid-1970s and the turn of the century, there have been large increases in average crop yields in the Wheatbelt (Figure 1.6). However, in their analysis Ludwig et al. (2009) only considered changes in climate up until 2004. The empirical data in Figure 1.6 extends to 2013, during which time the warming and trend continued. It shows that in more recent times yield growth has stagnated. Whether this stagnation is caused by management factors, or more recent changes to climate (and thus is perhaps indicative of future trends), or is a combination of both, remains a matter of debate (W. Anderson, M Robertson pers. comms.). Similar stagnation of yield growth has been noted elsewhere in the world (Lin and Huybers, 2012), with the role of climate change also a matter of debate (Lobell, 2012).

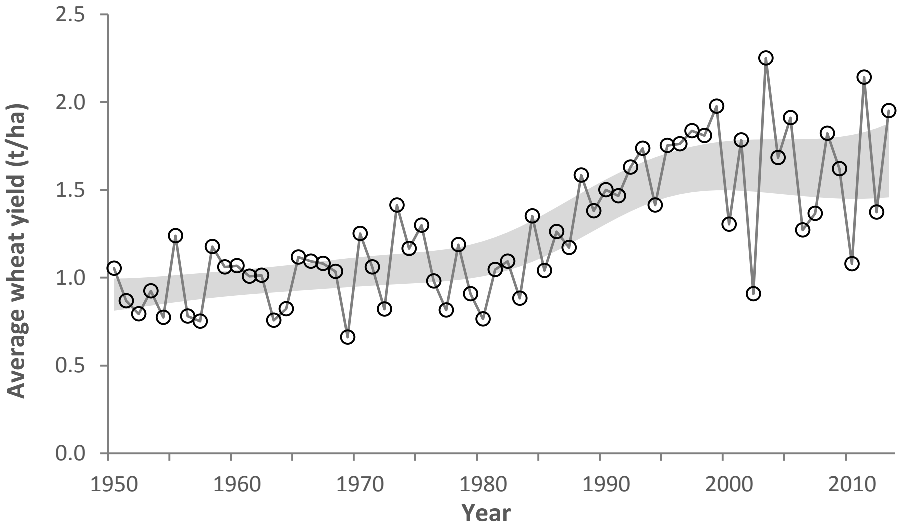

Figure 1.6. The average wheat yield in the Western Australian Wheatbelt from 1950 to 2013. Shading indicates of 95% confidence interval of a non-parametric trend.
Data source: Australian Bureau of Statistics

A number of studies have investigated the potential impact that future changes in climate may have on agriculture in the Wheatbelt region (e.g., van Ittersum et al., 2003; Asseng et al., 2004; John et al., 2005; Ludwig and Asseng, 2006; Ludwig et al., 2009; Farre and Foster, 2010; Crimp et al., 2012; Moore and Ghahramani, 2013; Anwar et al., 2015). These studies have primarily considered only biophysical impacts, on lone agricultural enterprises, in isolation (i.e., impacts on just wheat yields, pasture growth or barley yields, independent of each other). For example, van Ittersum et al. (2003); Asseng et al. (2004); Ludwig and Asseng (2006); Ludwig et al. (2009) and Farre and Foster (2010) all considered wheat. Anwar et al. (2015) considered impacts on wheat, barley, canola, lupin and field pea but each crop was still considered separately. Likewise, Moore and Ghahramani (2013) investigated impacts on pasture-based sheep production, but by itself and not as part of a greater farming system. Comparing these studies is problematic for a variety of reasons. They employed different analytical approaches, different climate scenarios, different soil types considered and different locations. Identifying the threshold levels of rainfall, temperature and/or CO_2 change, beyond which particular outcomes prevail is especially difficult to discern. Nevertheless, broadly speaking these studies found small to moderate temperature increases could benefit production by enhancing growth during the cooler winter months, and similarly, rainfall reductions also could reduce waterlogging during these months. These benefits, when coupled with the plant-growth enhancing effect of elevated atmospheric CO_2 could offset much of the detrimental effects of climate change, or even see production increased, particularly in locations that typically received higher rainfall and/or were located in the cooler southern parts of the region (van Ittersum et al., 2003; Asseng et al., 2004; Ludwig and Asseng, 2006; Farre and Foster, 2010; Anwar et al., 2015). But in hotter, drier locations; or where the changes in temperature and/or rainfall were large; then the impacts were potentially quite detrimental (Ludwig and Asseng, 2006; Farre and Foster, 2010; Moore and Ghahramani, 2013; Anwar et al., 2015). For example, Crimp et al. (2012) predicted that for the lower-rainfall regions of the Wheatbelt, yields of wheat would not fall by more than 10% by 2030 in most scenarios, but by 2050 reductions of 40-50% were possible.

These existing analyses of climate change impacts in the Wheatbelt region have several limitations. Firstly, their consideration of impacts on single enterprises in isolation is unrealistic. This ignores that the majority of the farm businesses in this region are run as mixed cropping-livestock farms. Whilst wheat is indeed the dominant enterprise (Planfarm, 2015), it is always grown in rotation with other crops and often in rotation with pasture as well. These enterprises interact and affect each other. For example, crop residue (stubble) remaining after harvest can be used as supplementary source of livestock fodder, legume crops and pastures provide nitrogen for subsequent non-legume crops, and the rotation of crops affects the weed and disease burden present in the farming system. A change in the performance of one enterprise can therefore also have flow-on impacts on other enterprises, meaning the financial performance of the farm is dependent on the performance of the farming system as whole (Scott et al., 2013; Kollas et al., 2015; Reidsma et al., 2015).

Secondly, the focus of the existing analyses has overwhelmingly been on the biophysical impacts of climate change. How these biophysical impacts then translate into economic impacts has rarely been considered, and if so, in only very simplistic ways (e.g., Ludwig and Asseng, 2006). When it comes to assessing climate change impacts on the region, economic viability is ultimately the most meaningful metric to farmers. Whilst biophysical and economic impacts are obviously inherently related, one is not always a good indicator of the other (Scott et al., 2013) and their sensitivities to change can differ. For instance, a 10% reduction in yield may seem like a relatively modest change, but economically this may be the fraction of the yield that generates much, if not all, of the income that forms the profit margin once costs have been met. Moreover, much of the agricultural output from the region is exported. Hence, how climate change impacts on the global supply of these exported commodities and thereby affects their prices, will directly affect the incomes that farmers in the study region receive in coming decades. How global climate change may affect future prices of agricultural commodities is potentially crucial, but outside the scope of this thesis.

Lastly, the impact of changes in climate will depend on how well agricultural systems can be adapted to accommodate the change. Nonetheless, in many of the aforementioned existing analyses for the region, adaptation has not been allowed for. For those analyses this means they have effectively assumed that agronomic practices, land uses and management will all remain fixed in the face of changing farming conditions. Mendelsohn et al. (1994) went as far as to describe this assumption of no adaptation as analysing the “dumb farmer scenario”. Obviously adaptation options that may become available in the future cannot be known. However, it is unrealistic not to at least allow for adaptation with existing/currently known options. Where adaptation has been allowed for in existing analyses (e.g., Crimp et al., 2012), it has been considered in a simulation setting. This requires adaptation options to be identified before they are simulated which can be problematic should different options interact with each other (White et al., 2011). Theoretically, an optimisation modelling framework is superior in this regard for it will endogenously identify the most beneficial adaptation option or combinations of options (Klein et al., 2013). Furthermore, many adaptation options may involve changes at the farm level, for instance changes in land uses or rotations. Under the single-enterprise approach used in nearly all existing analyses, only isolated land uses (e.g., wheat production) are be considered, meaning adaptation of this type cannot be modelled (Reidsma et al., 2015).

All existing analyses of climate change impacts for the case study region suffer from at least some, but mostly all of the above limitations—with one exception. John et al. (2005) conducted a bioeconomic analysis of climate change impacts for the eastern part of Wheatbelt region at the whole-farm level, meaning they simultaneously considered impacts on all enterprises that make-up the farming system, with adaptation (with existing options) occurring endogenously. However, this analysis has its own unique set of limitations. The sophistication with which John et al. (2005) considered the biophysical impacts of climate change was limited, with the influence of increased CO_2 on plant growth not considered. Impacts on pasture production were considered particularly simplistically. The model that they used allowed livestock to be agisted out in times of low fodder supply, but in the context of climate change across a whole region, agistment opportunities may be scarce. Lastly, canola, an important breakcrop from an agricultural systems perspective—and both by planted area and economic value, the second most significant crop in the region (ABARES, 2015; Planfarm, 2015)—was not included in their analysis. Notwithstanding these limitations, they found climate change could potentially reduce farm profit by more than 50%.

1.3 Climate change mitigation and agriculture

In addition to direct impacts on production, agriculture in the study region may also be indirectly affected by climate change through policies aimed at mitigating greenhouse gas emissions. This could include policies to address the greenhouse gases emitted by agriculture, policies that encourage the sequestration of carbon on agricultural land, or both.

1.3.1 A brief history of climate policy in Australia

Climate change did not earnestly enter the mainstream political debate in Australia until 2007 when, for a number of reasons including the success of Al Gore’s *An Inconvenient Truth* and the ‘Millennium Drought’ (the worst drought on record in much of Australia—(van Dijk et al., 2013)), it became a major issue in that year’s election. In response to a rising public concern, the incumbent Liberal[5]-National Party coalition government, whose position on climate policy during its previous 11 years has been described as “obstructionist” (Macintosh, 2008, p.52), promised during the election campaign to enact a nationwide emissions trading scheme (ETS) by 2012. However, details about the proposed policy were vague. The emission target that would underpin the ETS was not specified and the government refused to ratify the Kyoto Protocol as part of its proposal (making Australia the only developed country apart from the United States not to have done so) (Macintosh, 2008; Rootes, 2008). In contrast, the opposition Labor Party promised to immediately ratify the Kyoto Protocol if elected, and to establish an ETS by 2010, with the target of reducing Australia's emissions by 60% compared to 2000 levels by 2050 (Macintosh, 2008).

[5] Although the term ‘liberal’ is synonymous with the left side of politics in many parts of the world, in Australia the Liberal Party is the dominant party from the right.

When the Labor Party prevailed at the election, the first official act of the new Prime Minster, Kevin Rudd, was to ratify the Kyoto Protocol (Bailey et al., 2012). However, progress on developing mitigation policy was much slower, and after having twice failed to secure Senate support to pass legislation to implement the promised ETS (by then known as the Carbon Pollution Reduction Scheme or CPRS), in early 2010 Prime Minister Rudd indefinitely deferred its implementation (Rootes, 2011). Following a leadership spill in June 2010, Rudd was replaced as leader of the Labor Party and Julia Gillard became Prime Minister. Seeking to obtain her own electoral mandate, Gillard soon called an election. The Labor Party failed to win a majority of seats in this 2010 election and was forced to rely on the support of independents and the Greens Party to form a centre-left minority government. One of the conditions demanded in return for this support was that the minority Government would enact comprehensive, meaningful, policy to mitigate emissions (the Greens had twice opposed the CPRS on the grounds that the policy was too weak) (Rootes, 2011). This led to the passing of the Clean Energy Future legislation, which created a carbon-pricing mechanism that was implemented in July 2012 (Andersson and Karpestam, 2012). Though technically an ETS with an initial fixed-price period (\$23/tCO_2-e[6] increasing at 2.5% p.a. in real terms for three years) (Australian Government, 2012), this policy was promptly labelled a 'carbon tax' by the opposition Coalition, a moniker by which it soon became known. This 'carbon tax' only applied to energy, industrial processing and waste sectors, and only to polluters in these sectors emitting over 25,000 tCO_2-e annually, meaning it did not directly apply to farm businesses.

Instead, as part of the Clean Energy Future package, the agricultural/land sectors were covered by a separate policy known as the Carbon Farming Initiative (CFI). This CFI was a baseline-and-credit offset scheme, similar to the Kyoto Protocol's Clean Development Mechanism. Land managers could claim 'credits' for projects which reduced emissions in the agricultural/land sectors or that sequestered carbon in soil or through reforestation (subject to the regulatory approval of a methodology to assess the amount of abatement generated by the project and also to ensure additionality, and permanence and prevent leakage) (Macintosh, 2013). Polluters could buy these credits in lieu of paying the carbon 'tax' (thereby giving the credits effectively the same value as the \$23/tCO_2-e 'tax') (DCCEE, 2010a).

[6] Carbon dioxide equivalents or CO_2-e is a common unit that allows different greenhouse gases to be expressed on equivalent terms, based on their 100 year global warming potentials relative to CO_2.

The Liberal-National Party coalition opposition prosecuted a very strong (and politically successful) campaign against the implementation of the carbon tax, with Opposition Leader Tony Abbott giving his “pledge in blood” to “axe the tax” if his party was elected at the next election (Packham and Vasek, 2011; Frankel, 2015). Their campaign was mainly targeted at what effect the carbon tax would have on electricity prices. However, many claims about the impact it would have on other industries were also made. This included agriculture, with the Senate Leader of the rural-based National Party (in)famously declaring “It'll be the end of our sheep industry. I don't think your working mothers are going to be very happy when they're paying over $100 for a roast.” (Henderson, 2014).

The Coalition opposition proposed to repeal the legislation that created the ‘carbon tax’ and create an alternative mitigation policy called ‘Direct Action’. The fundamental premise of Direct Action was that instead of charging large polluters a fee for the externalities caused by their emissions, the government would instead use public funds to directly purchase abatement. Despite the use of public funds, the scheme was still said to be ‘market-based’ because abatement would be purchased via a reverse auction process. To supply this abatement the Coalition looked first and foremost toward the agriculture/land sectors, proposing that by 2020 at least 150 million tonnes of CO_2 could be sequestered in agricultural soils annually, and for a price of $10/t$CO_2$ (Coalition, 2010). Even allowing for political hyperbole, the implications of this policy proposal were clear: there was considerable potential to abate greenhouse gas emissions through agriculture/land management, and at a low cost. It was hoped that as well as providing a win for the environment, sequestration would also create a lucrative new industry for farmers. Whilst experts questioned this (e.g., Taylor, 2011), at the time there was little research in Australia—particularly using detailed agricultural modelling, and especially about soil carbon—to provide more robust guidance. Indeed, as Garnaut (2011, p.86) commented, “There is great uncertainty about the claims [of how much income] that the land sector may make on carbon revenue, but they are potentially large.”.

At the 2013 Federal election the Liberal-National Party coalition prevailed, and in July 2014 the carbon tax was repealed. Consistent with the central role that the agricultural/land sectors were anticipated to play in the new approach to climate mitigation, the Direct Action policy to replace the carbon tax was implemented by revamping and enlarging CFI to form an ‘Emissions Reduction Fund’ (or ERF). The legislation to morph the CFI into the ERF was passed in November 2014.

At around the same time in late 2014, a mandatory statutory review of the original CFI was due. This review found that during the CFI’s lifetime (and for a carbon price upwards of \$23/tCO_2-e) it had abated just 10 million tonnes of CO_2-e (an average rate of approximately 2.5 million tonnes per year) (Climate Change Authority, 2014a). Only 5% of this abatement had come from reforestation and other forestry projects, and just 1% from agriculture. The majority of abatement came from existing projects capturing fugitive emissions from landfill, a non-farming activity that was also eligible under the ‘Carbon Farming Initiative’. The relatively slow approval of methodologies for assessing the amount of abatement generated by an activity (thereby restricting the amount of activities credits could be claimed for, especially in the first years of the CFI), and policy uncertainty and associated doubt about the future price of credits due to the extremely acrimonious political debate about mitigation policy in Australia were both acknowledged as having restricted the amount of abatement provided by the CFI (Climate Change Authority, 2014a). Nonetheless, the quantity achieved was well short of the abatement that the Coalition (when in opposition) had proposed that soil sequestration alone would be delivering in 2020, once the CFI was revamped and morphed into the ERF. In an effort to improve participation and performance, it was promised that compared to the CFI, reporting, auditing and administration would be streamlined in the ERF, yet despite this streamlining the fund would still purchase only genuine, high-quality abatement (Australian Government, 2014).

Clearly, there are many questions and unknowns about the possible role of agriculture in climate change mitigation in Australia. Research on this topic could potentially make an important and useful contribution to this policy debate. To properly fill this research gap requires an understanding of agriculture’s contribution to greenhouse gas emissions, the options for abatement potentially available from agriculture, and how these options could be incorporated into an effective policy framework.

1.3.2 Agriculture as an emissions source

Agricultural production is estimated to be directly[7] responsible for 10 to 12% of greenhouse gas emissions globally (Smith et al., 2014). In addition to being responsible for 90% of CO_2 emissions that are not caused by the combustion of fossil fuels, agriculture accounts for 50% of methane (CH_4) and 60% of nitrous oxide (N_2O) emissions globally (Muller, 2012; Tubiello et al., 2013; Smith et al., 2014). The latter two are powerful greenhouse gases: based on the 100-year Global Warming Potentials stipulated in the IPCC's 4th Assessment Report, CH_4 and N_2O are 25 and 298 times more potent than CO_2, respectively.

Agriculture accounts for 15.7% of Australia's total greenhouse emissions (Department of the Environment, 2015b). This includes 59% and 69% of Australia's total CH_4 and N_2O emissions respectively (Australian Greenhouse Emissions Information System, 2016). On a per-capita basis Australia's agricultural emissions are amongst the highest in world (largely due to the relatively large number of ruminant livestock and relatively small population) (Garnaut, 2008). Agriculture's contribution to Australia's emissions is also large relative to its contribution to GDP (Garnaut, 2008) (i.e., its emissions intensity is high). In Western Australia, sources of agricultural emissions include (in decreasing order of significance): enteric fermentation by ruminants; burning of savannas in rangeland agriculture; emissions from soils (N_2O emitted due to the application of nitrogenous fertilisers, manure and crop residues); adding lime to soils; manure management and; urea hydrolysis (Department of the Environment, 2015b; Australian Greenhouse Emissions Information System, 2016).

Several policy approaches could be used to address agricultural emissions. Placing a 'carbon price' on agricultural emissions is one option: farmers could be required to either pay a tax or purchase a permit for their on-farm emissions (e.g., Kingwell, 2009). This price would be mandatorily imposed upon farm businesses. This policy approach carries with it the risk of 'leakage', whereby the implementation of a mitigation strategy or policy in one location, or targeting one type of greenhouse gas, causes an increase in emissions at another location and/or of another greenhouse gas (Cacho et al., 2008). Australia's agricultural sector is very export orientated, with around 60% of all

[7] This figure refers to emissions directly generated by agricultural activity. That is, emissions that occur on farms as a direct result of agricultural production. It does not include emissions indirectly related to agricultural production, like those caused by fertiliser manufacture or food processing.

production exported (ABARE, 2009). Therefore applying a carbon price domestically, without commensurate action on agricultural emissions internationally, may cause domestic producers to lose competitiveness and reduce production, with the shortfall made up for by increased production in countries not taking measures to reduce emissions (Cooper et al., 2013). Such leakage can mean that a policy causes 'economic pain for minimal environmental gain'. Fears of this have contributed to the stifling of proposals to include agriculture in a broad-based emissions price in New Zealand[8] (Cooper et al., 2013). While there are strategies to help emissions-intensive, trade-exposed industries adjust to the imposition of mitigation policy (such as transitional payments or the granting of free permits), imposing a carbon price on the agricultural sector is currently not on the political agenda in Australia.

An alternative policy option is the approach of the CFI/ERF, where agriculture is not a party to mandatory policy but where agricultural producers can take voluntary action to reduce their emissions in return for saleable credits or financial payment (e.g., Cottle et al., 2016). Garnaut (2011) suggests that in the longer term, the superior policy approach is the inclusion the land sector in a mandatory pricing mechanism; whether that is politically feasible is another matter.

An issue for either policy approach is that agricultural emissions are inherently difficult to estimate. The dispersed nature of agricultural production means quantifying how much of an emission-causing activity is occurring is difficult (Olander et al., 2013). This is greatly compounded by the fact that the amount of emissions that this activity will generate often differs with climate and/or the weather, soil type, type of agricultural practices employed and the timing of their employment, nutrition, genetics, etc. (e.g., Gibbons et al., 2006; Hegarty et al., 2010; Baldock et al., 2012; Berdanier and Conant, 2012; Zehetmeier et al., 2014; Finn et al., 2015; Young et al., 2016). As a result, the uncertainty associated with estimates of agricultural emissions globally is thought to be in the range of 10 –150% (as opposed to 10 –15% uncertainty for fossil fuel emissions) (Eggleston et al., 2006). Even in the relatively well-developed EU 15 group of countries (who tend to have more resources to devote to emissions accounting), the level of uncertainty associated with estimates of agricultural emissions is thought to be 80%, as compared to 1.1% and 8.8% for fossil fuel combustion and industrial processing,

[8] On a per capita basis New Zealand's agricultural emissions are more than double Australia's, and nearly eight times the OECD average (Garnaut, 2008).

respectively (EEA, 2015). The difficultly of accurately measuring agricultural emissions is a significant obstacle for the development of mitigation policy to address these emissions (Garnaut, 2008).

1.3.3 Agriculture as a carbon sink

By storing carbon, agriculture can also be an emissions sink. Carbon can be sequestered by changing land-uses or management strategies to increase the carbon content of the soil, or by reforesting agricultural land.

1.3.3.1 Sequestration in vegetation

Of all the mitigation options potentially available in the agricultural/land sectors, sequestration in vegetation is perhaps the most studied, both internationally (e.g., van Kooten et al., 1999; Lewandrowski et al., 2004; Antle et al., 2007b; Torres et al., 2010; Luedeling et al., 2011; Wise and Cacho, 2011 etc.), and in Australia (e.g., Harper et al., 2007; Burns et al., 2011; Paul et al., 2013a; Paul et al., 2013b; Polglase et al., 2013). There is little endemic woody vegetation remaining on farms in the Wheatbelt region (Schur, 1990), meaning cleared farm land would have to be reforested to sequester carbon in vegetation. Several bioeconomic analyses have evaluated such reforestation at the whole-farm level in the Wheatbelt region (Petersen et al., 2003a; Flugge and Schilizzi, 2005; Flugge and Abadi, 2006; Kingwell, 2009; Jonson, 2010). Differences in analytical approach (differences in timescales, study area, planting species and spatial configuration of planting, dedicated sequestration planting or harvested forestry, and policy settings) complicates comparisons across these studies. Nonetheless, broadly speaking they found reforestation could be viable, usually on the most marginal soil types for agricultural production, if upwards of \$40 –70/tCO_2-e could be received for sequestration. These studies also found that if a mandatory carbon price was imposed on agricultural emissions then sequestration became more attractive (because agricultural production decreased in profitability), and depending on the carbon price, could potentially more than compensate for the burden of the carbon price on the rest of the farm.

Similar to agricultural emissions, the measurement of sequestration also presents a challenge. Whilst modelling can be used to estimate sequestration, the accuracy of these models tends to be limited to the level of regional averages (e.g., Paul et al., 2015).

Estimates of sequestration obtained with these regional level models may be conservative compared to actual rates measured in the field (Jonson, 2010).

1.3.3.2 Sequestration in agricultural soils

Unlike when arable land is reforested, sequestering carbon in soil enables agricultural production to continue (though the productivity and/or commodity produced may change). At the global level, some uncertainty exists about the scope of soil carbon as a mitigation option. Some studies are strongly optimistic in their assessment of soil carbon's potential. For instance, Lal (2004a; 2004b) describe it as a truly win-win mitigation strategy that could offset 5 –15% of global fossil fuel emissions. Others are more circumspect, suggesting there is a considerable gulf between the mitigation that is theoretically achievable with soil carbon and the potential that is feasible economically, and therefore that the mitigation capacity of soil carbon has been over-emphasised in the literature (e.g., Freibauer et al., 2004; Powlson et al., 2011; Alexander et al., 2015). As mentioned in Section 1.3.1 above, uncertainty about the mitigation potential offered by soil carbon has also pervaded the debate about climate policy in Australia.

In an effort to lessen this uncertainty, Sanderman et al. (2010) conducted a thorough review of carbon sequestration in relation to Australian agriculture. Practices they identified as having the potential to increase soil carbon which could apply to the Wheatbelt study region included the adoption of conservation tillage, the retention of crop residues ('stubbles') and increased areas of pasture relative to crop.

The adoption of conservation agriculture (minimum and no-tillage practices) is perhaps the most commonly discussed way of sequestering carbon in agricultural soils, both in Australia and internationally (e.g., Lal and Kimble, 1997; Follett, 2001; West and Marland, 2002; Manley et al., 2005; Antle et al., 2007a; Grace et al., 2010; Syswerda et al., 2011; Lal, 2015). Whilst some question how much carbon will actually be sequestered by the adoption of conservation agriculture (Dalal and Chan, 2001; Chan et al., 2003; Baker et al., 2007; Luo et al., 2010a; Chan et al., 2011; Maraseni and Cockfield, 2011; Robertson and Nash, 2013; Kirkegaard et al., 2014; Powlson et al., 2014; Conyers et al., 2015; VandenBygaart, 2016), in the context of the study region, this is a moot point. Conservation agriculture is already widely adopted, with more than 90% of the crops in Western Australia being established using no-tillage practices (D'Emden and Llewellyn, 2006; Llewellyn and D'Emden, 2010; Llewellyn et al., 2012), meaning that any increase in soil carbon (whatever size it may be) due to the adoption of minimum and no-tillage practices is mostly already happening, in the absence of any climate mitigation policy to incentivise them. Like conservation tillage, stubble retention is also a relatively common practice: in the Wheatbelt region around 80% of crop residues are retained (Llewellyn and D'Emden, 2010).

Pasture is also already a component of mixed crop-livestock farming systems. However, typically 60 –85% of the farm area is cropped in the mixed farms of the Wheatbelt region (Planfarm, 2015), meaning there is scope to increase the amount of land under pasture relative to crop. Depending on soil type, initial carbon levels, climate, pasture type, duration of the pasture phase and the total time period considered, estimates of the amount of carbon that could be sequestered with the conversion of cropped land to pasture in southern Australia range from 0.26 to 2.6 tCO_2/ha/year (Chan et al., 2011; Thomas et al., 2012; Hoyle et al., 2013; Sanderman et al., 2013; Conyers et al., 2015; Meyer et al., 2015). Perennial species of pasture plants have been identified as potentially being able to sequester more than annual species because: a) they can utilise out-of-season rainfall to photosynthesise more carbon, and; b) perennial species tend to allocate a greater proportion of the carbon they photosynthesise to their root system, where it cannot be removed by grazing (Sanderman et al., 2010; Sanderman et al., 2013; Eyles et al., 2015).

A key determinant of whether soil carbon offers an effective way for agriculture to participate in climate change mitigation will be cost. Whilst a number of studies across the globe have considered the economics of sequestering carbon in soil (e.g., Pautsch et al., 2001; Antle et al., 2002; Lee et al., 2005; Meyer-Aurich et al., 2006; Diagana et al., 2007; Choi and Sohngen, 2010; Popp et al., 2011; Grace et al., 2012; Alexander et al., 2015), the results of these studies may not be relevant to Australian conditions. Due to unfavourable climatic conditions and/or their inherent edaphic characteristics, soils in Australia generally store less carbon than soils found in agricultural regions of the northern hemisphere (Sanderman et al., 2010). Furthermore, differences in farming systems, economic conditions (e.g., labour costs, access to finance), agricultural policy settings, crop types and yields mean that the economics of changing land use or management strategy to influence soil carbon levels will differ in Australia.

Despite the uniqueness of conditions in Australia's principal agricultural regions, very little economic analysis of carbon sequestration in Australian agricultural soils has been conducted. Across an approximately 9 million hectare study area in south-eastern Australia, Grace et al. (2010) found that for a \$13.6/tCO_2-e carbon price, 6.4 million tonnes of CO_2 could be sequestered over a 20 year period by adopting no-tillage practices. Whilst this may sound like a large amount of abatement, this equates to the removal of only approximately 0.32 million tonnes of CO_2 from the atmosphere annually, which is less than 0.06% of Australia's total emissions for the year 2013[9] (Department of the Environment, 2015b). If the carbon price doubled to \$27.3/tCO_2-e, then the estimate of sequestration increased to 12.3 million tonnes of CO_2 (which when averaged to an annual rate, would be equivalent to 0.11% of Australia's 2013 emissions). A factor contributing to the relatively low amount of sequestration estimated by Grace et al. (2010) was that approximately 65% of farmers in their study area in south east Australia were already practising conservation tillage (i.e., they were doing it for a \$0/tCO_2-e carbon price), meaning the potential for further adoption (and sequestration) is relatively low. As mentioned previously, adoption of conservation tillage is even higher in the Western Australian Wheatbelt region. Such is the paucity of research on soil sequestration in Australia that Grace et al.'s (2010) study represents the first bioeconomic analysis of the feasibility of it (for any of Australia's major agricultural areas). The second such analysis is presented in this thesis.

1.3.3.3 Sequestration policy can be challenging

The existence of an opportunity to cost-effectively sequester carbon is a necessary condition for a policy to promote such sequestration to be effective and efficient, but it may not be sufficient. For that, the opportunity also has to be capable of being cost-effectively employed within a mitigation policy.

As a mitigation option, sequestration is arguably best employed under an offset-credit type policy framework in which land managers receive saleable credits or financial payment in return for carbon they voluntarily sequester (Lewandrowski et al., 2004). This is the policy approach that has been adopted thus far in Australia, first with the CFI and later with the ERF. It is also the approach that has been favoured internationally, for instance in the Clean Development Mechanism or Specified Greenhouse Gas Emitters

[9] As of May 2016, 2013 is the most recent year for which comprehensive data on Australia's emissions is available.

Regulation scheme in Alberta, Canada. In offset-credit schemes that work via financial incentive and in which participation is voluntary, there is a need to ensure the additionality of sequestration. Purchasing abatement that is 'non-additional' (i.e., which would have occurred anyway, without payment), sees funds wasted for no gain (Horowitz and Just, 2013). The problem of leakage, raised above in relation to agricultural emissions, is also an issue. Furthermore, sequestration is a reversible process: to maintain carbon in its sequestered state requires the continuation of the sequestering activity (or at least that it not be replaced by an activity that would cause the sequestered carbon into be re-released to the atmosphere) (e.g., Janzen, 2015). This issue is known as 'permanence'.

Preventing leakage and ensuring additionality and permanence can greatly complicate the incorporation of sequestration into mitigation policy (e.g., Murray et al., 2007; Cacho et al., 2008). If these requirements are not fulfilled then the effectiveness and integrity of abatement provided by sequestration will be reduced. But at the same time, the transaction costs associated with ensuring them will make sequestration more expensive.

1.3.3.4 Climate change impacting sequestration

In an overwhelming majority of the literature on the agriculture/land sectors, the mitigation of climate change and the impacts of climate change and adaptation are researched as separate topics. In the future, however, both are likely to occur simultaneously, and therefore they may interact with each other. In particular, sequestration is a mitigation option with the potential to be impacted by climate change, given that it relies on climate-mediated photosynthesis to remove carbon from the atmosphere either directly (sequestration in vegetation) or indirectly (soil carbon). This, combined with the long timeframes associated with sequestration and its reversible nature, make sequestration arguably more sensitive to future climatic conditions than most mitigation options.

With Western Australia's Wheatbelt already semi-arid, further warming and drying of its climate—especially if large in magnitude—could have a detrimental effect on tree growth, and therefore rates of sequestration (per area) from reforestation. The result of this would obviously be a reduction in the income sequestration can generate per unit of area. Intuitively, if sequestration's capacity to generate income was reduced then it would seem to become less attractive to landowners. However, the impact that climate change has on the attractiveness of sequestration—and therefore its cost-effectiveness as a mitigation option (i.e., on the amount of abatement that can be obtained from sequestration for a given carbon price)—will also depend on the impact that the same climatic changes have on competing land uses like conventional agricultural pursuits (i.e., on the opportunity cost of using land for sequestration). Changes in the competitiveness of alternative land uses may or may not compensate for a reduction in the biophysical rate of sequestration. Policies to reduce agricultural emissions may also affect the viability of reforesting farm land for sequestration.

Whilst climate change impacts and mitigation policy for agriculture tend to be researched independently (as stated above), some recent Australian bioeconomic studies have considered interactions between the two (Bryan et al., 2014; Connor et al., 2015; Bryan et al., 2016a; Bryan et al., 2016b; Grundy et al., 2016). This series of related analyses all employ essentially the same (integrated assessment) methodological approach: through equilibrium modelling they estimate economic growth, carbon pricing, and demand for energy and agricultural commodities into the future. These estimates are coupled with climate scenarios and the resultant spatial changes in land use across Australia are projected. The complexity with which these analyses have predicted future trends in macro-level economic parameters is impressive. However, in other aspects these studies have some limitations. For instance, they did not account for the possibility of increased atmospheric CO_2 levels boosting crop growth. This means they only considered the effect of changes in precipitation and rainfall. In addition, to determine the effect that these changes in rainfall and temperature would have, they used a regression model (as opposed to the more typical approach of using complex, process-based crop models to directly simulate the effect of climate change on yields).

In the aforementioned studies, a constrained partial-equilibrium linear programming model of land use was applied to a spatial grid distributed across the Australian continental landmass. An alternative approach—analysis instead of the operation of the farm production system as a complete business package at the farm-level—would allow for better consideration of how interactions might affect the interrelationships between different enterprises and the business performance of different farming systems (as discussed in Section 1.2.2). Further, a whole-farm optimisation framework can take into account how management and enterprise choices (at least those that are known and presently available) could be adjusted to reduce the impact on farm businesses of the interactive effects of mitigation policy and climate change.

An analysis that has previously attempted to investigate the simultaneous effects of mitigation policy and climate impacts within a whole-farm optimisation framework is John et al.'s (2005) study for the eastern part of the Wheatbelt region. However, in their analysis John et al. (2005) unrealistically assumed that that the growth of woody perennial vegetation (a sequestering option) would be unaffected by changes in climate. Also, as discussed at the end of Section 1.2.2 above, there are limitations with how John et al. (2005) modelled climate change impacts on agricultural production. A farm-level analysis that more rigorously gives consideration to the simultaneous and potentially interactive effects of changes in climate and the implementation policy to mitigate it would therefore represent useful contribution to the literature, and a potentially important component of investigating how agriculture in the Wheatbelt region may be affected by climate change.

1.4 Thesis aims and objective

Western Australia's Wheatbelt is currently one of Australia's major agricultural regions. However, in the future its climate is almost unanimously predicted to become warmer and drier. The potential bioeconomic impact of such climatic changes have not been well explored previously at the farming-systems level. At the same time, the potential to mitigate climate change through the agricultural sector has featured prominently in the public policy debate in Australia. Whilst government policy has been developed on the basis that agriculture can make a significant and relatively immediate contribution to climate mitigation, in reality, there are many questions and unknowns about the possible role of agriculture. Research on this topic could therefore potentially make an important and useful contribution to this policy debate.

This thesis has two main aims: (a) to examine the potential for agriculture to provide cost-effective emissions abatement in the Wheatbelt region, and more generally, how agriculture may be incorporated into policies to mitigate global warming, and; (b) to investigate the bioeconomic impact of the future changes in climate that may occur in the Wheatbelt region. In pursuit of these aims, and consistent with the topics and issues explored earlier in this chapter, the objectives of this thesis are as follows (the relevant thesis chapters are shown in parenthesis):

- Critically explore the issues associated with designing policies that facilitate the abatement of climate change through the agricultural/land sectors, particularly in regard to sequestration [Chapter 2 and Chapter 5]
- Analyse the cost-effectiveness of using agricultural land in the Wheatbelt region to store carbon, either through reforestation or in soil [Chapter 3, Chapter 4, Chapter 5, and Chapter 7]
- Assess the potential impact of a carbon price on agricultural emissions for a typical Wheatbelt farming-system [Chapter 5 and Chapter 7]
- Analyse the potential bioeconomic impact of climatic change at the farming-system level [Chapter 6]
- Explore how changes in climate in the study region may interact with, and affect, the efficacy of mitigation options such as sequestration [Chapter 7]

1.5 Thesis structure

This thesis is presented as a series of journal articles, in accordance with Rule 40(1) of the University of Western Australia's regulations for higher degrees by research. Hence each of Chapters 2 through 7 represents a manuscript that has been prepared, submitted, or accepted for publication in a peer-reviewed journal.

The paper presented in each main chapter differs from its corresponding journal article slightly in terms of style: to ensure continuity and consistency across the thesis, the labelling and numbering of sections, tables and figures have been changed. Journal-specific referencing styles used in each paper have also been replaced with a universal format, with a single, amalgamated reference section presented at the end of the thesis. In matters other than style (i.e., in terms of content and results), the paper contained in each chapter is presented exactly as it has been published or submitted for publication. Therefore, because each paper has been constructed as a separate, independent document for publication in different journals, there is naturally some repetition—particularly in matters related to methodology and background material—when these papers are read as chapters of the same document. Abbreviations and terminology remain unique to each chapter.

At the conclusion of the thesis the findings of the different papers are synthesised and their collective implications, in terms of how agriculture in this region may be affected by climate change (physical impacts and the impact/opportunity for mitigation policy), are reflected upon. Limitations of the analyses presented in the thesis are also discussed and areas of future research about climate change and its effect on the study region are suggested.

Chapter 2. Paper 1. Challenges in developing effective policy for soil carbon sequestration: perspectives on additionality, leakage, and permanence

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The candidate's overall contribution to the published paper was approximately 80%, as certified in the Statement of Student Contribution.

Challenges in developing effective policy for soil carbon sequestration: perspectives on additionality, leakage, and permanence

2.1 Abstract

If carbon sequestration is to be cost-effective substitute for reducing emissions then it must occur under a framework that ensures that the sequestration is additional to what would otherwise have occurred, the carbon is stored permanently, and any leakage is properly accounted for. We discuss significant challenges in meeting these requirements, including some not previously recognised. Although we focus on sequestration in soil, many of the issues covered are applicable to all types of sequestration. The common-practice method for determining additionality achieves its intention of reducing transaction costs in the short term but not in the medium-long term. Its design results in the least costly, additional abatement measures being excluded from policy support and fails to address how, in the case of sequestration, revisions to the additionality of sequestering practices should apply not just to the future, but in theory, also retrospectively. Permanence is sometimes approximated as 100 years of sequestration. Re-release of sequestered carbon after this will not only reverse the sequestration, but may raise atmospheric carbon to higher levels than they would have been if the sequestration had never occurred. Leakage associated with sequestration practices can accumulate over time to exceed the total level of sequestration; nonetheless adoption of such practices can be attractive to landholders, even when they are required to pay for this leakage at contemporary prices.

Policy Relevance: Globally much has been written and claimed about the ability to offset emissions with sequestration. The Australian Government plans to use sequestration to source much of the abatement required to reach its emissions targets. Designing effective policy for sequestration will be challenging politically, and will involve substantial transaction costs. Compromises in policy design intended to make sequestration attractive and reduce transaction costs can render it highly inefficient as a policy.

Keywords: additionality, sequestration, carbon credits, climate change mitigation, climate policy frameworks, environmental economics

2.2 Introduction

Much has been written about the potential to offset greenhouse gas (GHG) emissions by sequestering carbon (e.g., Subak, 2000; Lal, 2002; García-Oliva and Masera, 2004). To be effective, sequestration policies need to: encourage sequestration that is ‘additional’ to what would occur in the normal course of business (Meyers, 1999; Woodhams et al., 2012); avoid ‘leaking’ or transferring emissions to another location, time, or form (e.g., Cowie et al., 2012; van Kooten and de Vries, 2013); and ensure that carbon is stored ‘permanently’(McCarl, 2006; Murray et al., 2007) and not undone in the future without being replaced with other abatement.

Various authors have identified and analysed challenges in meeting these three requirements. For example, Montserrat and Sohngen (2009) estimated that in some voluntary offset schemes, up to 90% of the claimed emissions ‘savings’ may have been shifted or ‘leaked’ to another location. Gustavsson et al. (2000) articulated how additionality is an inherently uncertain concept due to its dependence on an unobservable counterfactual scenario, lamenting that the buyers and sellers of offsets share a common motive to exploit such ambiguity and overestimate abatement, meaning sound regulatory oversight is an imperative. Murray et al. (2007) considered the feasibility of different policy designs for dealing with permanence. Others have warned that implementing policies that overvalue temporary sequestration relative to permanent abatement may be (politically) convenient, but ultimately inefficient (Feng et al., 2002; Gramig, 2012).

A common theme in the literature is that satisfying requirements for additionality, non-leakage and, to a lesser degree, permanence increases transaction costs. These may be reduced by simplifying the policy, but at the cost of increased levels of uncertainty and reduced efficiency of the program (Subak, 2000; Cowie et al., 2012; Cacho et al., 2013; Capon et al., 2013).

With this paper we aim to: i) elucidate major issues around additionality, leakage, and permanence in the design of policy for sequestration of soil carbon, in the context of transaction costs and uncertainty; ii) identify potential perverse outcomes and inefficiencies in some of the policy approaches that have been proposed; and iii) consider the policy implications of our findings. Our study builds on the existing literature, extending it by focusing on issues that have not previously been emphasised, or in some cases, not previously recognised. These issues include that practices may be additional temporarily and yet not worth supporting in the short term; that practices can ultimately leak more emissions than they sequester and yet still be financially attractive to landholders; and that the use of a 100-year rule (or similar) as a proxy for permanence can lead to atmospheric carbon levels being higher than they would have been in the absence of a sequestration policy.

We focus on carbon sequestration in soils—when we use the term 'sequestration' we are referring to soil carbon—although many of the issues we identify are relevant to other forms of carbon sequestration and to the design of any future policies throughout the world that aim to encourage sequestration, be it in soil or in vegetation. As a policy example we use the Carbon Farming Initiative (CFI) introduced by the Australian Government in late 2011 and modified in 2014. Thus far, only a modest amount of abatement has been generated by the CFI (as of December 2014, 10.6 Mt of CO_2e), much of it from landfill gas projects that were instigated under previous state-based schemes (Climate Change Authority, 2014a) and currently no projects involving soil carbon have been implemented. However, a new Australian Government was elected in September 2013 with the stated intent of sourcing the majority of the Australia's future abatement from an expanded CFI. As recently as June 2014, the Government was aspiring to achieve over three-quarters of this abatement by storing carbon on farms (Neales, 2014). We will at times also refer to the only existing offset credit scheme in which soil carbon has played a major role: Alberta's Specified Greenhouse Gas Emitters Regulation (Climate Change Authority, 2014b). In this Canadian scheme 38% of credits generated have been from carbon sequestered by the use of use minimum- or no-till cropping practices (Swallow and Goddard, 2013).

This paper proceeds as follows. The next section considers potential policy approaches to, and dynamics of, carbon sequestration. Following this we consider different approaches for assessing additionality and the implications of its evolution through time. The risks created by the impermanent nature of carbon sequestration are then examined and the possible policy approaches for dealing with sequestration and leakage that occur over different timeframes are explored. We then reflect on the potential role of sequestration as part of a broader emissions mitigation strategy before summarising our findings.

2.3 Carbon sequestration: dynamics, policy approaches and concepts

2.3.1 Sequestration in Soil

Globally, the total amount of carbon stored in the top metre of soil (organic and inorganic pools) is estimated to be three times as much as the atmosphere and nearly four times as much as contained in living matter (Lal, 2002). However, with the expansion of agriculture, the carbon content of many soils has declined. It is estimated that in some regions up to 70% of these losses could be re-sequestered through improved land use or land management (Lal, 2002). For instance 'no-tillage' cultivation practices could increase soil carbon by about 16% worldwide (West and Post, 2002). For mixed cropping-livestock farms in Western Australia, increasing the portion of legume pastures from 30% of farmed area to 80% would sequester 6t of CO_2/ha across the entire farmed area over 30 years (Kragt et al., 2012). The amount of carbon sequestered when a new management practice is adopted depends on the initial carbon content, the practice, soil type and climate (Johnson et al., 1995; West et al., 2004).

As a mitigation activity, sequestration has two unique characteristics. First, when a sequestering practice is adopted, carbon storage typically increases[1], but at a diminishing rate through time until it plateaus at a new steady-state equilibrium (Figure 2.1) (West et al., 2004; Gramig, 2012; Hoyle et al., 2013). Consequently, only a finite amount of sequestration is possible. Furthermore, this finite opportunity can only be exploited once; the same management practice implemented at a later date will ultimately sequester the same amount of carbon (Figure 2.1). Second, sequestration is reversible. To retain stored carbon the sequestering (or an equivalent) practice must be continued; reverting to the previous practice re-emits the carbon. Importantly, these two characteristics are not shared by strategies that reduce emissions (i.e., that prevent GHGs from entering the atmosphere, as opposed to sequestration which instead removes CO_2 from the air). For this and other reasons, sequestration creates some particular challenges for policy design.

[1] Management changes can have a positive net effect on soil carbon levels in two ways: i) by bringing about absolute increases in soil carbon; ii) by preventing a decline in soil carbon that would otherwise occur had the business-as-usual practice continued. Although in this article we primarily focus on the first effect, many of the issues we raise also apply to the second, because in both cases the mitigation is typically finite, reversible, and occurs at a diminishing rate through time.

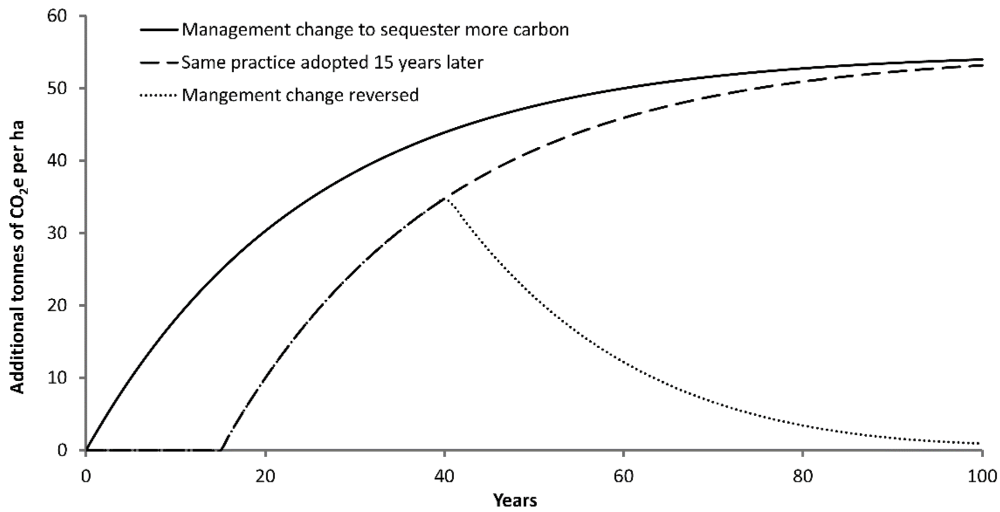

Figure 2.1. Stylised dynamics of carbon sequestration.

2.3.2 Potential policy approaches

Sequestering additional carbon to that which is optimal in the absence of a policy comes at an opportunity cost to farmers and so requires incentivisation[2]. One way to incentivise sequestration is to incorporate it within a comprehensive emissions trading or carbon tax scheme. In the original CFI, landholders could claim credits for additional carbon sequestration (subject to regulatory approval) and sell them to polluters who could use these credits to offset their liabilities to the Australian Government's then implemented carbon tax. After revisions to the policy in 2014, landholders can participate in a reverse auction and receive payment from the government if their bid is judged to provide sufficiently good value for money. In the United States, since the failure of "cap and trade" legislation there has been increased interest in more traditional payment-based approaches to climate policy (e.g., Jones et al., 2013; Lewandrowski and Zook, 2015; Murray, 2015). In either of these approaches (reverse auction or direct payment), payments might be made for estimates of sequestration, or just for the adoption of practices that are believed to sequester carbon (Subak, 2000; Cowie et al., 2012).

We now look at three concepts that are central to the design and implementation of an efficient policy for carbon sequestration: additionality, permanence and leakage.

[2] Sequestration practices may be financially attractive but still not adopted due to social or cultural factors (Pannell et al., 2006). Even though there may not be a financial opportunity cost in such a situation, incentivisation of some type will still likely be required to prompt practice change.

2.3.3 Concepts: additionality, permanence and leakage

For voluntary sequestration programs to work efficiently, whether they are offset- or payment-based, credits or incentives should only be provided for sequestration that is ‘additional’. In other words, the aim of the policy is to increase sequestering projects that “result in abatement that would not have occurred in the absence of the [policy]. There would be no reduction in emissions as a result of the [policy] if the project activity would have occurred in the normal course of business” (DCCEE, 2010a, p.9). Inadequate assessment of additionality is a common flaw in carbon sequestration schemes (Trexler, 2011).

As carbon dioxide (CO_2) emissions reside in the atmosphere for 300+ years[3] (Archer, 2005), carbon sequestration must be enduring if it is to offset them. Consequently, to be effective, policies need to ensure the ‘permanence’ of sequestration.

‘Leakage’ refers to GHG emissions that occur as a result of activities undertaken to mitigate or offset GHG emissions. The degree of leakage needs to be quantified and set against the benefits of an activity when its eligibility for payments is considered. The leakage from sequestering CO_2 may occur in another location, time, and/or form of GHG.

2.4 Additionality

When assessing additionality the following questions need to be addressed.

a) Is the sequestering practice additional?
b) If so, what is the ‘benchmark’ farming practice that it would displace?
c) How much of the abatement resulting from the new practice is additional?

Much of the discussion about additionality is focused on identifying a) and b) (e.g., Woodhams et al., 2012). However point c) is equally challenging. It requires determination of the net level of sequestration (i.e., also accounting for leakage) for both the sequestering practice and the alternative it displaces. The requirement to answer

[3] 300 years captures the atmospheric lifetime of the majority of the CO_2 from an emissions pulse; in reality, the atmospheric lifetime of a CO_2 emission is best represented by a decay function with an immensely ‘long tail’, such that it continues to influence climate at a diminishing rate for millennia (Archer, 2005; Eby et al., 2009).

question c) means that the measurement and monitoring (and associated transaction costs) that are often considered onerous for the sequestering activity, are also required for the benchmark. Furthermore, c) can vary in space and time even if a) and b) are unchanged.

2.4.1 Approaches to determining additionality

The economics of land uses are highly heterogeneous between (and even within) farms and over time. In principle, identification of the benchmark farming strategy and estimation of the opportunity cost of the sequestering practice might be done on a farm-by-farm basis using detailed bioeconomic models. In practice, this would have high transaction costs and would be hampered by poor information about each farm and each farmer's perceptions and preferences. Farm-by-farm modelling is not employed in the CFI because of concerns that the costs would discourage landholder participation (DCCEE, 2010a). A streamlined approach could potentially involve modelling for groups of farmers, by industry, by region or potentially by other factors. Whether the greater inaccuracy introduced by this approach would outweigh the savings in transaction costs would depend on the heterogeneity of farms and the process used to aggregate them (e.g., Antle et al., 2003; Capon et al., 2013). Regardless of if farms are modelled individually or in groups, economic models of farm businesses depend on subjective judgements about parameters and farmers' objectives (Robertson et al., 2012), and ultimately their results are indicative of 'financial' additionality, whereas the question is really one of 'behavioural' additionality (Meyers, 1999). Although economic motivations are important to most commercial farmers, their behaviour in terms of adoption of new or different practices is also influenced by a broad suite of social, cultural and personal factors (Pannell et al., 2006). Reasonably accurate determination of additionality would require consideration of both financial and social factors. This would no doubt increase transaction costs further.

An alternative approach for assessing additionality (with lower transaction costs) is the method used in the original CFI: using empirical survey data to determine whether a practice is common in a district. If, in the absence of a sequestration policy, a practice is undertaken by 5 to 20% or less of potential adopters it is treated as being additional (Woodhams et al., 2012).

This ‘common-practice’ approach to determing additionality has several limitiations. Firstly, it is subjective. The critical threshold of 5% adoption was choosen in the CFI because it approximates the ‘take-off’ point on a sigmoid-shaped adoption curve (Woodhams et al., 2012). However, it is not clear why this ‘take-off’ point is logically linked to additionality. It is essentially an arbitrary value.

Secondly, there is the issue of the scale-of-adoption or frequency-of-adoption. The common-practice approach treats adoption as a binary yes/no question. It ignores the possibility that farmers who have already adopted a sequestering practice on a small scale (or at a certain frequency) may like to respond to the policy by increasing their scale (or frequency) of adoption. For example, suppose that 30% of farmers have adopted a sequestering practice (exceeding the threshold for additionality) but each has adopted it on only 5% of their land. If they were eligible to participate in the sequestration program, they would adopt the practice on 50% of their land. The extra 45% adoption would be consistent with additionality, but would be ineligible with the common-practice approach.

The ineligibility of increased scale or frequency of adoption is concerning because it is likely to provide sequestration at relatively low cost. Practices rare in a district probably have higher marginal costs of adoption than ones already practised in parts of the district. And a practice that is already adopted on part of a farm is likely to have a very low marginal cost of increased adoption. This highlights an irony. Abatement practices with a low opportunity cost (the so-called ‘low-hanging fruit’) are more likely to be viable in their own right, making assessment of their additionality difficult and costly. Once these more expensive implementation and transactions costs are considered, the actual cost of utilising such abatement in a sequestration policy may be much higher (Fosci, 2013). Conversely those practices whose additionality is more easily determined are also likely to require larger incentives to be adopted. It is the latter practices that the common-practice approach seems more suited to.

A third limitation of the common-practice approach is that its main benefit—reducing the amount of costly analysis required to determine additionality—is only temporary. Once the program is in place, a new challenge emerges: estimating what the level of adoption would have been in the absence of the program. This information is required to ensure that practices that are common only because of the scheme, and which are therefore genuinely additional, remain eligible. However, if regulators are able to identify and quantify which instances of observed adoption are non-additional, then in principle there is no need to adopt the common-practice approach. Thus, reductions in transaction costs with the simplified common-practice approach will be short-lived. Either costly new research and analysis will be required to continue the program, or the accuracy of judgements about additionality will fall over time.

Fourthly, if it is judged that in the absence of the program, the level of adoption would fall below the specified threshold, then under the common-practice approach, all farmers who adopt the practice would be eligible to claim benefits in the program. Some of those claims will be for non-additional adoption, including all the adoption that occurred prior to commencement of the scheme. Conversely, once practices do exceed the threshold level of adoption and become ineligible, they are disqualified for all landholders, including those for whom adoption would have been genuinely additional. This results in a lesser supply of sequestration than would occur in an ideal program. To reduce these issues, Woodhams et al. (2012) proposed scaling estimates of sequestration down by the proportion of it adjudged to be non-additional, creating an ‘integrity buffer’ (McCarl (2006) proposed ‘additionality discounting’ based on a similar concept). Using this approach, a portion of the sequestration by a practice could remain ‘additional’, even though the practice is commonly-adopted overall.

Although this scaling approach appears attractive, it too has issues. Determining the appropriate amount of scaling requires regulators to do what they were trying to avoid: determine how much of the observed adoption is additional to what would have happened in an unobservable, counterfactual scenario. It is arguably also unjust. Farmers adopting a practice in response to the sequestration policy would have the amount of sequestration they could claim reduced, to subsidise the claiming of non-additional sequestration by those who had (or would have) adopted the practice without the policy. Lastly, scaling to compensate for abatement that is ‘lower quality’ (non-additional or otherwise) could also potentially be self-fulfilling, in that it could lower the quality of the abatement portfolio even more. By reducing the amount of income received for a sequestering practice, scaling reduces the incentive to undertake that practice. This is important. As incentives to adopt a practice decrease, the proportion of total adoption of that practice that is additional will decrease (Claassen et al., 2014). Suppose after scaling a landholder can earn only $5 for every hectare they undertake of a sequestering practice. This is most likely insufficient to induce many landholders to change practice, so nearly all those undertaking the practice and therefore eligible for the $5/ha would be those doing the practice anyway.

The Alberta scheme also uses a common-practice-type approach to additionality, with a threshold level of 40% adoption, and a scaled baseline, similar in nature to the ‘integrity buffer’ (Woodhams et al., 2012) and ‘additionality discounting’ (McCarl, 2006) concepts. When the scheme commenced in 2007, landholders *already* practicing reduced tillage were eligible to claim credits for the amount they were estimated to have sequestered over the previous five years. The justification for including non-additional adoption was that it created an incentive for landholders to maintain sequestration that could otherwise have been re-released (Government of Alberta, 2012). After claiming these initial credits, many smaller landholders are reported to have ceased involvement in the scheme because they felt it was not cost-effective (Climate Change Authority, 2014b). These landholders who ceased participating must have either: a) continued to practise reduced tillage but just not partaken in the scheme, as might occur if income from the offset scheme did not justify the transaction costs and the effort involved with participating in it, suggesting non-additionality or; b) also ceased practising reduced tillage, as might occur if income from the offset scheme was insufficient to cover the opportunity cost of altering tillage practices, in which case sequestration would be re-released.

Overall, there are serious limitations with the common-practice approach to determining additionality. Some (but not all) of those problems could be addressed by using good information about additionality, but if that information is available, the common-practice approach is not needed.

There are also some challenging aspects inherent to all approaches for assessing additionality. First, additionality must be assessed against a counterfactual benchmark situation that cannot be observed: the farming practices that would have been employed if the policy had never existed. As time passed following adoption of the sequestering activity, judgements about the unobserved benchmark would become increasingly difficult and speculative (Murray et al., 2007). Such uncertainty makes the determination of additionality potentially susceptible to political manipulation. Second, sequestration is a long process. After a change of practice, equilibration of soil carbon levels generally takes decades, and may take a century or more (Johnson et al., 1995). Conversely, additionality can be a transient quality. Therefore, regardless of the method used to assess it, assessments will have to be repeated or updated through time, meaning ongoing transaction costs.

2.4.2 Updates to additionality

Additionality is often mistakenly thought of as being a comparison of '*before* and *after*' the introduction of the policy when it is actually a question of '*with* and *without*' the policy. The 'before' situation is observable and fixed, whereas the 'without' scenario is unobservable and dynamic. To illustrate, suppose that a CFI-like policy with a common-practice approach to determining additionality had existed at the time that no-till was being adopted in Australia. Initially, early adopters could have claimed benefits for additional soil carbon because no-till was not commonly practised. It could well have been genuinely additional at this stage, as the knowledge and technology required to implement it profitably was still being developed. As no-till became more profitable, the sequestration it provides would have become less costly; at a certain point, it would become profitable enough not to require income from sequestration for widespread adoption and it would no longer be additional.

In time, adoption of no-till would have increased to around 90% of farmers (this is what happened without the policy) (D'Emden and Llewellyn, 2006). Clearly almost all of this adoption would be non-additional. Furthermore, although sequestration by early adopters was additional at the time, it was only temporarily additional because ultimately that sequestration would have occurred anyway without the policy. The GHG concentration in the atmosphere would eventually be the same with or without promotion of this practice by the sequestration policy.

If such a change in a sequestration practice's additionality is anticipated, it may make sense not to treat the practice as additional, even on a temporary basis. This would avoid spending money on actions that ultimately make little or no difference to climate change, and could otherwise have been spent on more effective mitigation. Sequestration practices that are 'cost-effective' because they are close to being viable in their own right (i.e., a low carbon price is sufficient to make the practice attractive to landholders) may often be those practices whose additionality is likely to change with time.

If a future change in additionality is not anticipated, but it becomes apparent that a practice once additional is no longer so, government appears to have two options: write-off the funds that have been lost (plus the climatic gains that could have otherwise been made from alternative investments), or require participants to bear the cost of the retrospective change in additionality (e.g., by purchasing replacement abatement). The latter option is unlikely to be politically feasible.

A second challenge related to the dynamics of additionality is that the benchmark agricultural practice may change even if the sequestering practice remains additional. Agriculture in the 20th century was characterised by rapid and dramatic improvements in production technologies, and large changes in prices, such that there would have been frequent changes in the benchmark practice in many cases.

To illustrate the potential consequences of this, suppose that a farmer replaces a benchmark practice that was neither storing nor emitting carbon with a sequestering activity that would accumulate an additional 13.2 tonnes of carbon over the next 35 years (areas *X* + *Y* in Figure 2.2(b)).

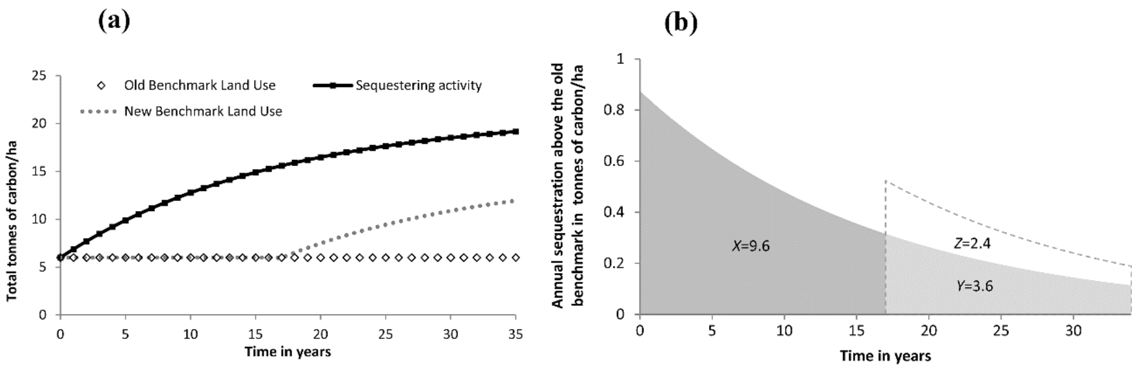

Figure 2.2. (**a**) Amount of carbon sequestered as a function of time. (**b**) The annual amount of sequestration above the old benchmark.

Imagine that after 17 years the landholder would have switched from the old benchmark practice to a new benchmark, had they not otherwise been engaged in the sequestration program. Suppose that this new benchmark also happens to sequester carbon—although this is not the reason it has become preferable to the old benchmark—and from years 18 to 35 it would have sequestered 6 tonnes of carbon (areas *Y* + *Z*). Notably, this is more than the claimed sequestering practice stores over years 18 to 35 (area *Y*), largely because sequestration occurs most rapidly shortly after a change in practice and then decreases over time. Therefore in principle, once the benchmark was updated, the farmer should have to relinquish offsets (or repay government payments) equivalent to area *Z*. However, like the previous example where a practice changed from additional to non-additional after a period of time, we anticipate that any requirement to repay payments for sequestration (undertaken in good faith but no longer additional) would be politically unacceptable. After all, the farmer would still be maintaining the sequestering practice, and the absolute amount of carbon in their soils would still be increasing by the amount shown in area *Y*.

These examples illustrate that in a theoretically-sound policy where participation is voluntary, landholders are not paid for the absolute amount of sequestration, but the additional amount sequestered relative to the benchmark. Consequently, a farmer's obligation should be to permanently maintain the claimed amount of sequestration over and above what would otherwise have happened (which may change with time), rather than the absolute amount of carbon that has been sequestered. This remains true even after claiming has ceased, and the stored carbon is being maintained for permanence reasons.

The need to revise additionality rulings over time because of the ephemeral nature of additionality is recognised in the literature (e.g., Gustavsson et al., 2000). However, it has not previously been acknowledged, either in the literature or by policymakers, that failing to apply these updates retrospectively to sequestration claimed in the past will reduce the benefits of sequestration, and result in an opportunity cost of having otherwise spent those funds on effective mitigation. In the original CFI, not only were updates to additionality not applied retrospectively, but sequestration could be claimed for another seven years after practices have been deemed to be no longer additional (DCCEE, 2012). This is clearly inconsistent with the principle of additionality.

Additionality is required with policies that provide financial rewards for mitigation (either by the sale of offsets or by government payments) and where participation is voluntary. It is not an issue for policy approaches that impose penalties on firms or individuals for not mitigating, by requiring them to pay a tax or purchase an emissions permit. Therefore, additionality is not an inherent requirement of sequestration *per se*. It is just that sequestering activities are implemented almost exclusively under the voluntary, financial reward type of policy framework, meaning that additionality is associated with sequestration almost as a matter of course. Furthermore with sequestration, ensuring additionality is made more challenging by the theoretical need to apply updates to additionality retrospectively. In contrast, the benefits of emissions reductions cannot be eroded in the same way, meaning retrospective updating is not an issue for them.

2.5 Permanence

To ensure permanence, sequestration schemes typically have rules that require sequestration to be maintained and not re-released for a certain duration or until replacement abatement is purchased (at the contemporary carbon price). The CFI's original permanence rules represented a combination of both: farmers were required to maintain sequestration for 100 years[4], but could opt out of the scheme before then by purchasing and surrendering permits (the 2014 revised CFI policy, based on government payments, introduced the option of a 25-year period for maintaining sequestration, with a discount on the payments).

The use of 100 years as the criterion for 'permanence' constitutes a gamble that climate change will have been solved by that time, so that the re-release of sequestered carbon will not be a problem. The gamble is increased because the physical dynamics of CO_2 in the atmosphere mean that the 100-year rule would result in higher future atmospheric concentrations of CO_2 than if there had never been sequestration. To illustrate, suppose that in 2015 *D* units of CO_2 emissions are mitigated either by offsetting them with sequestration, or by preventing/reducing emissions. The properties of the carbon cycle mean that a reduction in CO_2 has a diminishing effect on atmospheric CO_2 over time due to re-equilibration of carbon from the atmosphere with other sinks, particularly the ocean (Archer, 2005). As a consequence, releasing the sequestered carbon in 2115, adding *D* units of CO_2 to the atmosphere, raises the 'offset with sequestration' curve above the 'not mitigated' curve in Figure 2.3 (Kirschbaum, 2006 made a similiar observation). So although sequestration buys time for the '100-year gamble' to play out, if it turns out to be a losing wager, the approach could actually make the problem worse.

[4] 100 years after credits are first claimed. Subsequent claims made for the same project do not 'reset' the 100-year count, meaning that carbon sequestered later needs to be stored for progressively less time.

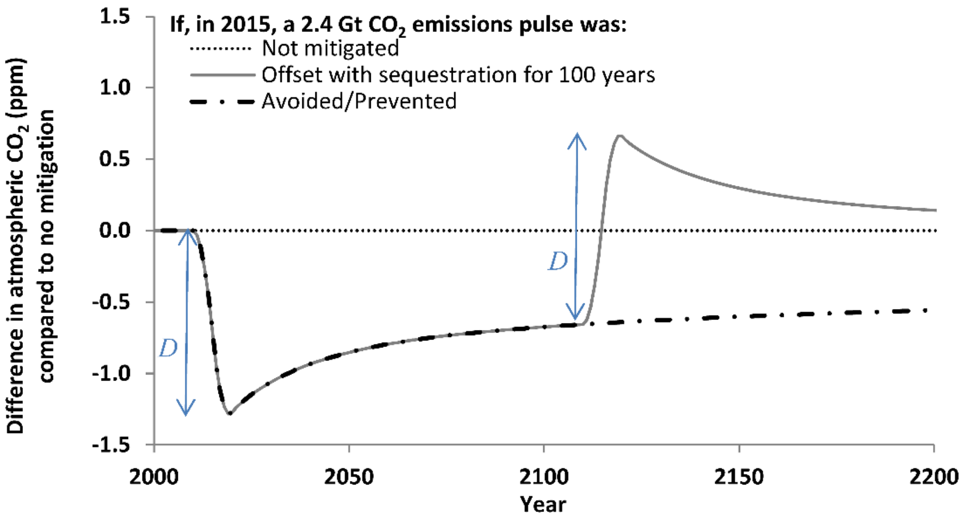

Figure 2.3. Implications of either i) offsetting with sequestration for 100 years or ii) preventing an emissions pulse, compared to not mitigating it at all (as predicted by the Model for the Assessment of Greenhouse Gas Induced Climate Change version 5.3 (Wigley et al., 2009), with the once-off emission pulse occurring against a background of the WRE450 emissions scenario).

The gamble can be avoided simply by not permitting the free release of sequestered carbon after what is essentially an arbitrary period such as 100 years. Instead, the concept of permanence would be applied literally. It is desirable to maintain the option of allowing participants to leave the scheme by purchasing replacement abatement as this would not sacrifice the GHG integrity of the program but would potentially increase participation.

Indeed encouragement of participation is one possible argument for the original CFI's 'free' release at 100 years. It might be seen as making permanence less onerous. However, for two reasons, the additional incentive for participation is likely to be minor. Firstly, the costs that would be saved by utilising this 'free' release are greatly reduced by discounting over 100 years (e.g., by more than 99%, at a discount rate of 5%). Secondly, there is a probability that the climate change problem will have been resolved by that time (e.g., through development of renewable energy technologies) so that the cost of emissions permits will have fallen greatly, potentially to zero. This outcome is, after all, what policymakers are relying on when they specify the arbitrary time frame of 100 years for 'permanence'. In that eventuality, the cost to participants from purchasing permits in order to leave the scheme would be lower still. In the alternative scenario, where climate change is unresolved, it seems preferable not to leave our descendants with a problem of large emissions of formerly sequestered carbon.

In contrast to sequestration, a reduction in emissions also buys time but without the risk of making the situation worse in the future (Figure 2.3). There are no questions over how long they should be maintained, nor any risk of re-releases exacerbating climate change in the future. It is therefore potentially inefficient for policies like the CFI to value emission reductions and less enduring sequestration equally (Feng et al., 2002; van Kooten, 2009; Gramig, 2012). Although there may be benefits from temporarily removing emissions from the atmosphere with sequestration, if this sequestration and permanent emissions reductions are both to be legitimately valued at the same carbon price then permanence requirements should have no time limit, and instead simply require landholders to relinquish replacement permits if they release previously sequestered carbon. Permanence provisions can interact with leakage to create further challenges, as outlined in the next section.

The Alberta scheme does not require landholders to maintain sequestration to ensure permanence. Instead, the amount of credits granted for an activity is scaled down by the probability that the sequestration might be undone in the next 20 years (7.5–12.5% probability, set by surveying experts) (Government of Alberta, 2012). We note that, if 7.5–12.5% is indeed an accurate reflection of the probability of losing the sequestered carbon in the absence of any mechanism to enforce its retention, then, by definition, it would appear that most of the sequestered carbon is not additional. That said, Roberts and Lubowski (2007) observed that a policy may partially influence land management beyond the life of the policy. A relevant example would be where reverting to the original practice is prohibitively costly (e.g., the cost of bringing land planted to trees back into arable cultivation). However, we suspect such that situations are less likely for agricultural practices sequestering carbon in soils than with conversion to forestry.

2.6 Leakage

Leakage can be categorised into two different forms:

1. ‘Indirect’—emissions resulting from substitutions or market adjustments occurring in response to the sequestration, potentially in other countries.

2. ‘Direct’—emissions directly resulting from the sequestration activities.

Indirect leakage can be significant, and has received considerable attention in the literature (e.g., Gan and McCarl, 2007; Lee et al., 2007; Montserrat and Sohngen, 2009; Sun and Sohngen, 2009; Alix-Garcia et al., 2012). Despite this scholarly attention, accounting for indirect leakage is very difficult when it comes to policy implementation, and so it tends to be ignored (as it is in the CFI). While not condoning this, we focus on direct leakage, which has received less attention in the literature. Perhaps this is because, at face value, it appears more straight-forward to address. However, there are challenges involved with developing policy to deal with emissions that occur as a direct result of adoption of a sequestering activity.

Many of the emissions directly associated with sequestration in soil result from agricultural activities. Agricultural emissions are dynamic, spatially heterogeneous, and difficult to measure. Methods for estimating them tend to be generic and may not reflect actual leakage at particular locations (Cowie et al., 2012; Thamo et al., 2013). Assessing them more precisely is possible, but involves higher transaction costs.

A second challenge with leakage is that there may be practices that appear desirable to landholders in the short term despite being undesirable in the long term. Consider the replacement of cropping land use with perennial pasture (e.g., Thomas et al., 2012). This land-use change was recently approved for crediting under the CFI (Department of the Environment, 2015c). Although stylised, the dashed line in Figure 2.4 is consistent with the pattern of increasing soil carbon following the conversion of cultivated crop land to permanent pasture (Sanderman et al., 2010). However, pasture is most commonly used to feed livestock, which emit methane, an important GHG. Suppose that the livestock grazing this pasture ‘leak’ 0.6 t/ha/year of carbon dioxide equivalents (CO_2e) (dotted line)[5]

Although the sequestration rate is initially much higher than the leakage, as it plateaus over 100 years it is eventually overtaken by the smaller, but constant, leakage (at around year 90 in Figure 2.4). The net level of sequestration would peak at about 30 years.

[5] Because they occur at the same location as the sequestration, many would classify these methane emissions not as ‘leakage’ but rather as part of the emissions balance of a sequestration project. However, for our purposes, whether they technically constitute leakage or not is immaterial. The key point is that these methane emissions are caused by the adoption of the sequestering practice and counteract it.

Unfortunately, it is not feasible to maintain the year-30 situation in subsequent years. If the pasture is retained (and grazed), leakage occurs more rapidly than ongoing sequestration. If the pasture is retained and not grazed, the opportunity cost of cropping income is likely to exceed the benefits of sequestration. If the pasture is converted back to cropping, to avoid further leakage, the sequestered carbon is re-emitted over time. In this situation, from a climate-change perspective, it appears preferable not to commence the sequestration activity in the first place. However, the exclusion of such practices from a scheme because they could be undesirable in the long term may not always occur for two reasons. Firstly, if the practice appears desirable in the short term it may be politically convenient to ignore longer-term undesirability. Secondly, even if the political will is present, determining long-term desirability of a practice *ex ante* may not be easy, as rates (and timing) of leakage may change in the future, and anticipated and actual measurements of sequestration may diverge.

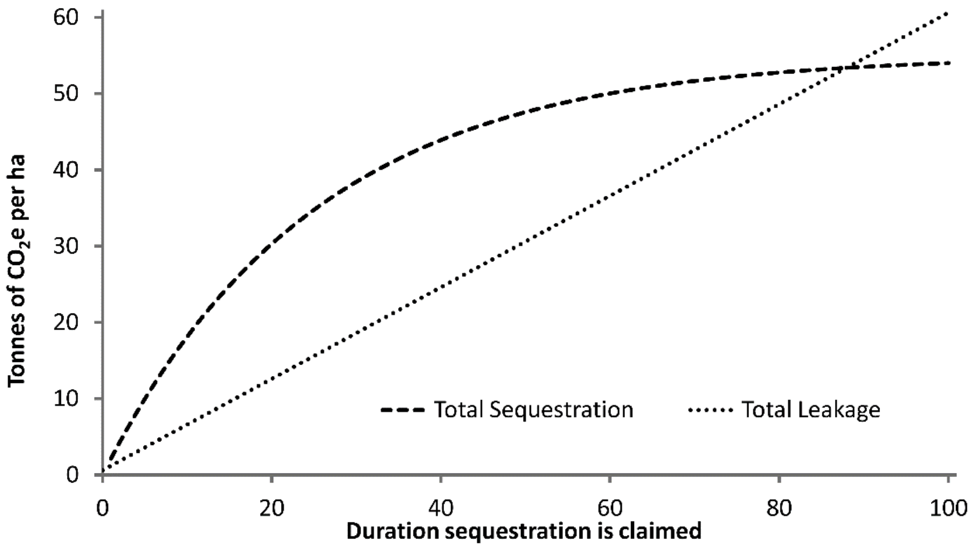

Figure 2.4. Cumulative sequestration and leakage as a function of the time credits are claimed.

The CFI legislation does not clearly define how sequestering practices that potentially generate more emissions than they sequester should be dealt with, although the legislation does allow some discretionary powers in these matters. Perhaps the simplest way for a policy to deal with this situation is to require leakage to be paid for when it occurs, at the contemporary carbon price. However, even if leakage is accounted for in this way, a sequestration practice that involves significant leakage could still appear financially attractive. This creates the possibility that landholders could be paid to undertake practices that, in the long run, leak more than they sequester, and that this leakage could be locked in by permanence conditions.

To illustrate, using the example from Figure 2.4, suppose that the price of offsets (or the government payment) is \$23/t of CO_2e (Australia's price in 2012), increasing at 2.5% per annum in real terms for the entire period, and that the real discount rate is 5%. Then claiming credits for more than 16 years would generate enough income to cover the expense of relinquishing credits for leakage over 100 years. From the farmer's perspective, under the conditions operating in the original CFI, the most profitable course of the action would be to continue with the sequestering practice (and associated leakage) until a 'free release' from permanence obligations could be obtained after 100 years. But implementing the project for this profit-maximising 100-year period would 'leak' 6.5 tonnes more CO_2e than it would sequester (Figure 2.4). In other words, despite eventually requiring more credits than it generates, the practice would be financially attractive for the farmer to pursue. This is because income from sequestration occurs mainly in earlier years, when sequestration rates are highest, whereas a larger share of leakage costs occurs later on and are therefore discounted heavily. This result is possible whenever the carbon price increases at a rate lower than the discount rate.

Importantly, potential participants in the program would need to expect that the carbon price will increase at a rate lower than the discount rate in order to be willing to participate. If they expect the carbon price to increase at a faster rate than the discount rate, then it would always appear optimal to defer commencement of sequestration until the future, when prices have increased by more than enough to outweigh the effect of discounting (this is because sequestration can occur only once: now or later). The landowner's optimal course of action would be to delay sequestration until the carbon price is increasing at a rate less than the discount rate (van 't Veld and Plantinga, 2005; McCarl and Sands, 2007).

Although these financial results provide only a stylised example, they do serve to highlight the potential risk for programs that accept sequestration activities with significant levels of leakage, particularly where the mitigation and resultant leakage occur over different timeframes. They may find that there are willing suppliers of offsets from these activities (or willing participants in a payment-based scheme), even where participants will be charged the going price for that leakage. Policymakers would be well advised not to include in programs any sequestration practices where leakage is likely to outweigh sequestration within a certain time frame.

Leakage can, of course, also be an issue for emission reductions. However, because emission reductions are not subject to permanence requirements, in their case there is less disparity between the timing of abatement and the associated leakage, meaning that the assessment of direct leakage is far simpler.

2.7 Other drawbacks of sequestration

Some argue that sequestration could provide an interim 'bridge' to the future, buying time for the development of improved technologies that allow lower-cost reductions in atmospheric CO_2 (e.g., Lal, 2002). However, there are reasons to doubt whether the benefits of this proposed 'bridging' are sufficient to outweigh the disadvantages of sequestration as an abatement strategy.

Firstly, a sequestration 'bridge' may be counterproductive by reducing incentives to innovate. The use of sequestration in the short term reduces the market price for emission permits (or in the absence of an emissions trading scheme, makes it easier to meet abatement targets), potentially reducing the incentive for early development of the required technologies (van Kooten, 2009; van Kooten and de Vries, 2013).

Secondly, carbon prices will need to increase for some years, perhaps decades, so that they reach a level that provides the required incentives for abatement and innovation (Keeler, 2005; van 't Veld and Plantinga, 2005). However, as noted earlier, if the carbon price is rising faster than the discount rate there is a financial incentive to defer the commencement of sequestration (because it can store a finite amount of carbon) (van 't Veld and Plantinga, 2005; McCarl and Sands, 2007). There is an irony here, in that a rapidly increasing carbon price, which might be thought to signal a need for urgent action, would actually discourage early sequestration. The same is not true of emissions reductions.

Thirdly, as we have demonstrated, leakage and/or the re-release of sequestered carbon may increase the amount of abatement that any future technologies will need to provide. This is particularly the case for programs that allow re-emission after an arbitrary fixed period of sequestration. Similarly, future changes in climate may reduce the equilibrium amount of carbon a given practice will maintain sequestered (Hoyle et al., 2013). If this new equilibrium is less than the amount previously stored, then what was a carbon sink may become a carbon source due to the changed climate (Goetz et al., 2013). Given that GHG concentrations are more likely to be approaching ‘dangerous thresholds’ in the future than they are currently, any short-terms gains made by sequestration now may be a false economy.

Fourthly, in addition to abatement, social and environmental co-benefits are often cited as an additional impetus for developing sequestration policies. However, the externalities from reducing or preventing emissions may be equally positive (Elbakidze and McCarl, 2007). Furthermore, policies focused on maximising other externalities may not achieve mitigation very efficiently and, *vice versa*, policies that maximise sequestration may deliver little in the way of other benefits (Caparrós and Jacquemont, 2003). Therefore, whilst the potential of a sequestration policy to provide additional co-benefits may be important, the extent to which these benefits are likely to be delivered should be analysed carefully rather than assumed (Bradshaw et al., 2013), particularly as there is potential for purported co-benefits to be overstated by rent seekers.

Fifthly, beyond the economic challenges discussed here, quantifying the amount of sequestration is very challenging and also involves (potentially large) transaction costs (Subak, 2000; García-Oliva and Masera, 2004). In comparison, transaction costs for quantifying emission reductions are likely to be lower in many cases.

Lastly, we have focused on the challenges of designing policies that aim to achieve the sequestration that is considered to be theoretically possible. However, the mitigation theoretically achievable with sequestration has itself been queried in recent studies (e.g., Powlson et al., 2011; Lam et al., 2013; Robertson and Nash, 2013). Furthermore, how immediately the mitigation potential of sequestration could actually be realised has also been questioned (Sommer and Bossio, 2014), throwing doubt on the capacity for sequestration to provide a short-term ‘bridge’ to future reductions in emissions.

2.8 Conclusion

There are many challenges to designing sequestration policies to ensure that they achieve genuine mitigation in a way that is cost-effective. Although additionality, leakage and permanence are issues for all voluntary abatement policies, unique characteristics of sequestration mean that these issues are particularly difficult for sequestration. These characteristics include: that the amount of abatement achievable with soil sequestration from a piece of land is finite (unless the CO_2 is first re-emitted), after which there is no option but to reduce emissions; that sequestration occurs rapidly at first but then plateaus at a maximum level; that sequestered soil carbon can be re-emitted if the new management regime is not maintained; and that some sequestration activities result in leakage of other emissions, potentially over different timeframes.

To contain transaction costs in the CFI, additionality is assessed based on the proportion of the relevant population undertaking the practice. However, this 'common-practice' approach rules out what are likely to be the least costly, genuinely additional abatement measures: increases in adoption where there is already (without the carbon price) moderate adoption. These increases could be in the number of adopters or, for farmers who have already adopted to a certain amount, in the extent the adoption is practised. The common-practice approach to assessing additionality also unavoidably results in non-additional practices qualifying for benefits, and genuinely additional practices being excluded from them. The choice of the threshold level of adoption determines the balance between these two problems, but cannot overcome them.

Most importantly, the intended advantage of this approach (reduced transaction costs) is at best temporary. Once the scheme is in place, assessment of additionality requires estimation of what the land use would have been in the absence of the scheme. The common-practice approach provides no assistance with this counterfactual question. Indeed it would seem to require the sort of analysis of optimal farming practices that was meant to be avoided by use of the common-practice approach.

Finally, the counterfactual scenario changes over time, so the additionality of sequestration cannot be presumed to be permanent. Theoretically this means that additionality should not only be periodically re-evaluated, but any updates should also be applied retrospectively, although this is likely to be politically infeasible.

In relation to the requirement for sequestration to be permanent, our analysis provides a warning against defining permanence as a period of arbitrary length, such as 100 years, after which re-emission of sequestered carbon is permitted. Because of the dynamics of the carbon cycle, this may result in future atmospheric carbon levels being greater than they would have been in the absence of sequestration.

We show that, under plausible circumstances, a sequestering activity can ultimately ‘leak’ more GHGs than it removes from the atmosphere. It can be financially attractive to a landholder for this to occur, even if leakage is quantified and charged for at the contemporary price. Added to this is the risk that, once the initial period of rapid sequestration is over, requirements for permanence may lock in leakage at annual rates that exceed annual sequestration rates.

In the face of these challenges and potential perverse outcomes, policymakers essentially have three options. Firstly, they can implement stringent systems to ensure the veracity of sequestration. The high transaction costs associated with this option will make sequestration less competitive, and sequestration’s overall contribution to international action on climate change may be minor. Secondly, they can simplify the process, reducing its stringency, as appears to have been done in Alberta’s offset scheme and Australia’s revised CFI. This would no doubt be popular with farmer groups and polluters, both of whom would stand to benefit. Although it will boost participation, it will also be inefficient and the amount of *genuinely* additional mitigation from carbon sequestration in the long run may only be small. The third alternative is to exclude sequestration options that suffer from the challenges identified here. In considering these options, sequestration should be seen as a means to an environmental end and not an end in itself (Trexler, 2011). It should also be remembered that, once implemented, poor policy can be difficult to remove as it creates a group of beneficiaries with an incentive to lobby for its continuation.

Overall, our judgement is that the third option of excluding soil carbon sequestration activities from carbon abatement policy should be carefully considered. Although there undoubtedly are benefits to increasing the carbon content of soils, given the challenges, risks and potential for perverse outcomes and high transaction costs, soil carbon sequestration may not be an efficient approach to mitigating climate change, especially if the sequestration is used as a direct substitute for preventing or reducing emissions.

Where sequestration practices are associated with significant leakage, or where they are anticipated to have the potential to become very widely adopted even without policy support, their inclusion in abatement policy seems particularly ill-advised.

Chapter 3. Paper 2. Assessing costs of soil carbon sequestration by crop-livestock farmers in Western Australia

This paper has been published as:

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The candidate's overall contribution to the published paper was approximately 20%, as certified in the Statement of Student Contribution.

3.1 Preface

This papers presented in this and subsequent chapters make use of the whole-farm Model of an Integrated Dryland Agricultural System (or MIDAS). MIDAS has a very extensive history of use and development in Australia, with a number of different versions developed for different regions of Australia. This 'family' of MIDAS models has featured in dozens of publications (for example Morrison et al., 1986; Abadi Ghadim et al., 1991; Kingwell et al., 1995; Pannell, 1995; Bathgate and Pannell, 2002; Petersen et al., 2003b; Flugge and Schilizzi, 2005; O'Connell et al., 2006; Gibson et al., 2008; Kopke et al., 2008; Doole et al., 2009; Robertson et al., 2009; Finlayson et al., 2010; Monjardino et al., 2010; Robertson et al., 2010; Kingwell and Squibb, 2015). The version of MIDAS used in this thesis is based on the central area of the West Australian Wheatbelt. The model is currently written in Excel spreadsheets and is executed with Visual Basic for Applications code, linked to an external solver-algorithm. In this Excel file there are more than half a million cells that contain values. In the process of updating this Central Wheatbelt MIDAS for the analyses reported in subsequent chapters, the values in a large majority of these cells were reviewed. Despite it being an existing, 'off-the-shelf' version of the model, during this process more than 15 bugs and errors (some quite serious) were discovered and fixed. Many improvements were also made.

Assessing costs of soil carbon sequestration by crop-livestock farmers in Western Australia

3.2 Abstract

Carbon sequestration in agricultural soil has been identified as a potential strategy to offset greenhouse gas emissions. Within the public debate it has been claimed that provision of positive incentives for farmers to change their land management will result in substantial carbon sequestration in agricultural soils at a low carbon price. However, there is little information about the costs or benefits of carbon sequestration in agricultural soils to test these claims. In this study, the cost-effectiveness of alternative land-use and land-management practices that can increase soil carbon sequestration is analysed by integrating biophysical modelling of carbon sequestration with whole-farm economic modelling. Results suggest that, for a case study model of a crop-livestock farm in the Western Australian Wheatbelt, sequestering higher levels of soil carbon by changing rotations (to include longer pasture phases) incur considerable opportunity costs. Under current commodity prices, farmers would forego more than $80 in profit for every additional tonne of CO_2-e stored in soil, depending on their adoption of crop residue retention practices. This is much higher than the initial carbon price of $\$23.t^{-1}$ in Australia's recently legislated carbon tax. This analysis does not incorporate the possibility that greenhouse gas emissions may increase as a result of including longer pasture phases. Accounting for emissions may substantially reduce the potential for net carbon sequestration at low carbon prices.

Keywords: APSIM; Bioeconomic Modelling; Carbon Farming; Climate Change Mitigation; MIDAS; Soil Carbon Sequestration

3.3 Introduction

Agriculture contributes significantly to increased atmospheric levels of greenhouse gases—such as CO_2, CH_4 and N_2O—through, for example, direct emissions from livestock or fertiliser use; and emissions from carbon lost as a result of deforestation, changing cultivation, and arable cropping. It has been estimated that agriculture accounts for about 14 per cent of anthropogenic greenhouse gas emissions worldwide

(FAO, 2001). In Australia, the agricultural sector contributed 15 per cent to net national greenhouse gas emissions in 2009 (DCCEE, 2010c).

Farmers can mitigate greenhouse gas (GHG) emissions by altering their management practices. One of the carbon sinks that is receiving considerable attention is the amount of soil organic carbon (SOC) that can be stored in agricultural soils (e.g., Smith et al., 2001; Kimble et al., 2002; Ostle et al., 2009; Sanderman et al., 2010). Trees, grasses, shrubs, forbs and legumes fix carbon dioxide (CO_2) into organic carbon through the process of photosynthesis. Some of this carbon (C) becomes soil organic carbon (SOC) through above and belowground decomposition (Fynn et al., 2009). By changing agricultural practices, it is possible to increase the amount of carbon stored in the SOC pool. Changes in land-use patterns and agricultural practices can also affect the amount of C released back into the atmosphere. Typically, CO_2-equivalents are used to compare the global warming potential of different GHG. For soil carbon, 1000 kg C = 3.667 tonnes CO_2-e.

Lal (2004a) estimates the global SOC pool at more than twice the size of the atmospheric pool of carbon, and 2.7 times the size of the carbon pool in vegetation. The potential of the world's agricultural soils to offset global GHG emissions has been estimated at 5 to 15 per cent (Lal, 2004a). Garnaut (2008) estimated the potential carbon removal by soils on Australian cropped land at 68 Mt CO_2-e per year (compared to a potential 16 Mt CO_2-e emission reduction by alternative livestock management). Practices that farmers can adopt to reduce SOC losses from the soil, and/or potentially reabsorb (*sequester*) carbon in their soil include:

- Conservation tillage;
- Increased retention of crop residues or "stubble";
- Regrowth of native vegetation;
- Reduced frequency of fallowing;
- Conversion from annual to perennial crops or pasture;
- Grazing and livestock management: for example, intensive rotational grazing;
- Sowing improved grass species that produce more biomass.

(Conant et al., 2001; Desjardins et al., 2001; van Caeseele, 2002; Campbell et al., 2005; Desjardins et al., 2005; Hutchinson et al., 2007; Sanderman et al., 2010)

Various policy programs support soil carbon sequestration as a strategy to offset GHG emissions. For example, the American Clean Energy and Security Act includes provisions to establish incentive programs for agricultural activities that can sequester carbon in vegetation or soils (US Congress, 2009), while the recently proposed Australian Carbon Farming Initiative (CFI) aims to give farmers, forest growers, and other landholders, access to voluntary carbon markets (Parliament of the Commonwealth of Australia, 2011). In these voluntary markets, farmers can choose to sell carbon credits for additional CO_2 sequestered in vegetation or soils as a result of a change in land use or management practices. Carbon sequestration achieved under the CFI will be credited as abatement under the National Carbon Offset Standard (NCOS—Department of Climate Change, 2010).

From a biophysical perspective it is possible to store SOC in agricultural soils by changing management practices. However, it is likely that farmers will only voluntarily adopt new management practices to increase SOC stocks if those practices are economically profitable. Some SOC sequestration management may lower farm profits (e.g., when changing from a high-value annual crop to a lower-value grazed perennial), in which case incentive schemes may be needed to compensate farmers. Although it has been claimed that SOC sequestration can be achieved for payments between $8-10 (Australian dollars) through to $25 per tonne (Taylor, 2011), there is currently little research into the financial impacts of changed management on farming businesses.

Our objective is to assess the costs of changing rotations to increase SOC sequestration, under varying levels of crop residue retention. We conduct a whole-farm bioeconomic analysis that quantifies the trade-offs between farm profit and potential SOC storage. Because changing the farm’s crop-pasture mix and residue retention can considerably affect SOC sequestration (Luo et al., 2010b), we analyse SOC sequestration for a wide range of potential crop-pasture rotations. Our analysis is limited in scope, and does not account for the possibility that greenhouse gas emissions may increase with a change of rotations. Considering that livestock emissions from enteric fermentation play a large role in the agricultural emissions (Garnaut, 2008),a full analysis of potential profitability of carbon farming would need to account for greenhouse gas emissions as well as sequestration potential.

3.4 Background

Despite a great deal of scientific research (Smith et al., 2000; Follett, 2001; Kimble et al., 2002; Post et al., 2004; Ostle et al., 2009; Sanderman et al., 2010 and http://www.csiro.au/science/soil-carbon-research-program.html), substantial bio-physical uncertainties about the achievable rates of SOC sequestration remain. Accurate measurement of soil organic matter and statistically verifying changes in SOC stock is complex because of the many, and heterogeneous factors affecting SOC-sequestration (such as temporal variability in vegetation coverage and spatial heterogeneity in soil environments—Sanderman et al., 2010).

Estimates for total potential SOC-sequestration vary widely, with the greatest increase generally found for conversion of cultivated lands to grassland, and for retirement or restoration of degraded agricultural lands (Hutchinson et al., 2007; Smith et al., 2008). Using a global dataset, West and Post (2002) concluded that enhancing rotation complexity can sequester an average 200 kg $C.ha^{-1}.yr^{-1}$. Agricultural soils in Australia can potentially store additional SOC by changing crop rotations (estimated 50-510 kg $C.ha^{-1}.yr^{-1}$) or by moving from conventional to no-till (up to 770 kg $C.ha^{-1}.yr^{-1}$) (Luo et al., 2010b; Sanderman et al., 2010). The estimated SOC-sequestration potential for Australian soils is, on average, lower than potential sequestration of northern hemisphere soils due to a less favourable climate and edaphic constraints (Sanderman et al., 2010).

Most studies that have assessed the impacts of carbon farming on whole-farm profitability have tended to focus on tree plantings (e.g., Plantinga et al., 1999; Plantinga and Wu, 2003; Flugge and Abadi, 2006; Antle et al., 2007b; Kingwell, 2009; Polglase et al., 2011) and a minority on SOC (Antle et al., 2001; Robertson et al., 2009). In general, the studies showed that any substantial improvements in SOC would come at a significant cost to farm profits.

A number of authors assessed the potential and the costs of reduced- tillage for SOC sequestration (Manley et al., 2005; Kurkalova et al., 2006; Pendell et al., 2007). (Grace et al., 2010) estimated how many farmers would adopt carbon-sequestering practices under varying carbon contracts, in the south east of Australia. At a carbon price of $200 per tonne of C, contract participation rates for minimum and no-tillage were only 11 and 16% respectively. These low participation rates were not a result of carbon prices *per* *se*, but rather due to the large proportion of farmers that has already adopted reduced or no-tillage practices in Australia, even without carbon incentives (Kearns and Umbers, 2010). Because of additionality requirements in the CFI (see Discussion), analyses of changing to conservation tillage therefore have limited relevance for Australian broad-acre mixed farm systems. Our study will instead focus on the other main tools available to farmers to manipulate SOC; changing crop-pasture rotations and stubble (crop residue) retention rates.

Stubble (crop residue) management practices vary widely (Anderson, 2009; Llewellyn and D'Emden, 2010), with potential consequences for SOC sequestration rates (Chan and Heenan, 2005). Different levels of residue retention can affect SOC sequestration rates, and the effectiveness of residue management on SOC storage will vary between soils (Lal et al., 1998). Only one study has estimated the costs of SOC sequestration from residue retention. For corn-soybean systems in the Mid-West of the USA, Choi and Sohngen (2010) found that modest SOC gains can be achieved at relatively low carbon prices of US$2 to US$10 per tonne C. More SOC sequestration would require higher carbon payments.

The study described in this paper builds on the bioeconomic modelling approach demonstrated by Robertson et al. (2009) by linking a process-based biophysical model to a whole-farm economic model, to jointly assess the impacts of changed crop rotations and residue management on farm profit and SOC sequestration. We extend their study by considering changes in SOC over varying time frames and estimate the potential costs of SOC sequestration in terms of reductions in farm profit.

3.5 Methods

We use the APSIM biophysical model to estimate SOC sequestration under different crop rotations and varying residue retention rates (results are available online as ancillary material to this paper). These estimates are linked to the MIDAS whole-farm bioeconomic model for a representative farming system in Western Australia. The setups for the APSIM and MIDAS models used for this analysis are available upon contacting the first author.

3.5.1 Case study area

The bioeconomic model was developed for a representative farm in the Central Wheatbelt of Western Australia (Cunderdin—Figure 3.1). The region is one of Australia's main grain growing regions, producing nearly one-third of Australia's total bulk wheat exports (ABS, 2012). Recent grain yields for the Central Wheatbelt area average 1.6t.ha^{-1}, compared to 1.4 and 1.3t.ha^{-1} for Western Australia and the whole of Australia respectively (Hooper et al., 2011).

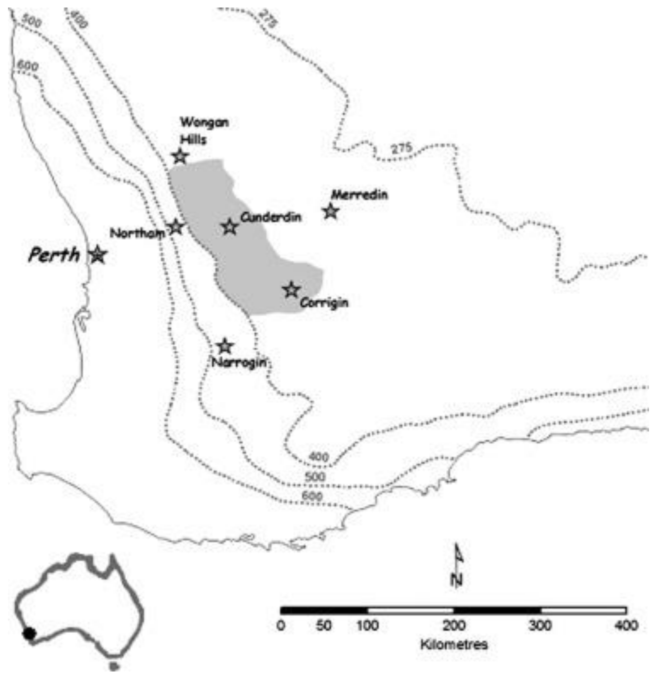

Figure 3.1. Location of case study area in the Central Wheatbelt of Western Australia. The dotted lines represent rainfall isohyets.

The region receives an average of 350-400 mm annually, with the majority of rainfall falling between May and October, during which time crops and pastures are grown. The weather is characteristic of the Mediterranean climate in south-western Australia with long, hot and dry summers and cool, wet winters. Farm size in the Central Wheatbelt varies from 1500 to 4000 hectare (average 2000 ha) comprising multiple soil types (Table 3.1). Typically, 20 to 70 per cent of arable land is sown to crops, which can be grown in rotation with lucerne and various annual pasture species (see Appendix). Nearly 90% of growers in the region have adopted some form of no-till or minimum tillage sowing techniques (Llewellyn and D'Emden, 2010). Sheep (mostly the Merino breed) are the dominant livestock enterprise, producing wool and meat.

Table 3.1. Soil types and areas included in the bioeconomic model (adapted from Kingwell, 2009).

Soil categories in the biophysical model*	Land management unit in the farm model	Farm area (ha)
Poor sand	Poor sands	140
Deep sand	Average sandplain	210
Loamy sand	Good sandplain	350
Loamy sand	Shallow duplex soils	210
Loamy sand	Medium heavy	200
Loamy sand	Heavy valley floors	200
Loamy sand	Sandy surfaced valley	300
Loamy sand	Deep duplex soils	390

* Soil categories were defined as follows: Poor sand = 55mm of plant available water capacity to 250cm; Deep sand = 93mm plant available water capacity for wheat to 150cm; Loamy sand = 130mm plant available water capacity to 250cm.

3.5.2 Biophysical modelling

Although some Australian monitoring data is available on potential rates of SOC sequestration (e.g., Sanderman et al., 2010) field measurements are highly variable and confounded by soil types and climatic conditions of the study site. We therefore used simulation modelling, which can dissect the separate and interacting effects of management, soil type and climate, to estimate rates of SOC sequestration. The process-based model, APSIM (Agricultural Production Systems Simulator—Keating et al., 2003), is comprised of individual modules that simulate components such as soil water balance, soil nitrogen and SOC balance, surface residues, crop production, pasture production, and livestock production. It accounts for the interactions between increasing SOC levels and nutrient cycling through changes to the C/N balance, but does not incorporate other effects, such as changes in soil structure. APSIM predictions generally provide a satisfactory representation of observed SOC changes (Probert et al., 1998; Ranatunga et al., 2005; Luo et al., 2011).

APSIM was configured to produce annual output for crop grain yields and forage production, and SOC content (to a depth of 30 cm so as to conform with IPCC guidelines for C-accounting (IPCC, 2006a)). The simulations were conducted using the 120-year historical climate record for Cunderdin and so potential changes in future climatic conditions were not accounted for in the present analysis. Short-term and long-term trends in SOC were estimated by linear regression through the annual output for 10, 30, 50 and 120 years. This approach reduces fluctuations in results for SOC change induced by year-to-year, and seasonal variability associated with crop-pasture sequences. It is also an improvement upon the approach of Robertson et al. (2009) who looked at single year changes in SOC. A number of different regression models were estimated to determine the appropriate average carbon sequestration rates over time. Of these models, a simple linear regression provided a good model fit (minimum R^2 across crop-pasture sequences was 0.87).

The APSIM model was used to estimate SOC sequestration rate under a range of crop-pasture rotations. A total of 64 crop rotations were analysed, comprising combinations of wheat, barley, oats, canola, lupins, field pea, chickpea, faba bean, annual pasture and lucerne. Three representative soil types were simulated (Table 3.1). These three soil types corresponded to the eight land management units used in the farm model. The crops and pastures included in each rotation were simulated with representative fertiliser inputs at sowing so that long-term mean yields and forage produced were comparable to those assumed in the farm economic model for each land management unit. A number of the sequences included lucerne phases of lengths varying between 2 and 4 years. Lucerne leys were sown between May and June and removed in November. Annual pastures and lucerne were grazed whenever above-ground biomass exceeded 2000 kg/ha.

Predicted rates of SOC changes will depend on the initial levels of SOC in the soil. The initial SOC levels in each soil type are typical of sandy soils subjected to continuous annual cropping and pastures since clearing for agriculture: 0.9 per cent in the 0–10 cm surface layer, 0.3 per cent in the 10–20 cm layer, and 0.1 per cent in the deeper soil to 250cm.

Farmers in the Western Australian Wheatbelt may graze, burn, or bale crop residues to varying degrees following harvest. This can lead to different rates of SOC sequestration and different future steady-state levels of SOC (Lal et al., 1998; Chan and Heenan, 2005). To investigate how alternative rotations affect SOC-sequestration potential under varying crop residue retention levels, we ran the APSIM simulations for each rotation with a base-case for crop residue retention, and a ‘full-residue retention’ scenario which is expected to increase SOC sequestration. In the base case, 50 per cent of crop residues were removed at the end of each year, after the cropping season has finished, while no residues were removed in the full-residue retention scenario.

3.5.3 Farm modelling

The farm economic analysis was based on the whole-farm bioeconomic model MIDAS (Model of an Integrated Dryland Agricultural System—Kingwell and Pannell, 1987). MIDAS is a steady-state mathematical programming model that aims to maximise annual net profits. Profits are defined as farm income remaining after deducting all overhead and variable costs, plus depreciation and opportunity costs associated with farm assets (apart from land). The almost 2000 activities in MIDAS include crop-pasture rotations on each of eight land management units (Table 3.1); crop sowing opportunities; feed supply and feed utilisation by different livestock classes; yield penalties for delays to sowing; cash flow recording; machinery and overhead expenditures (Kingwell, 2009). Constraints on the availability of land, labour and capital are also included in the model.

One of the major strengths of MIDAS is its ability to incorporate a range of costs and benefits at a whole-farm scale. The model takes into account the effect of changes in the farming system by considering its integrated impact on various factors affecting farm profitability such as weed control costs, fertilizer requirements, machinery requirements, labour costs, nitrogen fixation by legumes, and crop disease effects. Because of the limited biophysical evidence and biochemical uncertainties about the relationships between soil organic matter and crop production (Baldock and Nelson, 2000), the model does not quantify possible changes in crop productivity due to increased SOC levels (i.e., the model does not ascribe any production benefits due to the level of SOC *per se*).

The model was run to analyse farm profits at base commodity prices plus four additional scenarios (Table 3.2). We constrained the percentage of pastures in the enterprise mix as a way to perturb this mixed farming system. This approach represents the current practical focus of Australian Carbon Farming policies which aim to stimulate management changes (rather than achieving some target level of abatement). Crop-percentage curves are also commonly used as a sensitivity analysis in MIDAS modelling. The MIDAS model selects the combinations of rotations that maximise farm profit on each land management unit and thus provides information about the maximum annual farm profits that can be achieved for different crop-pasture mixes. These estimates are linked to the predicted soil carbon sequestration rates for the MIDAS-selected crop-pasture mixes to show the trade-offs between profit at varying cropping percentages and soil carbon sequestration. In calculating farm profit, payments for SOC sequestration are not included. We aim to quantify the trade-offs between profit and SOC sequestration, to estimate the likely sequestration response of farmers under different carbon prices.

The base case scenario assumes 50 per cent crop residue retention. A second scenario was run with the level of crop residue retention specified at 100 per cent. The model thus identified financially optimal rotations endogenously, while setting the level of residue retention exogenously. This strategy was adopted because residue retention is a "best-practice" management strategy that is widely adopted by farmers in Australia, and is therefore unlikely to satisfy the additionality requirements for carbon payments (see below for definition and discussion of "additionality"). However, it is not practised universally. According to Llewellyn and D'Emden (2010), around 22 per cent of farmers remove (a proportion of) their cereal residues through burning and grazing. It is therefore important to examine partial retention in the analysis, and to assess the SOC benefits of increasing retention rates.

Table 3.2. Price scenarios used in the farm modelling (FOB price).

Commodity	Price scenario				
	Base prices*	Low crop	High crop	Low sheep	High sheep
Wheat ($/t)	314	235	392	314	314
Barley ($/t)	348	261	435	348	348
Oat ($/t)	307	230	384	307	307
Lupin ($/t)	297	223	371	297	297
Canola ($/t)	582	437	728	582	582
Field Peas ($/t)	317	238	396	317	317
Faba Beans ($/t)	275	206	344	275	275
Chick Peas ($/t)	543	407	679	543	543
Wool (WMI, c/kg)	974	974	974	731	1218
Lamb ($/kg DW)	4	4	4	3	5
Ewes ($/hd)	54	54	54	41	68
Wethers ($/hd)	77	77	77	58	96

*2006-2011 average real commodity prices

3.6 Results

Following the methodology outlined in Robertson et al. (2009), APSIM predictions of annual rates of SOC sequestration were linked to MIDAS output, to evaluate the trade-offs between profit maximisation and the SOC storage potential under different rotation and residue management scenarios.

3.6.1 Base case—carbon sequestration rates and farm profit

In the base case scenario, SOC sequestration rates are simulated at 50 per cent crop residue retention and base commodity prices. The results for our typical Central Wheatbelt farm are shown in Figure 3.2, at varying constrained proportions of farm land allocated to cropping. The bar-graphs in Figure 3.2 show the potential rates of SOC sequestration for the profit-maximising combinations of crop-pasture rotations. Three different simulation periods are shown (10, 30 and 120 years).

When varying the area of the farm devoted to cropping, sequestration rates are highest when approximately 20 per cent of the farm's arable area is allocated to cropping, while the rest is devoted to pastures for sheep production. The predominant rotations in this enterprise mix are continuous pastures, pasture-wheat rotations or lucerne-wheat-barley rotations (see Table 3.3 in the Appendix). Perennial pastures contribute to high SOC sequestration rates. Over a 10 year timeframe, an average of approximately 217 kg of carbon could be sequestered per hectare per year. The predicted annual rates of SOC sequestration decrease over longer timeframes; to an average of 103 kg $C.ha^{-1}.yr^{-1}$ over 30 years and 76 kg $C.ha^{-1}.yr^{-1}$ over a 120 year period. These model predictions are in line with previous empirical measurements (e.g., West and Post, 2002; Luo et al., 2010b). The decline shows that SOC sequestration rates are highest in the first few years after a change in management, and decrease as the carbon stock increases.

When more land is used for annual cropping—wheat, canola, barley, or lupin-based rotations—SOC sequestration rates decline because much of the carbon-containing plant mass is removed via grain harvest (van Caeseele, 2002). For example, if 60–80 per cent of the farm was cropped, the average SOC sequestration rates over a 30 year period range between 57 and 44 kg $C.ha^{-1}.yr^{-1}$ for the profit-maximising mix of rotations (Figure 3.2).

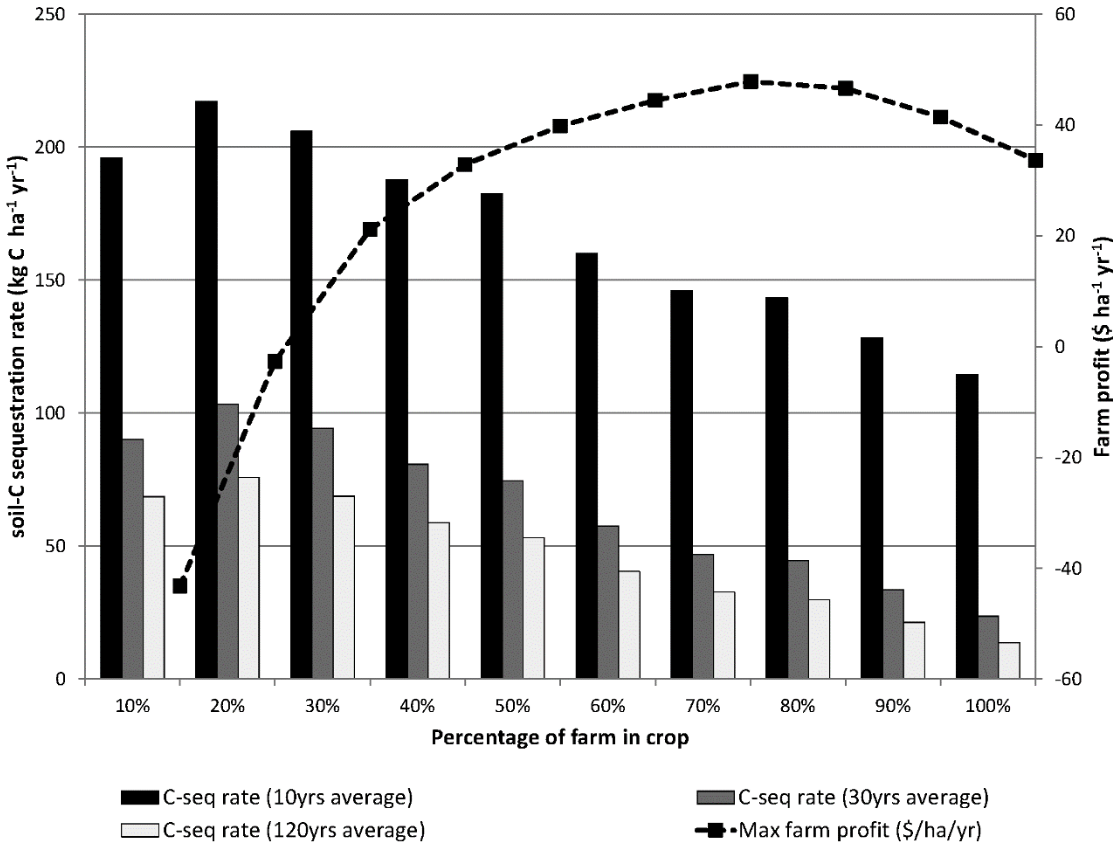

Figure 3.2. Maximum attainable profits ($\$.ha^{-1}.yr^{-1}$) and average SOC-sequestration rates in 0-30 cm soil over 10, 30 and 120 yr simulation periods (kg $C.ha^{-1}.yr^{-1}$) for profit-maximising enterprise mixes.

The MIDAS model provides information about the maximum attainable annual farm profit under optimal crop-pasture rotations. Under a base-case scenario, a farmer can maximise profit at about \$48 ha^{-1} by using approximately 70 per cent of the available land for cropping activities (Figure 3.2). The various profit-maximising rotations include annual or perennial pastures, cereal crops, and grain legumes (see Table 3.3 in the Appendix). Note that the representative farm comprises eight different land management units and that the selected cropping and pasture activities are selected for each soil type simultaneously to provide the most profitable farming system overall. Figure 3.2 illustrates that SOC sequestration rates decline when more than 20 per cent of the land is committed to cropping, while profit increases up to a maximum at about 70 per cent cropping. This highlights a potential tension between the optimal enterprise mix for farmers and policy objectives to increase SOC.

3.6.2 Profit - SOC trade-offs

The SOC sequestration rates predicted by APSIM were combined with the profit-maximising rotations selected by MIDAS to show the relationship between potential SOC storage and farm profit, varying the area of the farm constrained to growing crops (Figure 3.3). These results are based on the 30-year simulated average SOC sequestration rates. Although 30 years may be considered a short-term time period in a carbon sequestration context (where planning periods of more than 100 years are used—Parliament of the Commonwealth of Australia, 2011), a 30-year period is more appropriate from the perspective of generation-long farm management planning. The curves in Figure 3.3 show the trade-offs between potential SOC sequestration and maximum profits at 50 per cent residue retention and three price scenarios. Similar figures were generated for other simulation periods and price scenarios.

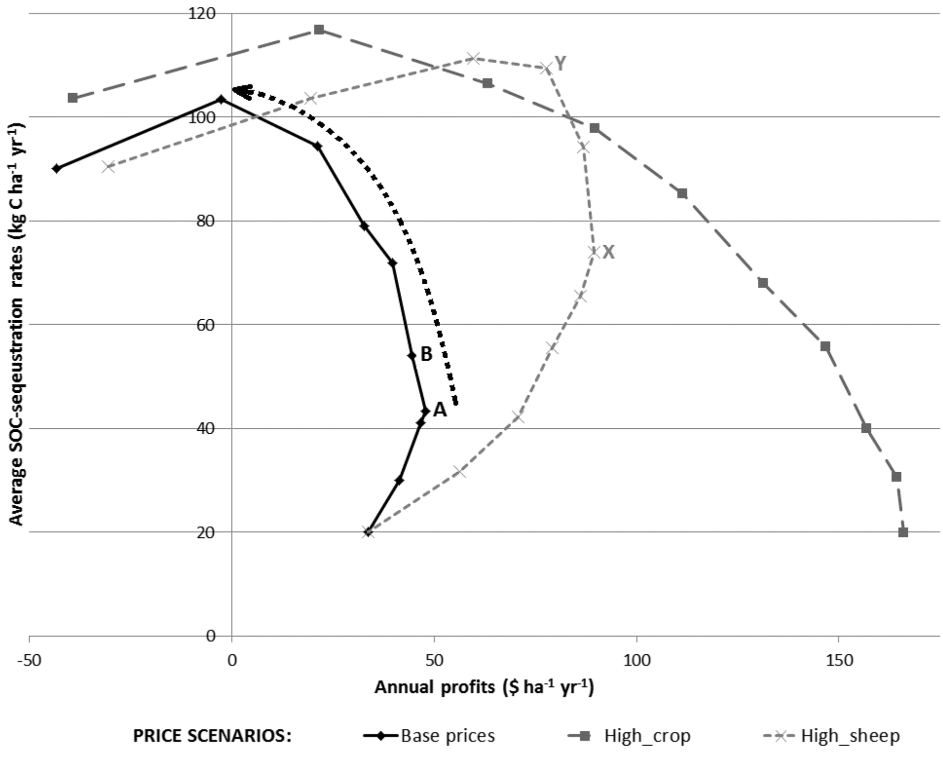

Figure 3.3. Trade-offs between annual profit and average SOC-sequestration rates in 0 –30 cm soil layer.

A change in enterprise mix to achieve higher rates of SOC-sequestration is likely to reduce farm profits. In Figure 3.3, this movement along the base-case tradeoff curve is indicated by the black dotted arrow. Different levels of sequestration require different levels of economic sacrifice, with the opportunity cost (in terms of reduced profits) tending to increase at higher rates of sequestration. Relatively small increases in SOC sequestration may be achieved at relatively low costs. For example, under a base-case price scenario, a profit-maximising mix of rotations would yield an annual farm profit of approximately \$48 ha^{-1}. Reducing crop area by 10 per cent below the profit maximising area (a movement from A to B in Figure 3.3) would reduce annual profits by only \$3.4 ha^{-1}—as would be expected given the flat payoff curve around the point of profit maximisation (Pannell, 2006)—while increasing the average SOC sequestration rate by about 10.7 kg $C.ha^{-1}.yr^{-1}$ (= 0.034 tCO_2-e). This means that the extra sequestration will cost the farmer approximately \$87 per tonne of CO_2 (as average reduced profits over 30 years). More substantial increases in SOC sequestration (moving further up along the curves in Figure 3.3) come at much higher cost. For example, a change in rotations from maximum profits to maximum SOC-sequestration rates (top of the curve) would reduce the annual farm profit by more than \$50 ha^{-1} under the base-case commodity price scenario. SOC sequestration rates would increase from 47 to 103 kg $C.ha^{-1}.yr^{-1}$, implying a cost per tonne of CO_2 sequestered of more than \$240. Our estimates should be considered as indicative values, given limitations of the model and data. Nevertheless, these SOC sequestration costs illustrate the limited potential for low carbon prices to drive sequestration of SOC in this farming system.

Prevailing commodity prices and costs will determine how much land is allocated to cropping to maximise farm profits. Increasing SOC sequestration rates requires the farmer to include more pasture-based rotations in their enterprise mix, and the costs of increased SOC sequestration will thus depend on a range of factors including commodity prices. We therefore analysed the sensitivity of our results to changing commodity prices, of which the high-price scenarios are presented here.
In a high crop-price scenario, a larger proportion of farmland will be allocated to growing crops, and the maximum attainable profit predicted by MIDAS, may be as high as \$166 per hectare per year. Under this price scenario, changing the mix of rotations to maximise SOC sequestration (i.e., limiting the amount of land under crop from 100 to 20 per cent) would considerably reduce farm profits—from \$166 to approximately \$22 $ha^{-1}.yr^{-1}$ at 50 per cent residue retention—while SOC sequestration rates increase by about 94 kg $C.ha^{-1}.yr^{-1}$ (= over 400 $\$.t^{-1}$ CO_2-e). On the other hand, when sheep prices are high, it will be profitable to commit more land to grazing. With more farm land devoted to pastures or lucerne rotations, the farmer can increase sequestration rates with a smaller reduction in profit. However, even under high prices for livestock products,

attempting to achieve sequestration rates over about 35 kg $C.ha^{-1}.yr^{-1}$ would cost more than \$190 t^{-1} CO_2-e (See Compensatory Payments Section below).

3.6.3 Impacts of residue retention

The above analysis shows the trade-offs between profit and SOC sequestration potential for different farm enterprise mixes and commodity price scenarios with the base-case scenario of 50 per cent residue retention. As noted earlier, varying levels of crop and pasture residues retention are observed in Australia (Anderson, 2009; Llewellyn and D'Emden, 2010). The level of residue retention may alter the cost-effectiveness of changing rotations as a strategy to increase SOC sequestration. Therefore, we also analysed a scenario where none of the crop stubble could be grazed or removed.

From a biophysical perspective, 100 per cent residue retention generally increased the amount of sequestration, because more organic material remained in the system where it could contribute to SOC. However, it also saw profits of the mixed-cropping livestock farm decrease because the crop stubbles—which represent a significant source of summer feed—were no longer available (Figure 3.4). The combinations of rotations at which a farmer can maximise profits are indicated by points C_{50} and C_{100} in Figure 3.4. A profit-maximising farmer who currently retains 50 per cent residue would store SOC at an average rate of 47 kg $C.ha^{-1}.yr^{-1}$ over a 30 year period (C_{50} in Figure 3.4). If this farmer were to move to full residue retention (C_{100}), SOC-sequestration rates could increase to more than 130 kg $C.ha^{-1}.yr^{-1}$. This indicates that, if residue retention were not already widely adopted, policies aimed at promoting residue retention could achieve significantly higher rates of SOC sequestration.

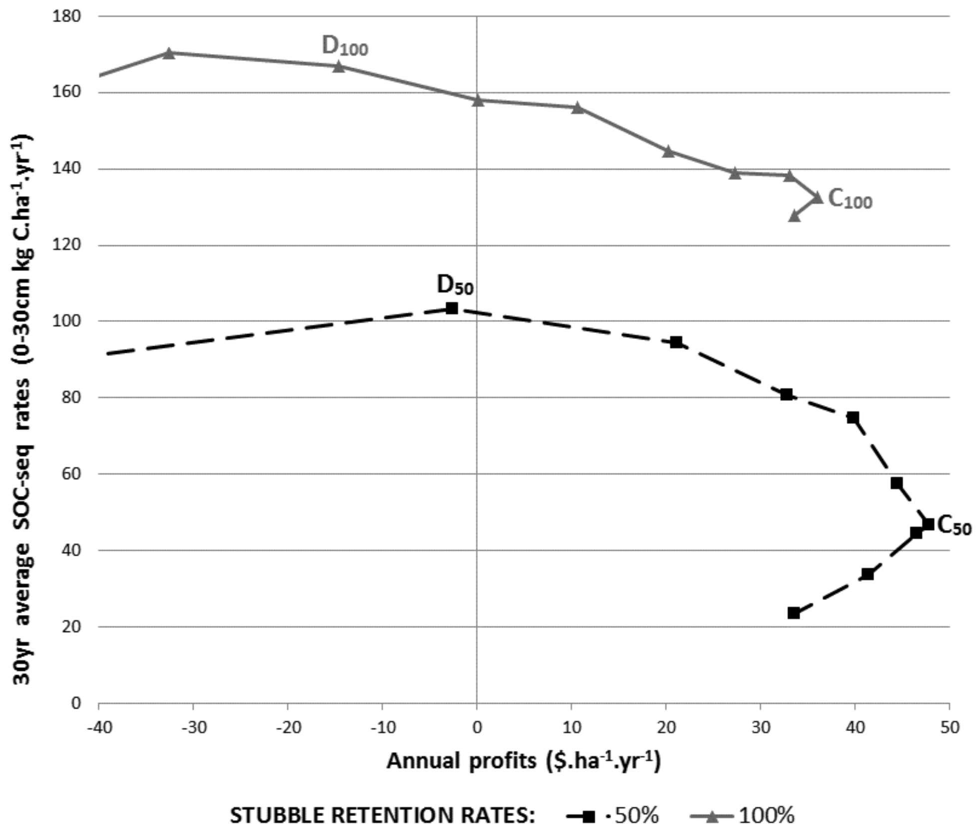

Figure 3.4. Trade-offs between annual profit and SOC-sequestration (top 30 cm soil, averaged over 30 year period) at 50% and 100% residue retention.*

*The points on each curve represent varying proportions of farm in crop. C = profit maximising mix of rotations, D = SOC-maximising mix of rotations

Consider a situation where a farmer has already adopted full residue retention practices, and is operating at point C_{100}. If this farmer were to increase the area of pastures to increase SOC sequestration close to maximum attainable rates (to the point indicated by D_{100}), profit would reduce by nearly \$51 $ha^{-1}.yr^{-1}$. Although this reduction is similar to a farmer who would move from C_{50} to D_{50}, the increase in SOC sequestration rates is distinctly lower. Under the base-case retention scenario, moving from 70 to 20 per cent cropping (C_{50} to D_{50}) would increase the annual rate of SOC sequestration by nearly 57 kg $C.ha^{-1}.yr^{-1}$. The same reduction in crop area would increase annual SOC-sequestration rates by only 31 kg $C.ha^{-1}.yr^{-1}$ under a full-retention scenario (C_{100} to D_{100}). Thus, at full retention, there is less potential to increase SOC sequestration rates through a change in the crop-pasture mix.

3.7 Compensatory payments

Given the trade-offs between increasing profit and increasing SOC sequestration, a profit maximising farmer is unlikely to change the enterprise mix to increase SOC sequestration unless compensatory payments are available. A voluntary carbon offset market could provide such payments. We calculated the annual incentive payments required to stimulate profit-maximising farmers to change their enterprise mix for increased SOC-sequestration rates. Given the discrete nature of our analysis (based on constraining the proportion of farm land allocated to cropping), the changes in profit and average SOC sequestration were calculated for a step-wise, 10 per cent, reduction in proportion of crop land. It is assumed that the farmer will initially operate under a profit-maximising mix of rotations (ignoring carbon payments). The annual payment p_{comp} required to compensate for the reduction in profits as calculated as:

$$p_{comp} = (\Delta\pi \,/\, \Delta SOC) \cdot 3.667\text{x}10^{-3} \qquad \textbf{Eq 3.1}$$

where $\Delta\pi$ is the change in annual profits, and ΔSOC is the average annual SOC sequestered in the top 30cm of soil in the first 30 years after a change in farm rotations (in tonnes per hectare). Since carbon prices are typically expressed in $ per tonne of CO_2-equivalents, results are multiplied by 0.003667 to convert sequestration from SOC to CO_2-equivalents.

Figure 3.5 shows the payments required to compensate for reductions in farm profit at three commodity price scenarios. The compensatory payments depend on the target level of SOC sequestration. For example, under a base-case scenario (Section 3.6.1), the offset payment required to achieve a maximum increase in SOC sequestration of an extra 60 kg C per hectare per year would be over \$240 t^{-1} CO_2-e. In the same base-case scenario, smaller increases in SOC-sequestration are feasible at a lower reduction in profit. Nevertheless, even a small increase in SOC sequestration of about 10 kg $C.ha^{-1}.yr^{-1}$ would still require payments of \$87 t^{-1} CO_2-e (at base-case prices). This is considerably more than the initial carbon price of $23 per tonne proposed in Australia climate policies (Garnaut, 2011).

The ‘flat’ areas along the curves in Figure 3.3 (e.g., the move from crop-pasture mix X to mix Y in the high-sheep price scenario) might suggest that large increases in SOC sequestration are achievable at low costs. However, the results indicate that the increase of approximately 20 kg $C.ha^{-1}.yr^{-1}$ would still reduce farm profits by about \$5 $ha^{-1}.yr^{-1}$. This equates to a compensation of about 35 $\$.t^{-1}$ CO_2-e (asterisk in Figure 3.5).

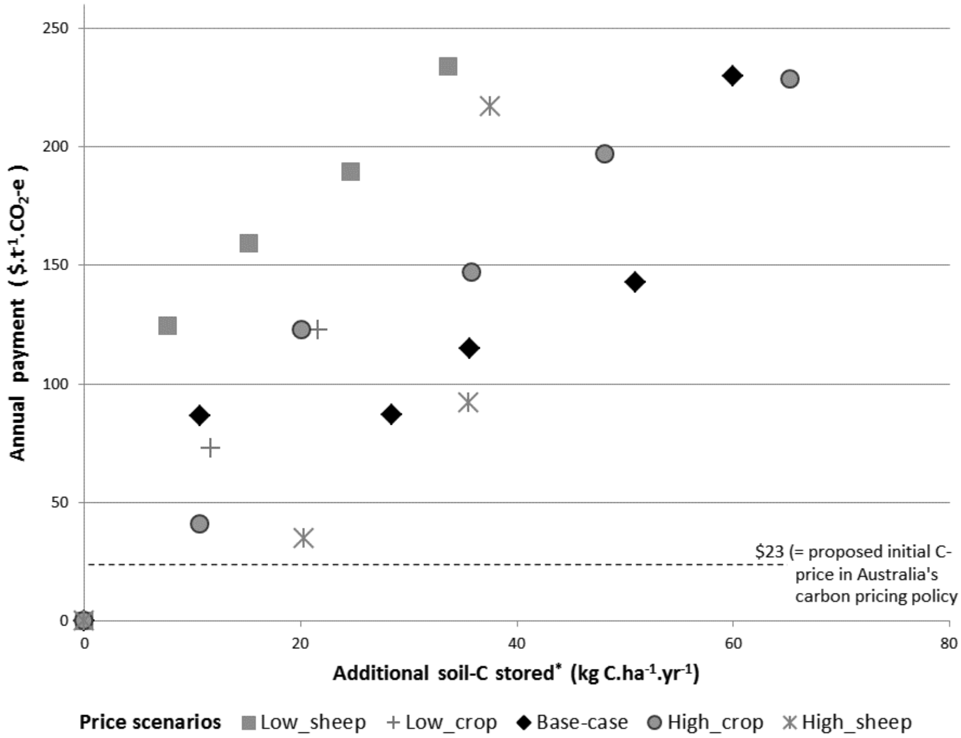

Figure 3.5. Carbon offset payments required to compensate for costs of additional soil organic carbon stored* under varying commodity price scenarios (at 50 per cent residue retention).

*Compared to carbon sequestration rate under a profit-maximising mix of crop-pasture rotations

The costs of sequestration could vary between farmers practising different rates of residue retention (Figure 3.6). The compensatory payments depicted in Figure 3.6 are for the base-case commodity prices, at base-, and full-residue retention. As discussed in Section 3.6.3, moving from 50 per cent to 100 per cent residue retention may reduce farm profit (by about \$12 $ha^{-1}.yr^{-1}$) but can increase SOC-sequestration rates (by about 86 kg $C.ha^{-1}.yr^{-1}$). This implies that \$38 t^{-1} CO_2-e would be needed to compensate this farmer for reductions in profit (Figure 3.6). However, there are notable differences in the sequestration rates that can be achieved by changing crop-pasture rotations given a certain level of residue retention. If profit losses would be compensated, less than \$100 t^{-1} CO_2-e could achieve up to about 27 kg C-sequestration per hectare under both the 50 per cent and 100 per cent residue retention scenarios. But increasing sequestration further (by changing rotations) will come at a considerably higher profit loss for the farmer who has already adopted residue retention.

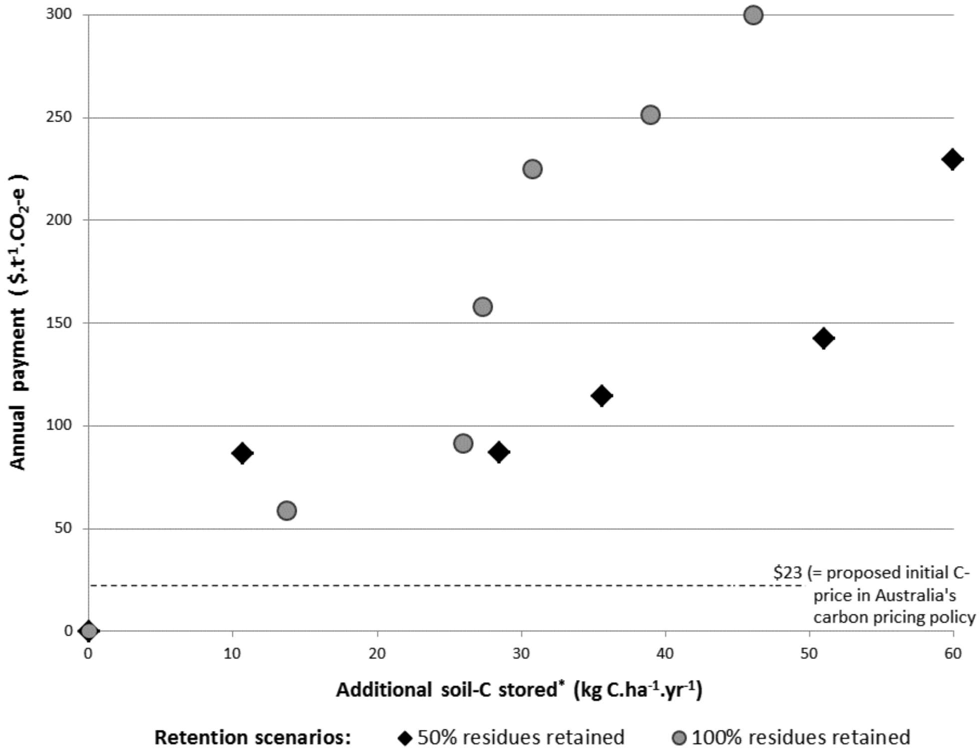

Figure 3.6. Carbon offset payments* under varying residue retention rates.

3.8 Discussion and conclusion

In this study, results from a biophysical model were combined with whole-farm economic modelling to assess the trade-offs between farm profit and SOC sequestration for a crop-pasture farming system in the Western Australian Wheatbelt. The results consistently show that increasing SOC-sequestration by changing crop-pasture rotations will reduce farm profit. Annual farm profits are maximised if approximately 70 per cent of the farm's available land is allocated to annual cropping. Under a base-case scenario, a profit-maximising farmer in the Western Australian Wheatbelt could make approximately \$48 $ha^{-1}.yr^{-1}$, and would sequester about 47 kg $C.ha^{-1}.yr^{-1}$ over 30 years in the top 30 cm of soil. Enterprise mixes with a larger proportion of pastures are associated with higher SOC sequestration rates, but generate lower farm profits than annual cropping. A farm with approximately 80 per cent of the available land under pasture could potentially sequester over 103 kg $C.ha^{-1}.yr^{-1}$, but would make a loss of about \$3 $ha^{-1}.yr^{-1}$. This indicates that changing crop rotations to increase the level of SOC will result in reduced profits to farmers in the study area.

The reduction in profit relative to carbon gains depends on prevailing commodity prices, input costs, and the target level of SOC to be sequestered. Under a base-case price scenario and 50% residue retention, increasing SOC sequestration rates by about 10 kg $C.ha^{-1}.yr^{-1}$ (compared to C-storage under the profit-maximising rotation mix) would cost the farmer approximately $87 per t CO_2-e. Under a scenario that favours a high percentage of the farm being in pasture—such as high commodity prices for livestock products—an increase in SOC sequestration may cost farmers less, but would still require a compensation of more than $340 per t CO_2-e, to store an additional 17.5 kg $C.ha^{-1}.yr^{-1}$. Given carbon prices discussed in the 2010/2011 Australian public debate never exceeded $30 per t CO_2-e this suggests that the potential to mitigate emissions through SOC sequestration is likely to be limited in this farming system.

The relative increase in SOC as a result of changing farm enterprise mix is affected by residue retention rates. SOC-sequestration rates are greater at higher rates of residue retention. Based on this analysis, one could argue that policymakers should stimulate farmers to retain a higher proportion of residues to achieve higher SOC sequestration rates. However, paying farmers to adopt residue retention may be inconsistent with the current proposed criterion for “additionality” in Australia. Additionality is a key feature of most carbon policies, and involves a requirement that the activity creates additional sequestration / reductions in emissions than would have occurred under a ‘business-as-usual’ scenario. Previous studies have shown that a large proportion of farmers have already adopted residue retention systems (Kearns and Umbers, 2010; Llewellyn and D’Emden, 2010). Therefore, increasing residue retention rates may not satisfy the “additionality” criterion.

A similar point could be made regarding increased proportions of pastures or perennials in the farm enterprise mix. To be eligible as a genuine offset, the activity must not be common practice in the region (Parliament of the Commonwealth of Australia, 2011). Given the variation in crop-pasture mixes between farms and regions, it is still uncertain (at the time of writing) under what conditions increasing pastures would be recognised as an additional practice under the CFI.

A number of issues should be considered when interpreting our results. First of all, the current analysis does not incorporate how different crop-pasture mixes affect agricultural GHG emissions. It has been estimated that current livestock production contributes nearly 42 per cent of Australia’s total rural GHG emissions (Sparkes et al., 2011). Although increasing annual pastures in the enterprise mix will enhance SOC sequestration, the subsequent increase in the number of sheep on a (profit-maximising) farm will significantly increase GHG emissions generated through enteric fermentation and animal waste (Kingwell, 2009). Such an increase in emissions is likely to be classed as ‘leakage’ under the current Australian policy proposal—and should accordingly be deducted from any sequestration gains. Further work is needed to compare the emissions associated with agricultural production against soil carbon sequestration potential. GHG emissions would include those generated by livestock through enteric fermentation and animal waste; fertiliser emissions; nitrogen fixing crop emissions; crop residue emissions; and fuel emissions produced during crop establishment, harvest, chemical and fertiliser application (Kingwell, 2009). A second important issue is that soil carbon sequestration may require application of additional nutrients (e.g., nitrogen and sulphur) to allow the carbon to be stored in a stable form (Kirkby et al., 2011). Nutrient application in the form of fertiliser would involve additional cost that would further reduce the economic attractiveness of the sequestration activities.

Readers should bear in mind that the estimated sequestration potential depends largely on assumptions about soil types and climatic conditions. The analysis presented in this paper is based on a representative bioeconomic farm model for the Central Wheatbelt of Western Australia; a crop dominant and fairly dry Mediterranean agricultural zone, with low SOC soils. Different soil types, farming systems, or climatic conditions in other cropping regions in Australia will affect the predicted SOC-sequestration rates. Moreover, Western Australia is predicted to experience adverse impacts of future climate change (Ludwig and Asseng, 2006). Negative effects on plant production can reduce inputs of organic matter in the soil, and thus reduce SOC sequestration potential. Further work is required to assess the impacts of possible adverse climate change on SOC and the changes in farm profitability under such conditions.

Changing farm management to increase SOC-sequestration will only be eligible for offset payments if activities represent permanent abatement. The proposed Australian Carbon Farming Initiative stipulates that a farmer who participates in a carbon offset market will be obliged to maintain the higher level of SOC for 100 years (after the last year that credits were claimed—Parliament of the Commonwealth of Australia, 2011). These long planning periods are likely to increase the level of risk and uncertainty to participants in a carbon offset scheme. Commodity prices are likely to vary considerably over a 100-year period, which means that the potential reduction in farm profit is highly uncertain. This, combined with the irreversibility that participation may involve, will generate an option value from delaying participation. While uncertainties in costs and prices can be challenging for farmers, additional factors that may impose a risk on the farmer who has entered into a carbon contract include: climate change or natural disasters that could reduce or re-release SOC in the atmosphere; possible changes of the policy program sometime in the future; and future technology developments that could either mitigate climate change effects more cost-efficiently than SOC sequestration or that could raise the opportunity cost to farmers of participating in SOC enhancement. It is not unrealistic that the combination of the 100 year maintenance period and these uncertainties will reduce the preparedness of farmers to adopt activities that enhance SOC, such that greater incentives may be required to achieve SOC sequestration than those estimated here. To design an effective and cost-efficient carbon offset scheme, research is needed into the farmer’s evaluation of the risks involved with participation in an offset market and the potential losses in option values, in light of a variable climate, changing commodity prices, and different carbon offset payments.

The current analysis considers the impacts of changed management on farm profits through changes in production costs and revenues. It is likely that participation in a carbon offset scheme will yield additional costs that are not directly associated with agricultural production, such as learning, transaction, monitoring, and reporting costs. Such additional costs are not included in the current model and are likely to present additional barriers to adopting carbon farming practices.

3.9 Appendix

Table 3.3. Profit-maximising crop-pasture rotations selected in MIDAS in the base price scenario.

Proportion of farm-land in crop	Most profitable rotations (allocation proportions varying per soil-type)†
0%	No feasible solutions
10%	PPPP, PPPW, 3UWB
20%	PPPP, PPPW, 3UWB, WWF
30%	PPPP, PPPW, 3UWB, WNWL, WWF
40%	PPPP, PPPW, 3UWB, WBL, WNWL,
50%	PPPP, PPPW, 3UWB, WNWL, WWBK
60%	PPPP, PPPW, 3UWB, WBL, WNWL, WWBK
70%	PPPP, PPPW, 3UWB, NWBLD, WBL, WNWL, WWBK
80%	PPPP, 3UWB, NWBLD, WBL, WNBF, WNBK, WNWL, WWBK
90%	PPPP, 3UWB, NWBLD, WBL, WNBF, WNBK, WNWL, WWLD
100%	NWBLD, WBL, WNBF, WNBL, WNBK, WNWL, WWLD

†3U = 3 years lucerne; B = barley (*Hordeum vulgare*); F = field pea (*Pisum sativum*); K = chick peas (*Cicer arietinum*); L = lupin (*Lupinus angustifolius*); LD = dry sown lupin; N = canola (*Brassica napus*); P = annual pasture; W = wheat (*Triticum aestivum*).

Table 3.4. Selected rotations on different soil types and their modelled 30 year average soil C-sequestration rates (kg C.ha^{-1}.yr^{-1} in 0–30 cm topsoil) by soil category.

	C-sequestration (kg C.ha^{-1}.yr^{-1})		
Rotation ‡	**Poor sand**	**Deep sand**	**Loamy sand**
NWBLD			20
PPPP	157	157	76
PPPW			55
3UWB			129
WBLD			25
WBL		53	
WNBLD			19
WNBF			23
WNBK			-1
WNBL			19
WNWL		49	18
WL		62	
WWBF			24
WWBK			-1
WWLD	57		26
WWL		62	26
WWWF			24

‡ B =barley; F =field pea; K =chick peas; L =lupin; LD =dry sown lupin; N =canola; P =annual pasture; W =wheat

Chapter 4. Paper 3. Dynamics and the economics of carbon sequestration: common oversights and their implications

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The candidate's overall contribution to the published paper was approximately 70%, as certified in the Statement of Student Contribution.

Dynamics and the economics of carbon sequestration: common oversights and their implications

4.1 Abstract

Accurate assessment of the cost of carbon sequestration is important for the development of mitigation policies globally. Given that sequestration in soils or vegetation is a lengthy process, such assessment requires financial discounting, and making realistic assumptions about changes over time in the rate of sequestration, the price of carbon, and the opportunity cost incurred by adopting sequestration practices. Our objective is to demonstrate how these assumptions affect estimates of the cost of sequestration-based mitigation strategies. Using an Australian case-study of soil carbon sequestration, our estimates of the carbon price required for financial viability are highly sensitive to dynamic assumptions, varying by a factor of four with different assumptions. Yet the influence of these time-related assumptions is poorly acknowledged in the literature, with many studies either failing to disclose their assumptions, or employing questionable assumptions and methods. Recommended global strategies are for researchers to report their assumptions related to dynamics much more transparently and to improve their research methods and the realism of their assumptions when analysing the economics of carbon sequestration. We recommend that policymakers become better aware of the issues created by dynamics, so that they are able to validly interpret assessments of the cost of sequestration and to ensure that they design policies in a way that facilitates fair comparison of the costs of mitigation strategies that operate over different timescales.

Keywords: carbon sequestration, economics, dynamics, discounting, carbon price, assumption, time

4.2 Introduction

Policies or schemes to provide financial incentives for sequestration of carbon in soil or vegetation have been proposed, planned or implemented by many governments worldwide. Examples of existing programs include the Offset Credit System in Alberta (Government of Alberta, 2015), or Australia’s Emission Reduction Fund (a re-named extension of the previous Carbon Farming Initiative scheme, Department of the

Environment, 2015a). To assess the appropriate balance between emission reductions and sequestration, policymakers need accurate estimates of the cost of each.

As the physical process of sequestering carbon can take many years, the cost of sequestration cannot be estimated without making (implicitly or explicitly) assumptions about the fate over time of three key factors: (i) the price of carbon; (ii) the opportunity cost of diverting land from its current use to one with higher carbon sequestration; and (iii) the rate of sequestration on land that has been converted. It is often assumed that each of these factors will remain unchanged over time, but there are often good reasons to expect that they will not.

The objective of our analysis is to demonstrate how assumptions about the dynamics of the above three factors can greatly affect estimates of the cost of sequestration. We provide evidence for this and draw out the lessons and implications for policymakers and researchers, namely that assumptions regarding the dynamics of these factors need not only to be realistic but be also expressed transparently. Because many existing analyses of the cost of sequestration fail in this regard, their contribution to the global literature is of less value to policymakers than it should be.

4.3 Methodology

4.3.1 Theoretical background and literature survey

Consider a change in land use that sequesters *X* tonnes of carbon dioxide (tCO_2) in either soil or vegetation over a given period of time (*T* years). The opportunity cost of this activity—profit that would be foregone if the current land use was ceased—is \$*Z*/ha per year. Changing land use would be attractive to the land manager if she can claim carbon credits for the sequestration with a value that exceeds the opportunity cost. A key question is thus: What is the minimum price of carbon credits ($\$/tCO_2$) that would make sequestration no less profitable than the existing land use?

We term this minimum price the breakeven price. Estimates of the breakeven price depend on what is assumed to happen to the sequestration rate, opportunity costs, and carbon price over *T* years. To examine what is typically assumed about these dynamic matters in the global literature, we surveyed 32 existing studies that have sought to estimate such a breakeven carbon price (these studies, which were conducted across five continents, are summarised in the Appendix provided at the end of this paper).

4.3.1.1 Rate of Sequestration

Existing studies calculate credits for sequestration based on either:

a) a constant (i.e. average or linear) sequestration rate (59% of the 32 studies reviewed); or

b) a dynamically-varying sequestration rate (41% of studies)

Figure 4.1 illustrates the difference between these two assumptions. Rates of sequestration in soil and vegetation are often highest soon after a sequestration activity has commenced, declining over time as the system approaches a new steady state (Silver et al., 2000; Ingram and Fernandes, 2001; van ’t Veld and Plantinga, 2005; Harper et al., 2007; West and Six, 2007; Lam et al., 2013). Consequently, the dynamic rate of sequestration in any particular year can differ appreciably from the constant (average) rate over the full term, particularly at the beginning and end of the sequestration period.

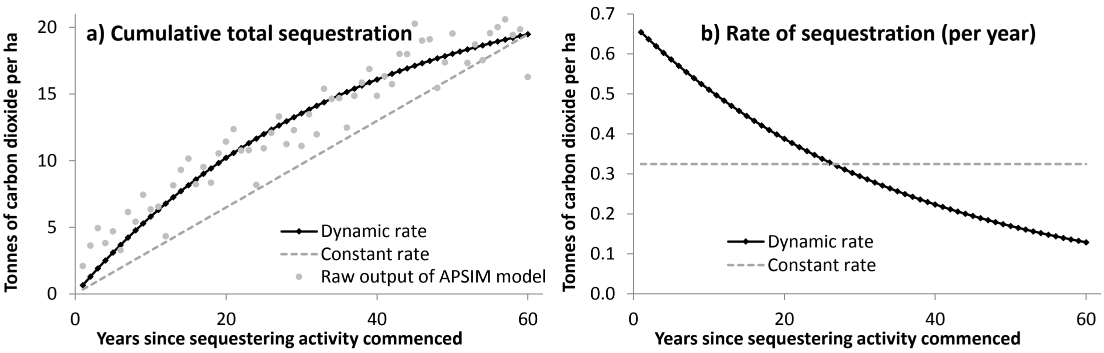

Figure 4.1. a) Total accumulation of sequestered carbon and **b**) the annual rate of sequestration (the first derivative of the curves shown in a)). The dynamic rate of sequestration is initially faster but then falls below the constant or average rate. The grey dots in a) show the raw output of the Agricultural Production Systems sIMulator (APSIM) model.

4.3.1.2 Opportunity cost

In the literature, the opportunity cost incurred by changing land use is often assumed to remain constant (59% of studies reviewed). However, only 6% of the studies reviewed clearly state whether the opportunity cost was constant in real terms (after allowing for inflation) or nominal terms (not adjusted for inflation). Distinguishing between nominal and real terms is important because it can make a large difference to the opportunity costs assumed for later years of the study period. In addition, 38% of studies did not state what they had assumed about the fate of the opportunity cost into the future at all, meaning that for 91% of studies, results cannot be properly interpreted or compared.

4.3.1.3 Carbon price

Very few (9%) of studies reviewed fully disclosed their assumptions about carbon price dynamics. This included two studies that assumed the price of carbon would increase in real terms over time (at least initially), and a third study that assumed that the carbon price would remain constant in real terms. Of the remaining studies, about half assumed a constant carbon price, but failed to disclose whether this was constant in nominal or real terms, and the other half did not state their assumption about carbon price dynamics at all. Across the literature, only two studies were fully transparent about all three of their dynamic assumptions.

4.3.2 Process for determining the economics of sequestration

Our procedure for determining the breakeven carbon price includes four steps. We use these steps to investigate the effect of the assumptions from Section 4.3.1. Twelve combinations of assumptions are examined: constant or dynamically-varying sequestration rates; opportunity costs that are constant in real or nominal terms; and carbon prices that are constant in nominal terms, constant in real terms, or increasing in real terms. Although the mathematical equations to use when performing the first three steps below differ depending on the combination of assumptions being made (Table 4.1), the steps are otherwise identical for all combinations of assumptions:

Step 1: Calculate what the price for a tonne of carbon dioxide will be in year i (CP_i), as dictated by the assumed dynamics of the carbon price, given an initial price at the commencement of sequestration (CP_0).

Step 2: Multiply the CP_i by the mass of carbon dioxide stored in year i (C_i). Depending on which type of sequestration rates is assumed, this C_i could be constant or dynamically-varying. Convert the result into present value terms by discounting and sum for all years i to T, where T is the duration for which sequestration credits are to be claimed. This produces the C_{PV}, the total present value of all income that the sequestration in year i will generate.

Step 3: Convert this total present value into the annuity C_A, which is a constant annual payment over the T_{TOT} years that the sequestration activity has to be implemented for (i.e., the time over which the opportunity cost will be incurred). T_{TOT} may or may not be equal to T, depending on the permanency requirements of the relevant sequestration policy. Whilst the annuity is not the actual income from sequestration, receiving it for T_{TOT} years would be equivalent to receiving the income from sequestration (i.e., the total present value of the annuity would be the same as the C_{PV} calculated in Step 2).

Step 4: Numerically solve for the CP_0 that produces an annuity equal to the \$*Z*/ha/year opportunity cost. This CP_0 is the breakeven price; at prices above it, the income from carbon credits over T exceeds the present value of the opportunity costs incurred over the total period T_{TOT} so a change in management to the sequestering practice is financially attractive.

Table 4.1. The equations required to calculate the viability of a sequestration activity when using discounting and annualising[a].

Combination of assumptions		***Step 1*** $CP_i =$	***Step 2*** $C_{PV} =$	***Step 3*** $C_A =$
Opportunity cost constant in nominal terms	Carbon price constant in nominal terms	CP_0		
	Carbon price constant in real terms	$CP_0 \times (1 + r_{INF})^i$	$\sum_{i=1}^{T} \frac{C_i \times CP_i}{(1+r_N)^i}$	$C_{PV} \Big/ \left[\frac{1-(1+r_N)^{-T_{TOT}}}{r_N}\right]$
	Carbon price increases at r_{CP}	$CP_0 \times [(1 + r_{INF}) \times (1 + r_{CP})]^i$		
Opportunity cost constant in real terms	Carbon price constant in nominal terms	$\frac{CP_0}{(1+r_{INF})^i}$		
	Carbon price constant in real terms	CP_0	$\sum_{i=1}^{T} \frac{C_i \times CP_i}{(1+r_R)^i}$	$C_{PV} \Big/ \left[\frac{1-(1+r_R)^{-T_{TOT}}}{r_R}\right]$
	Carbon price increases at r_{CP}	$CP_0 \times (1 + r_{CP})^i$		

[a]Where CP_0 is the initial price of a tonne of carbon dioxide when sequestration commences (\$/tCO_2); r_{CP} represents the real rate of carbon price increase, r_{INF} is the inflation rate, r_R and r_N are the real and nominal discount rates respectively (all in % p.a.); C_i is the mass of carbon dioxide stored in year i (tCO_2/ha); CP_i is the price of a tonne of carbon dioxide in year i (\$/tCO_2); T represents the years sequestration credits are claimed for, T_{TOT} is years that the sequestration activity has to be implemented; C_{PV} is the total present value of all income generated from sequestration (\$/ha); C_A is an annuity or constant annual payment (\$/ha)

Implicit in the steps above are the assumptions that credits for sequestration are claimed and sold at the end of each year at the contemporary carbon price and, for simplicity, that there are no transaction costs. In Table 4.1, r_{CP} represents the assumed real rate of carbon price increase, with r_R and r_{INF} denoting the real discount rate and the inflation rate respectively. The nominal discount rate (r_N) is calculated as follows (Pannell and Schilizzi, 2006):

$$r_N = (1 + r_R)(1 + r_{INF}) - 1 \qquad \textbf{Eq 4.1}$$

Our methodology employs nominal and real terms consistently throughout the calculations; where opportunity cost is assumed to be constant in nominal (real) terms, everything in the analysis, including the annuity, is expressed in nominal (real) terms. This means the analysis can be conducted without having to adjust the values of future opportunity costs. Alternatively one could still assume a nominally-constant opportunity cost but express it, and perform the analysis in real terms (and *vice versa*). This would require different formulas to those in Table 4.1, but if analysed correctly the results would not change.

4.3.2.1 Time-value of money

The process outlined above utilises financial discounting and annualising techniques. Discounting reduces the weight given to costs and benefits the further into the future they occur, thereby assigning a time-value to money. A principal reason for discounting is to account for the opportunity cost of money that has been spent (or not received) over time, such as the loss of income that could have been earned had that money instead been invested in another financial investment. Additional reasons for discounting include: to reflect how people typically prefer benefits that occur sooner rather than later; to allow for a predicted increase in wealth in the future, resulting in lower marginal utility from additional income and; to reflect risk and uncertainty surrounding the future (Pannell and Schilizzi, 2006; Arrow et al., 2013). Despite discounting being a mainstay of economic analysis, 30% of the studies we reviewed did not properly take the time-value of money into account. This echoes Boyland's (2006) observation that about a third of studies of the economics of sequestration in forests do not take into account the timing of when the benefit occurs. Instead these studies tend to use an averaging approach, where the opportunity cost associated with adopting a sequestering practice (\$*Z*/ha/year) is multiplied by the duration of sequestration (*T* years), and the result divided by the total carbon sequestered (XtCO_2/ha). We demonstrate in our case-study below that this approach can lead to erroneous estimates of the breakeven carbon price.

4.3.3 Case-study

The methodology (Section 4.3.2) is applicable to any sequestration practice; the case-study we apply it to involves soil carbon sequestration in the Central Zone of the Western Australian Wheatbelt. Farming systems in this 9,503km^2 study area are purely rainfed, with no irrigation (average annual rainfall ranges from 280mm to 500mm). Cropping is the main enterprise (Lawes and Kingwell, 2012). Crops are generally sown with minimum tillage practices, and typically receive 30-50kg/ha of nitrogen fertiliser. On the sandy-duplex soil types (soils with a sandy surface-horizon overlaying a clay-rich subsoil or World Reference Base for Soil Resources Arenic Solonetz soil) of this area a common crop rotation is the sequence of canola (*Brassica napus*), wheat (*Triticum aestivum*), barley (*Hordeum vulgare*) followed by lupins (*Lupinus angustifolius*). However, the carbon content of the soil can be raised if crops are instead grown in sequence with pastures. Hence farmers who increase the amount of pasture in their rotations are eligible to receive sequestration credits under Australia's Carbon Farming Initiative/Emissions Reduction Fund (Department of the Environment, 2015c).

An example of an eligible land-use change would be switching from the continuous-cropping rotation described above to a rotation with a three-year phase of perennial lucerne (*Medicago sativa*) pasture followed by two years of wheat crop. Using the biophysical Agricultural Production Systems sIMulator (APSIM) model (Keating et al., 2003) we estimate that, over 60 years, this land-use change would sequester an additional 19.5tCO_2/ha in soil. The output of the APSIM model is shown in Figure 4.1a. This raw output was then smoothed by fitting the equation: sequestration in tCO_2/ha = $24.08 \times (1 - e^{-0.028.\text{years}})$ ($R^2 = 0.94$). As the sequestering rotation is less profitable than the common rotation, adopting it would incur an opportunity cost. What this opportunity cost will be in the future is one of the dynamic assumptions we need to make, but using the whole-farm bioeconomic Model of an Integrated Dryland Agricultural System (or MIDAS, Kingwell and Pannell, 1987) we estimate that currently it is AU$12/ha/year (during 2001-2015, AU$1 was on average equal to €0.64 and US$0.80). This opportunity cost comprises the net revenue lost when the amount of cropping in the rotation is reduced, minus the net revenue gained from the pasture. These net revenues represent averages for a hectare of the rotation (e.g., for 1 hectare of the continuous-cropping rotation it represents the net revenue for 0.25 hectares each of canola, wheat, barley and lupins summed).

Permanence rules in the original Carbon Farming Initiative legislation required a sequestering practice (in this case, the new rotation) to be continued for 100 years (Thamo and Pannell, 2016). Only after this can the sequestered carbon be released without penalty. Therefore although credit claiming for sequestration ceases after 60 years (T = 60), the implementation of this management change (and bearing of the associated opportunity cost) must continue for another 40 years so as to not re-release sequestered carbon (i.e., T_{TOT} = 100).

For our case-study we use a real discount rate of 5% and an annual inflation rate of 2%. However, we also test real discount rates of 3% and 7%, and lastly we treat the discount rate as an uncertain parameter with a discrete distribution of discount rates: 3%, 5% or 7% with probabilities of 25, 50 and 25% respectively (for each discount rate, we calculate a stream of discount factors, which are applied to the benefits and costs, and then weighted by the appropriate probabilities). This is to explore Weitzman's (1998) insight that an uncertain discount rate has similar effects as a discount rate that declines over time.

4.4 Results

4.4.1 Assumptions about dynamics

Depending on the combination of assumptions made about the dynamics of the sequestration rate, opportunity cost, and carbon price, the breakeven carbon price varies by a factor of almost four, from \$14/tCO_2 to \$53/tCO_2 (Figure 4.2). Clearly, results are highly sensitive to these assumptions.

Across the literature surveyed, the most common approach is to assume that carbon credits will be granted at the constant (average) annual rate of sequestration over the sequestration period, and that the opportunity cost and carbon price will both remain constant at its initial value (e.g., Antle et al., 2001; Petersen et al., 2003a; Lewandrowski et al., 2004; Diagana et al., 2007; Popp et al., 2011; Kragt et al., 2012 etc.). We use this as our reference approach, with the further assumption that the opportunity cost and carbon price are both constant in nominal terms. Under this reference, the breakeven initial carbon price would be $38/t$CO_2$ for our case-study.

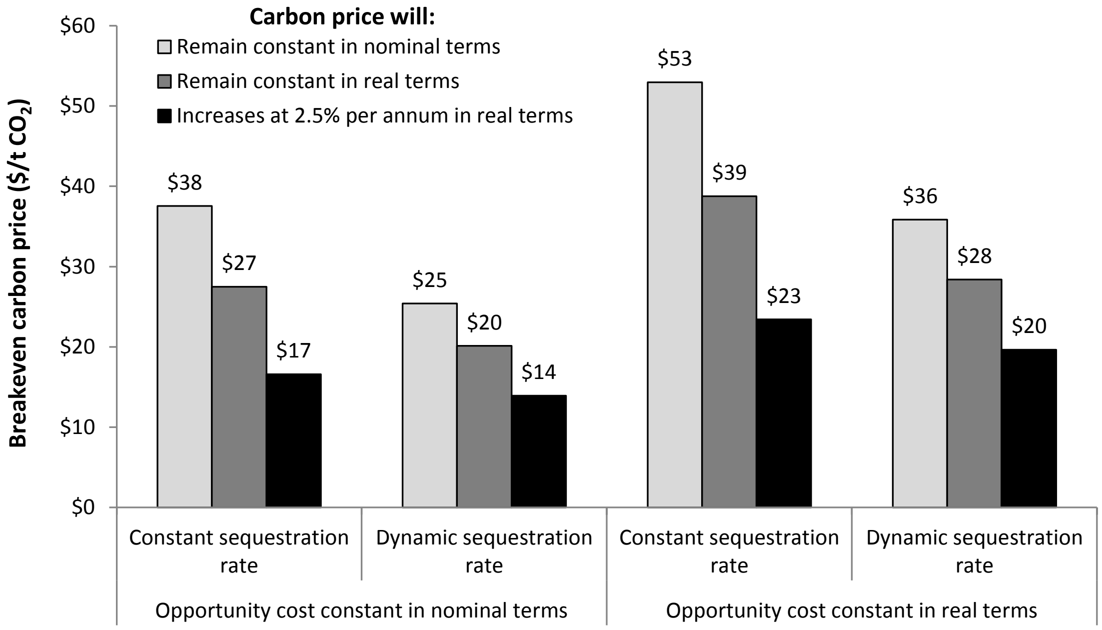

Figure 4.2. The carbon price required (initially at time zero) for the sequestering activity to breakeven differs notably between different combinations of assumptions about the sequestration rate, opportunity cost and carbon price (using 5% real discount rate).

If, instead, we assume that the carbon price is constant in real terms, the breakeven carbon price falls from $38 to $27/t$CO_2$. If we assume that the carbon price increases at 2.5% per annum in real terms, the breakeven initial carbon price falls further, to $17/t$CO_2$. In fact, across the four possible combinations of assumptions about the opportunity cost and the sequestration rate (the four groups of results in Figure 4.2) assuming a constant real carbon price reduces the breakeven price by 21-27% compared to assuming a constant nominal carbon price, and assuming the initial carbon price increases at 2.5% in real terms reduces the breakeven initial price by 45-56% relative to constant nominal price. What will actually happen to future prices is unknown, but, in our judgement (and that of others e.g.,van 't Veld and Plantinga, 2005), if effective climate policies are enacted then increasing, or at least constant, real carbon prices are most likely in the short to medium term. Under that scenario, using a constant nominal carbon price will substantially overestimate the breakeven price, making the sequestering activity appear less financially attractive than it should.

Now consider the opportunity cost. Starting from the $38/t$CO_2$ reference scenario, switching from constant nominal to constant real opportunity cost raises the breakeven initial price by 41% to \$53/tCO_2. In considering which assumptions to use, we will need to judge whether a constant real or constant nominal opportunity cost (or, indeed, another dynamic pattern) is more likely to be realistic, taking into account future demand for agricultural commodities. Compared to the constant-real assumption, assuming a constant nominal opportunity cost substantially under-estimates the breakeven price.

Finally, consider sequestration dynamics. If the opportunity cost and carbon price remain constant in nominal terms then a switch from a constant to a dynamic sequestration rate reduces the initial breakeven price by 32%, from \$38/tCO_2 to \$25/tCO_2. Across all combinations of assumptions in Figure 4.2, switching from a constant to a dynamic sequestration rate reduces the breakeven carbon price by between 16-32%. This occurs because more of the income is received earlier with a dynamic rate, meaning the income from credit sales is less affected by discounting.

The results in Figure 4.2 were obtained with a real discount rate of 5%. In Table 4.2 we conduct a sensitivity analysis of different discount rates. A higher 7% rate may be more realistic for a commercial landholder with many profitable investment opportunities and thus a higher opportunity cost of capital; the lower 3% rate may be more relevant to a public landholder. Results show that the breakeven price is relatively insensitive to the discount rate for 10 of the 12 combinations of dynamic assumptions. The exceptions are the two combinations where a constant real opportunity cost is combined with a constant nominal carbon price; for these cases the breakeven price is \$14/tCO_2 larger with a 3% discount rate than with a 7% rate. As the discount rate increases (i.e., as the future is discounted more) assumptions about the future have less impact. Therefore, overall, there is greater variation in breakeven prices between the different combinations of assumptions when the discount rate is 3% than when it is 7%.

Table 4.2 also shows that using an uncertain real discount rate (with a mean of 5%) causes slightly higher breakeven prices relative to a certain rate of 5%. This is because the lower discount rate in the discrete distribution (3%) has more influence on the results than does the higher rate (7%) (Arrow et al., 2013). We will not be examining uncertainty in the discount rate in the remainder of the paper, since it makes so little difference in this case-study.

Table 4.2. Breakeven initial carbon price (in Australian Dollars per tonne of carbon dioxide) with three certain and one uncertain discount rates.

Sequestration rate	Opportunity cost	Carbon price	Real discount rate (p.a.): 3%	5%	7%	Uncertain Discount Rate[a]
Constant rate	Constant nominal	Constant nominal	$39	$38	$37	$38
		Constant real	$26	$27	$29	$27
		Increasing real	$14	$17	$19	$16
	Constant real	Constant nominal	$62	$53	$48	$55
		Constant real	$42	$39	$38	$40
		Increasing real	$23	$23	$25	$23
Dynamic rate	Constant nominal	Constant nominal	$28	$25	$24	$26
		Constant real	$21	$20	$20	$20
		Increasing real	$13	$14	$14	$14
	Constant real	Constant nominal	$46	$36	$31	$38
		Constant real	$34	$28	$26	$30
		Increasing real	$22	$20	$19	$20

[a]A discrete distribution of discount factors based on real discount rates of 3, 5 or 7% with probabilities 25, 50 and 25% respectively

4.4.2 The time-value of money

Thus far, we used discounting when calculating the breakeven carbon prices. We now explore the impact of ignoring the time value of money when analysing the case-study.

Table 4.3 shows that the breakeven initial carbon price is highly sensitive to the omission of discounting. In most cases, the breakeven carbon price is larger when discounting is not performed (on average, it is 68% larger, but depending on the assumptions made about the three dynamic variables, the difference varies markedly ranging from −28% to +156%). The error is greater under a constant nominal carbon price, constant real opportunity cost, and with a dynamic sequestration rate.

We acknowledge that there is one particular set of circumstances in which failing to discount does not introduce any error. That is where all three of the following conditions are met: (1) the carbon price and opportunity cost both remain constant in nominal terms or both remain constant in real terms; (2) constant average rates of sequestration are used; and (3) there is no permanence requirement, such that credit claiming at the average rate occurs for the entire time the sequestering practice must be implemented. (In calculating Table 4.3 we assumed a 100-year permanency requirement.) Many of the surveyed studies that failed to discount also did not specify their assumptions about the carbon price and opportunity cost, so we do not know if they met these conditions. However, we judge that, in many situations, the three conditions are unlikely to be met. Failing to discount would also introduce errors if there are other costs associated with a sequestering process that are dynamically-variable, like upfront costs for tree establishment in the case of reforestation or registration fees to initiate a sequestration project. Therefore, as a rule, discounting should always be performed when analysing the economics of sequestration.

Table 4.3. Breakeven initial carbon price (in Australian Dollars per tonne of carbon dioxide) with and without discounting (at a real rate of 5%) to allow for the time-value of money.

Sequestration rate	Opportunity cost	Carbon price	With discounting	Without discounting	Percentage difference
Constant rate	Constant nominal	Constant nominal	$38	$62	64%
		Constant real	$27	$32	16%
		Increasing real	$17	$12	−28%
	Constant real	Constant nominal	$53	$106	101%
		Constant real	$39	$62	59%
		Increasing real	$23	$27	13%
Dynamic rate	Constant nominal	Constant nominal	$25	$62	143%
		Constant real	$20	$37	86%
		Increasing real	$14	$17	22%
	Constant real	Constant nominal	$36	$92	156%
		Constant real	$28	$62	117%
		Increasing real	$20	$32	65%

4.5 Discussion

We show that assumptions about dynamics can have a strong influence on estimates of the carbon price required for sequestration to be financially attractive. Although our analysis is illustrated by means of a single case-study, the factors that make the financial performance of sequestration sensitive to time-dependent variables—long timeframes, during which rates of sequestration are often non-linear, or over which prices or costs may change—are relevant to many situations globally. Of the four issues investigated (sequestration rate, carbon price, opportunity cost and discounting), no single issue stands out as having a clearly greater impact on the results than the others. Furthermore, there are interactions between the four. For example, in our case study, allowing for real increases in the carbon price reduced the impact of varying assumptions about the sequestration rate. Therefore, all four issues need to be dealt with appropriately and reported clearly. That so few of the published papers do so is concerning.

The role of discounting has caused much debate in the literature on climate change economics. The debate is not whether discounting should be used (Pannell and Schilizzi, 2006) but rather over what discount rates and discounting procedures should be used (Weitzman, 1998; Gollier and Weitzman, 2010). For example, a key factor driving differences between the conclusions of Stern (2007) and other prominent economic studies of policy for climate change mitigation is the discount rate used (Quiggin, 2008). Recently, a team of leading environmental economists, assembled by the US Environmental Protection Agency, set out to establish what should be considered best practice for evaluating benefits and costs in a long-term context (Arrow et al., 2012). They unanimously agreed that discounting is required. The team also agreed that the long-term discount rate is uncertain, and that this causes the certainty-equivalent discount rate to decline in the more-distant future. Such an approach may help mitigate criticisms that discounting can reduce future costs (such as permanency obligations) to a value smaller than decision-makers perceive them to be (Hertzler, 2006). Nonetheless, in our case-study, for a particular set of assumptions about discount-rate uncertainty, we found that the use of a certainty-equivalent discount rate did not greatly alter the breakeven carbon price relative to using the expected value of the discount rate.

The debate about discounting relates to its use by governments when making public decisions on behalf of the community. In this paper, we are concerned with benefits and costs to private landholders considering sequestration. Landholders do apply discounting (at least implicitly) when making financial decisions (Teklewold, 2012), consistent with maximisation of their private net benefits (Robison and Barry, 1996).

Potential inaccuracies introduced by a failure to discount the benefits and costs of sequestration include: (i) overestimating the impact of permanency obligations that require carbon to be maintained after the sequestration period; (ii) underestimating the relative importance of upfront costs; (iii) ignoring the opportunity cost of capital tied-up in sequestration activities; and (iv) overlooking the benefits of rapid initial sequestration when rates are recognised as being dynamic. Due to this interaction between dynamically-decreasing sequestration rates and discounting, in our results the carbon price required to breakeven was always lower with dynamic rates (in some scenarios considerably lower).

The highest discount we considered was 7%. In developing countries—where more than one third of the studies we surveyed were conducted—real discount rates can be over 10% or even 20% (e.g., Sasaki and Yoshimoto, 2010; Teklewold, 2012; Ndjondo et al., 2014). Such high rates reflect greater institutional instability and uncertainty, credit constraints leading to higher opportunity costs of capital, and the potentially higher rates of economic growth in those countries. Under these higher discount rates, the difference between assuming dynamic sequestration rates and constant average rates when evaluating the economics of sequestration will be further amplified.

West and Six (2007) cautioned that when sequestration is quantified as an average rate, the assumed duration of sequestration will have a large effect on the total estimate of mitigation. If activities sequester carbon at similar average rates but over disparate time-spans, the total sequestration from each activity will be different. Our results highlight the need for additional caution because, even with accurate information about the duration of sequestration, assuming a constant average rate can still give misleading results about the economic viability of a sequestering activity, if in reality policy allows credits to be claimed at dynamic rates.

In our case-study, assumptions about the trajectory of carbon prices or opportunity costs changed the breakeven carbon price more than three-fold. However, these assumptions reflect judgements or expectations about the future which are inherently uncertain. It is therefore difficult to conclusively identify what the most appropriate assumptions are. For example, assuming a constant opportunity cost in nominal terms could be defended on the basis that real prices for many agricultural commodities declined through the twentieth century as productivity growth outstripped demand. However, there are a number of factors that may lift prices in the future including growing populations, rising wealth, increasing competition for land and climate change. Options for representing such unknowns in economic analysis include the explicit representation of uncertainty with probability distributions, resulting in a probability distribution for breakeven carbon price, or sensitivity analysis to explore the consequences of alternative assumptions.

Our findings have implications for the development of mitigation policies worldwide. Because many existing studies are unclear or silent about their assumptions regarding dynamic matters, one cannot interpret their results with confidence. As noted earlier, for opportunity cost alone, 91% studies surveyed provided insufficient information for the results to be interpreted. Even where information is provided, the assumptions and/or procedures used are sometimes questionable (e.g. constant sequestration rate, no discounting). Amongst the 32 studies surveyed, only one was clear about their assumptions (including the terms of their assumptions) and used a defensible methodology utilising financial discounting. Clearly, there is a need to improve the analytical approach and reporting in this area. Currently, there is a risk that readers will compare the results of different studies that employ incompatible assumptions (Richards and Stokes, 2004). If studies of the economics of sequestration are used to inform policy without the required transparency, it is likely that decision-makers will be led to over-rate or under-rate the importance of sequestration as a component of the mitigation policy portfolio.

It follows that decision-makers need a sound grasp of the impact of dynamic issues on the economics of mitigation when designing policy. They need this understanding to evaluate the evidence placed before them. It would also help them to recognise a number of policy-relevant implications of the dynamics we have studied here. For instance, permanence obligations may not be as onerous as claimed by those on whom the obligations are imposed, due to discounting. Likewise, the use of dynamic rather than constant sequestration rates in a policy or scheme will, *ceteris paribus*, lead to the wider uptake of sequestration at a lower carbon price. If instead credits are granted based on the average sequestration rate (or if they are granted upfront), the rate at which the abatement occurs over time is not captured, hindering equitable (and market-efficient) trade-off between different mitigation options. In fact, this applies not only to sequestration, but also to mitigation activities that reduce emissions over time, even if reductions occur after an upfront change in practice, technology or management has been made.

4.6 Conclusion

Estimates of the cost of using carbon sequestration to mitigate climate change are sensitive to assumptions about the dynamics of carbon price, the opportunity cost incurred by adopting the sequestering practice, the dynamics of sequestration, and the use or non-use of discounting to compare benefits and costs that occur at different points in time. However, this appears to be poorly acknowledged and scrutinised in many existing analyses of the cost of sequestration, with more than 95% of the studies we surveyed failing to fully disclose their assumptions about dynamic matters.

The study has important implications for global strategies for climate change mitigation and adaptation. We face difficult and costly decisions when striking the balance between mitigation and adaptation; and in the context of mitigation we must strike the balance between emission abatement and sequestration. For both of these purposes, it is crucial that managers and policymakers identify and undertake the least-costly strategies that achieve global objectives for limiting climate change.

Identifying the least-costly strategies is not possible without sound evidence about the economic performance of each of the options. We have shown that, despite the existence of a large body of literature on the economics of sequestration, the required evidence is almost completely absent. With only a single exception, all of the studies we reviewed employed unrealistic assumptions about the dynamics of sequestration, and/or were silent about crucial assumptions about the dynamics of prices and costs, and/or employed a methodology that is inconsistent with economic best practice. Ideally, none of these deficient studies should be used by decision-makers or policymakers; but if they are used, caution should be applied.

Of the weaknesses we have identified, one of them has clear consequences for decision-making. In studies that correctly employ discounting, the failure to represent the dynamic changes in the rate of sequestration biases the economics of carbon sequestration away from sequestration. If they have relied on studies with this bias, it may be that decision makers need to tilt the balance between abatement and sequestrations somewhat more towards sequestration, and they should potentially tilt the balance between mitigation and adaptation somewhat more towards mitigation. This latter point is relevant to specific circumstances where adaptation previously appeared more favourable than sequestration, but no longer does once the bias against sequestration is removed.

On the other hand, we cannot be certain about whether and how studies are biased without full information on, and justification for, the assumed dynamics of carbon price and opportunity cost and the approach to discounting that has been used. It may be that there is a need to commission new studies of the economics of sequestration, with requirements for full transparency, good justification of assumptions, and sound economic methodology.

4.7 Appendix

Table 4.4 shows the details of the 32 studies we surveyed.

Table 4.4. Details of, and assumptions used by, the 32 studies we surveyed that assess the financial viability of a sequestration activity.

			Dynamic assumptions			
Reference	**Carbon sequestered in:**	**Study location**	**Opportunity cost**	**Price of carbon (credits)**	**Sequestration rate**	**Considered time-value of money?**
Antle et al.(2001)	Soil	USA	Constant (terms not specified)	Constant (terms not specified)	Constant	Yes
Antle et al. (2003)	Soil	USA	Constant (terms not specified)	Constant (terms not specified)	Constant	Yes
De Jong et al. (2000)	Vegetation (primarily)[A]	Mexico	Assumption not specified[B]	Assumption not specified[B]	Constant	Yes
Diagana et al. (2007)	Soil	Senegal	Constant (terms not specified)	Constant (terms not specified)	Constant	Yes
Doraiswamy et al. (2007)	Soil	Mali	Assumption not specified[B]	Assumption not specified[B]	Constant	No
Flugge and Abadi (2006)	Vegetation	Australia	Assumption not specified[B]	Assumption not specified[B]	Constant	Yes
Grace et al. (2010)	Soil	Australia	Constant (terms not specified)	Constant (terms not specified)	Constant	Yes
Harper et al. (2007)	Vegetation (primarily)[A]	Australia	Assumption not specified[B,C]	Assumption not specified[B]	Dynamic (& upfront payment[D])	Yes
Henry et al. (2009)	Vegetation	Kenya	Decreased (terms not specified)[E]	Assumption not specified[B]	Constant	No
Karky and Skutsch (2010)	Vegetation	Nepal	Constant (terms not specified)	Constant (terms not specified)	Dynamic	No[F]
Kragt et al. (2012)	Soil	Australia	Constant (terms not specified)	Constant (terms not specified)	Constant	No
Kingwell (2009)	Vegetation	Australia	Constant (terms not specified)	Increasing at fixed exponential rate in real terms	Dynamic	Yes
Lam et al. (2013)	Soil	Australia	Assumption not specified[B,G]	Assumption not specified[B]	Dynamic	No

Reference	Carbon sequestered in:	Study location	Dynamic assumptions			Considered time-value of money?
			Opportunity cost	Price of carbon (credits)	Sequestration rate	
Lewandrowski et al. (2004)	Soil and vegetation	USA	Constant (terms not specified)	Constant (terms not specified)	Constant	Yes
Luedeling et al. (2011)	Vegetation (primarily)[A]	Africa	Assumption not specified[B]	Assumption not specified[B]	Constant	No
McKenney et al. (2006)	Vegetation (primarily)[A]	Canada	Constant (terms not specified)	Constant (terms not specified)	Dynamic	Yes
Moulton and Richards (1990)	Vegetation (primarily)[A]	USA	Assumption not specified[B,C]	Constant[H]	Constant	Yes
Ndjondo et al. (2014)	Vegetation	Gabon	Assumption not specified[B]	Assumption not specified[B]	Dynamic	Yes
Nielsen et al. (2014)	Vegetation (primarily)[A]	USA	Constant (terms not specified)	Assumption not specified[B]	Dynamic	Yes
Newell and Stavins (2000)	Vegetation (primarily)[A]	USA	Constant (terms not specified)[I]	Constant[H]	Dynamic	Yes
Pautsch et al. (2001)	Soil	USA	Assumption not specified[B]	Assumption not specified[B]	Constant	No
Petersen et al. (2003a)	Vegetation	Australia	Constant in real terms	Constant in real terms	Constant	No [J]
Popp et al. (2011)	Soil	USA	Constant (terms not specified)	Constant (terms not specified)	Constant	No
Rootzén et al. (2010)	Vegetation (primarily)[A]	India	Assumption not specified[B]	Assumption not specified[B]	Not specified	Yes?[K]
Sasaki and Yoshimoto (2010)	Vegetation	Cambodia	Assumption not specified[B]	Assumption not specified[B]	Not specified	Yes
Seidl et al. (2007)	Vegetation (primarily)[A]	Austria	Both constant (terms not specified) and varied	Constant (terms not specified)[H]	Both constant & dynamic	Yes[L]
Takimoto et al. (2008)	Vegetation	West Africa	Constant (terms not specified)[I]	Assumption not specified[B]	Constant	Yes
Thamo et al. (2013)	Vegetation	Australia	Constant in real terms	Increase initially then constant in real terms	Dynamic	Yes
Torres et al. (2010)	Vegetation (primarily)[A]	Mexico	Constant (terms not specified)	Assumption not specified[B]	Both constant & dynamic[M]	Yes
Tschakert (2004)	Soil	Senegal	Assumption not specified[B]	Assumption not specified[B]	Constant	No[J]

Reference	Carbon sequestered in:	Study location	Dynamic assumptions: Opportunity cost	Dynamic assumptions: Price of carbon (credits)	Dynamic assumptions: Sequestration rate	Considered time-value of money?
van 't Veld and Plantinga (2005)	Vegetation (primarily)[A]	USA	Constant (terms not specified)	Increasing at fixed exponential rate (terms not specified)	Dynamic	Yes
Wilman (2011)	Soil	N. America	Constant (terms not specified)	Constant (terms not specified)	Dynamic	Yes

[A]Included carbon accumulated in soil as a result of afforestation/reforestation with woody vegetation in their calculations of total sequestration.

[B]We suspect it may have been assumed to remain constant, most likely in real terms. However we cannot be certain because the assumption was not clearly stated.

[C]The cost of leasing/renting land for sequestration interpreted as a *de facto* opportunity cost.

[D]Considered two scenarios: one with sequestration credits generated dynamically at five year intervals and another with upfront payment based on amount of sequestration predicted to occur over the entire duration of the project.

[E]Considered transaction costs rather than opportunity costs.

[F]Time-value of money ignored on the basis that the flows of benefits were relatively uniform and that only a five year timeframe was considered.

[G]Considered the opportunity cost of nitrogen required to stabilise soil carbon rather than the opportunity cost of land-use or practice change.

[H]Treated as cost per tCO_2 rather than price per tCO_2.

[I]Considered the possibility of different opportunity costs (e.g., by changing agricultural commodity prices or crop yield) but not changes in the opportunity cost *through* time.

[J]Discounted future benefits and costs but, rather than annualising, the resultant present values were simply divided by time.

[K]Stated that a 6% discount rate was used but not possible to check if the time-value of money was properly accounted for due to the lack of details provided.

[L]Except where the time-value of money was deliberately ignored for comparison's sake.

[M]Compared scenarios with annual payments based on a constant rate of sequestration against a scenario where payments for sequestration were instead made at a non-constant rate, with a bias toward the early years of the project.

Chapter 5. Paper 4. Measurement of greenhouse gas emissions from agriculture: economic implications for policy and agricultural producers

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Measurement of greenhouse gas emissions from agriculture: economic implications for policy and agricultural producers

5.1 Abstract

If agriculture were to be included in Australia's carbon price scheme, a key decision for government would be how to estimate greenhouse gas emissions. We explore the consequences of three different methods for measuring on-farm emissions: national accounting methods, an amended version of those methods, and use of best-available local data. Estimated emissions under the three methods can vary widely; for example, on a case study farm in Western Australia, local data indicated 44 per cent lower emissions than did the national accounts method. If on-farm emissions are subject to an emissions price, the impact on farm profit is large, and varies considerably with different measurement methods. For instance, if a price of \$23/t of $CO_{2\text{-}e}$ applies then farm profit falls by 14.4 to 30.8 per cent depending on the measurement method. Thus, the choice of measurement method can have large distributional consequences. On the other hand, inaccurate measurement results in relatively minor deadweight losses. On-farm sequestration through reforestation may lessen the impact of an emissions price on farm businesses, although it will require a high carbon price to be viable, especially if sequestration rates are underestimated or low.

Keywords: economic modelling, emissions measurement, greenhouse gas accounting methodology, nitrous oxide (N_2O), sequestration

5.2 Introduction

The Australian government, like many governments, is adopting policies and initiatives to reduce emissions of greenhouse gases (GHGs). An emissions trading scheme (ETS) comes into effect on 1 July 2012, initially with a fixed price of \$AUD23 per tonne of carbon dioxide equivalent (CO_2-e). Further, over \$1.7 billion is being invested in Australia's land sector from 2011 to 2016 to reduce and offset GHG emissions.

Agriculture accounts for 16 per cent of Australia's GHG emissions, yet is excluded from the ETS, at least in its initial stages, and so Australian agriculture will mainly be affected indirectly by the establishment of a price on emissions. Other sectors covered by the ETS, such as electricity generators and processors, will pass on to farmers their higher costs and/or use farm land as a source of emission offsets via carbon sequestration.

Although initially excluded from the scheme, agricultural emissions nonetheless are measured or estimated and reported in the national inventory of emissions using methods outlined in the National Inventory (2011). There is the prospect that agriculture may be included in the scheme at a future date. Under either scenario—agriculture excluded or included in an ETS—accurate measurement of agricultural emissions is important.

The National Inventory methods predict emissions using parameters based upon peer-reviewed science. For countries like Australia, however, this can be problematic as most studies of agricultural emissions that are the sources of these standard parameters consider northern hemisphere agriculture (Galbally et al., 2005; Stehfest and Bouwman, 2006; Barton et al., 2008), yet Australian soils, climate and agricultural operations can be very different. To mitigate this, standard parameters are sometimes updated with country-specific values. However there are often substantial regional differences in rates of emissions, attributable to differences in climate, soils and agricultural practices (Berdanier and Conant, 2012), especially in a large country like Australia. National accounting methods typically lack the detail and spatial resolution to accommodate all these differences (Williams et al., 2012).

It is therefore perhaps not surprising that one reason stated for excluding agricultural emissions from the ETS in its early years is that they are hard to quantify. Knowledge of their spatial and temporal variation is often poor (Leip et al., 2011; Misselbrook et al., 2011; Berdanier and Conant, 2012) and this impedes formulation of efficient policies to lessen agricultural emissions (e.g., Rypdal and Winiwarter, 2001).

Thus the accuracy of methods for estimating agricultural emissions is important for policy. On the one hand agriculture is a significant source of emissions (Garnaut, 2008), yet knowledge about emissions on actual farms in different environments is often inadequate. Addressing these knowledge gaps would involve transaction costs, so one possible response by policymakers is to apply a uniform national formula-based approach to estimation of emissions. Alternatively, programs could use more accurate (but more expensive) approaches that account for variations over time, space and farming practices. In this article we investigate three different measurement methods, including the national accounting method. We outline the farm business and emission consequences of applying these different emission measurement methods when carbon prices and different emission policy scenarios apply to agriculture.

Our analyses use the central grainbelt of Western Australia as a study area. This area is known to have agro-climatic conditions (semi-arid) that typically are not well represented by the emissions factors in the national inventory accounting system (Galbally et al., 2005; Barton et al., 2011). However, local scientific data on emissions (particularly of N_2O) exist for the study region (Figure 5.1) (e.g., Barton et al., 2008; Barton et al., 2010; Barton et al., 2011; Li et al., 2011).

The article is structured as follows. The next section includes outlines of the farm modelling approach, the methods for estimation of emissions, the representation of carbon pricing and the associated emissions policy scenarios investigated. We then present and discuss our results before drawing conclusions.

5.3 Methods

5.3.1 Farm modelling

MIDAS is a detailed steady-state optimisation farm model that accounts for biological, managerial, financial and technical aspects of dryland farming. Originally developed in the mid-1980s (Kingwell and Pannell, 1987), later versions of MIDAS and/or examples of its applications relevant to GHGs are described by Kingwell et al. (1995), Petersen et al. (2003b), Kopke et al. (2008), Kingwell (2009), Doole et al. (2009) and Kragt et al. (2012).

The model's objective is to maximise farm profit after deduction of all operating costs, overhead costs, depreciation and opportunity costs associated with farm assets (exclusive of land) from production receipts. The several hundred activities in MIDAS include alternative rotations on each of eight soil classes (S1 –S8), crop sowing opportunities, feed supply and feed utilisation by different livestock classes, yield penalties for delays to sowing, cash flow recording, and machinery and overhead expenditures. The model's solution is the set of activities that draws on farm resources to generate maximum profit subject to a range of constraints. Constraint types include resource constraints (e.g., on several different qualities of land, on machinery capacity), technical constraints (e.g., representing the demand for, and supply of, animal feed), logical constraints (e.g., determining the number of three-year-old sheep depending on the number of two-year-old sheep the previous year and the number of sales and purchases of sheep of relevant ages) and financial accounting constraints.

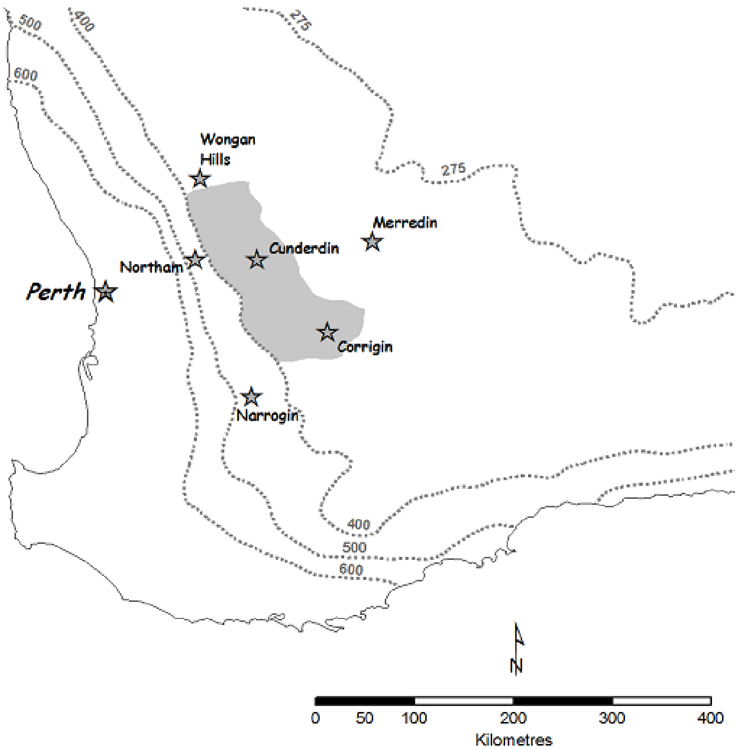

Figure 5.1. The central grainbelt represented by this MIDAS model with rainfall isohyets in mm. Source: Gibson et al. (2008).

Although versions of MIDAS exist for various regions (e.g., Flugge and Schilizzi, 2005), the model used in this paper represents a typical 2000 ha farm in the central Wheatbelt of Western Australia. The area is characterised by a Mediterranean climate with long, hot and dry summers and cool, moist winters (June-August) and a growing season (May to October) during which about 75 per cent of the 350-400 mm annual rainfall occurs. A typical farm engages in a mix of cropping and livestock enterprises across the soil types listed in Table 5.1. The crops grown include wheat, barley, oats, lupins, canola, field peas, chickpea and faba beans. These are sown in rotation with legume pastures including annual subterranean clover and serradella and perennial lucerne. Sheep, mostly Merino breeds, graze these pastures producing wool and meat.

Table 5.1. The eight soil types in the MIDAS model.

	Name	Main soil types	Area (ha)
S1	Poor sands	Deep pale sand	140
S2	Average sandplain	Deep yellow sand	210
S3	Good sandplain	Yellow loamy sand	350
S4	Shallow duplex soil	Sandy loam over clay	210
S5	Medium heavy	Rocky red/brown loamy sand/sandy loam; Brownish grey granitic loamy sand	200
S6	Heavy valley floors	Red/brown sandy loam over clay; Red/grey clay	200
S7	Sandy-surfaced valley	Deep/shallow sandy-surfaced valley floor	300
S8	Deep duplex soils	Loamy sand over clay at depth	390

5.3.2 Inclusion of agricultural emissions and a carbon price in MIDAS

MIDAS was updated with cost and price structures that were the average of real prices from 2007 to 2011. It was also modified to include: a carbon price; options to sequester carbon in trees; and formulas for estimating GHG emissions from farm activities, based on those in national GHG accounts.

5.3.2.1 Representing a carbon price in MIDAS

Input prices in MIDAS were adjusted upwards to account for the impacts of the ETS on input suppliers. Although initially all fuel use economy-wide will be excluded from a carbon price, the government intends to apply the carbon price to heavy transport vehicles from 2014 (Australian Government, 2011). Hence, in the medium term, goods and services dependent on energy and transport will become more expensive. In this analysis, fuel used by heavy haulage vehicles was assumed to be subject to a carbon price.

To model cost increases attributable to a carbon price, this study adopted the approaches of Keogh and Thompson (2008) and Kingwell (2009) who related increases in the transport/haulage fuel price attributable to a carbon price to goods and services used by farm businesses. To illustrate, combustion of one litre of diesel produces 2.7 kg of CO_2-e (NGA Factors, 2010). For each \$10 increment in the carbon price, the price of transport fuel would rise by 2.7 cent/L. Following Keogh and Thompson (2008) and Kingwell (2009), simple flow-on cost factors based mostly on fuel costs (see Table 5.2) applied to a range of farm inputs and services. As an example, if transport fuel prices increased by 5 percent, then chemical costs would be expected to increase by 1.25 percent (that is, 25 percent of 5 percent). However, some revisions to the cost-flow through factors of Keogh and Thompson (2008) and Kingwell (2009) occurred as their analyses were based on a previous policy proposal in which the carbon price would have applied to all fuel use, economy-wide.

Table 5.2. Factors for the flow-on of a carbon price for various farm inputs (based on Kingwell, 2009).

Farm input	Flow-on cost factor	Farm input	Flow-on cost factor
Contract seeding	0.15	Shire rates	0.10
Contract harvesting	0.15	Repairs & maintenance	0.15
Shearing	0.15	Grain handling	0.30
Sheep work	0.15	Transport	0.25
Fertiliser	0.25	Hired labour	0.15
Chemicals	0.25	Professional fees	0.05
Electricity	1.00	Fuel (transport)	1.00
Livestock processing	0.15	Fuel (on-farm)	0.05

Inputs like fertilisers and chemicals may not become much more expensive under the current carbon pricing policy. Although their manufacture is energy-intensive, only domestic producers of these inputs will face higher costs. Australian manufacturers of farm inputs that compete with imported substitutes not subject to a carbon price will have a limited ability to pass on the domestic carbon price to their customers. Therefore the impact of a carbon price on these farm inputs is likely to be minimal (Tulloh et al., 2009). Finally, because the major products from farms in the study area are unprocessed exports (e.g., wheat, wool, live sheep), we assumed that commodity sale prices would be unchanged by the establishment of a carbon price in Australia.

5.3.2.2 *Methods for estimating agricultural emissions*

The following sources of on-farm emissions were accounted for in the model using units of CO_2-e: CH_4 from enteric fermentation; N_2O from animal waste, N fertiliser, biological N-fixation and crop residues; and CO_2 from urea hydrolysis. With a carbon price the cost of fuel used on-farm was assumed to only increase by 5 per cent (Table 2) due to extra handling costs before it reaches the farm. If agriculture was included in the carbon pricing mechanism, then we assumed that emissions from fuel used on-farm for activities like crop establishment and harvest would count as agricultural emissions and so accordingly these emissions were included with those sources listed above.

The amount of on-farm emissions produced from these sources was estimated using three different GHG accounting methods:

i) Standard. The standard method used by the Australian Government in their national GHG accounting, as outlined in the National Inventory (2011).

ii) Amended. The National Inventory (2011) uses a process-based approach to estimate emissions, but in our judgement the approaches used for some sources of emissions are inconsistent with actual processes. For example, determining N_2O emissions from N-fixation requires quantification of how much N has been fixed. The National Inventory (2011) quantifies N fixation based upon just the N content of legume stubble, and fails to account for N removed in grain. Furthermore, the inventory accounts for N_2O emissions from N fixation and residues for legume crops, but for legume pastures, only N fixation is considered, ignoring that these pastures have N-rich residues. In the amended accounting method, these inconsistencies are corrected.

iii) Local. Where local scientific data exists, the Amended method was adapted and modified based on the best available results of local field trials conducted in the Wheatbelt region.

Exact detail of the assumptions and formulas used for each method is contained in the on-line appendix.

MIDAS was modified by inserting transfer rows for each of these aforementioned sources of agricultural emissions into the matrix. For every activity (column) in MIDAS that may cause any of these emissions a positive coefficient was inserted into the transfer row for that emission. This coefficient was set to the value (i.e., amount of emissions) estimated for that activity by the formulas in the appendix. Consequently, this value often changed depending on which GHG accounting methodology was used. For instance, a hectare of pasture-pasture-wheat rotation on soil type S3 would produce 16, 105 or 9 kg of CO_2-e yr^{-1} of N_2O emissions from the decomposition of crop residues when Standard, Amended or Local methods were used respectively. When a carbon price was placed directly on agricultural emissions the transfer rows were constrained to zero and the model forced to satisfy this constraint by undertaking sequestration or paying the carbon price—activities which both had negative coefficients in the matrix.

5.3.2.3 Sequestration

The option of being able to revegetate land to sequester carbon was also investigated. As with emissions, the amount of sequestration could be estimated using different methodologies. One option would be for governments to rely on the national GHG accounting methodologies such as the Australian Greenhouse Office's National Carbon Accounting Toolbox (NCAT) *FullCAM* model. NCAT was developed by combining process and empirical modelling at the continental scale (Jonson, 2010) and, like the Standard method for estimating emissions, was not originally intended for use at the farm-level. Alternatively, sequestration could be estimated from locally collected data. To represent this option, we used a non-symmetrical sigmoidal growth pattern, developed from data on tree growth in the study area (Jonson, 2010). Although NCAT's predictions of sequestration are much lower than locally measured data for the study area, both exhibit a broadly similar trend whereby the rate of carbon accumulation decreases over time, eventually plateauing after around 50 years. To ensure conservatism and to provide a 'buffer against the risk of reversal', estimates of sequestration were reduced by five per cent (DCCEE, 2010b).

Estimating the revenue from sequestration required translating the future returns from carbon sequestration into a form compatible with MIDAS, which represents a single year of production, assumed to be in a cyclical steady state (costs in MIDAS were assumed to stay constant in real terms). To do this, a stream of sequestration payments in future years was estimated using the aforementioned NCAT or local data—depending on the scenario under investigation—and an assumed carbon price (see below). This stream of payments was then discounted (using a rate of 7 per cent p.a.) and converted into an annuity to give the equivalent annual revenue expected from sequestration. The annuity was included in the MIDAS model as the annual sequestration income from planted trees. A similar technique was employed by Jonson (2010) and Kingwell (2009), except that in the current analysis we assume that sequestration is claimed for 50 years (when tree growth 'plateaus'), and that the carbon in the trees then has to be maintained for a further 100 years past the cessation of sequestration, in accordance with permanency requirements of Australia's relevant policy, the 'Carbon Farming Initiative' (DCCEE, 2010b).

The carbon price used in each scenario represents an initial starting price which is assumed to increase at 2.5 per cent[1] p.a. in real terms for the first three years. For the purpose of this analysis, it is assumed that national and/or international politics result in

[1] This rate of increase is used in Australia's recently legislated carbon tax, with a $23/t of CO_2-e initial price

a lack of political will to further increase the price (in real terms) after three years. If we were to assume further price increases, then the differences in results between scenarios would be increased, increasing the importance of accurate measurement of emissions.

5.3.3 Policy scenarios

Three policy scenarios were considered:

1. ‘Business-as-usual’. There is no price on emissions. Emissions have no impact on profit-maximising farm management decisions.
2. A carbon price is imposed domestically but on-farm emissions are excluded, as per current legislation. Under this scenario farmers can undertake (Kyoto-compliant) revegetation for sequestration.
3. A carbon price is imposed domestically, including on-farm emissions. As a ‘trade-exposed’ industry, agriculture is granted ‘free permits’ to partially shield it from adverse consequences of carbon pricing. If there are ‘excess’ free permits, scenarios are examined when their on-selling is either allowed or prohibited.

For the last two scenarios, we explore the consequences of using an inaccurate accounting method for farmers and then examine the implications for policy efficiency.

5.4 Results and discussion

5.4.1 Business-as-usual: greenhouse gas emissions and farm profit

All the results in this sub-section relate to the scenario where there is no price on emissions. In this case the optimal farming system has 73 per cent of the arable land allocated to crop and generates an annual profit of $96,800. This is consistent with survey results showing that farmers in the study area tend to crop about 70 per cent their arable land (Planfarm, 2010). Around this optimal strategy, a region of high profit (within 12.5 per cent of the maximum) occurs where approximately 55 to 85 per cent of the farm is cropped (Figure 5.2). Reasons for the occurrence of relatively flat pay-off regions like this are outlined by Pannell (2006).

If on-farm emissions are estimated with the Standard method, then under steady-state optimal management, the 2000 ha farm emits 1062 t of CO_2-e yr^{-1}. Of this, more than half (554 t of CO_2-e yr^{-1}) is associated with livestock—mainly CH_4 from enteric fermentation, but also N_2O from animal waste. Other sources include 263 t of CO_2-e yr^{-1} from N_2O as a result of N fixation, 124 t of CO_2-e yr^{-1} from N_2O released during the decomposition of residues, 79 t of CO_2-e yr^{-1} from fertiliser use (N_2O and CO_2 from urea hydrolysis) and 41 t of CO_2-e yr^{-1} from fuel used on-farm.

If the farming system is constrained to operate at different levels of cropping intensity (Figure 5.2a), emissions from livestock decrease as the area of cropping increases. Because pasture swards typically contain appreciable proportions of legumes, and because the Standard method fails to account for the N fixed by crops that is removed in pulse grain (see Section 5.3.2.2), estimated emissions from N fixation tend to increase when less area is used for cropping. Emissions from the decomposition of residues, fuel and fertiliser use increase with the area sown to crop, but they are relatively minor sources of GHGs. Hence as the amount of land allocated to cropping increases the overall quantity of agricultural emissions falls considerably.

Using the Amended accounting method the on-farm emissions for the optimal farming system (given no carbon price) are 1267 t of CO_2-e yr^{-1} (Figure 5.2b) (up from 1062 t for the Standard method). One of the Standard method's inconsistencies is its failure to account for the N-rich residues of legume pastures. Addressing this irregularity leads to emissions from residues increasing rather than decreasing as the amount of crop in the farming system is reduced. Yet at the same time when the N fixed by pulse crops that is removed in the harvested seed is also taken into account in the Amended method, emissions from N fixation at higher proportions of crop are larger than estimates based on the Standard method. Hence overall, on-farm emissions estimated with the Amended method are higher compared with the Standard method, especially for livestock-dominant farms.

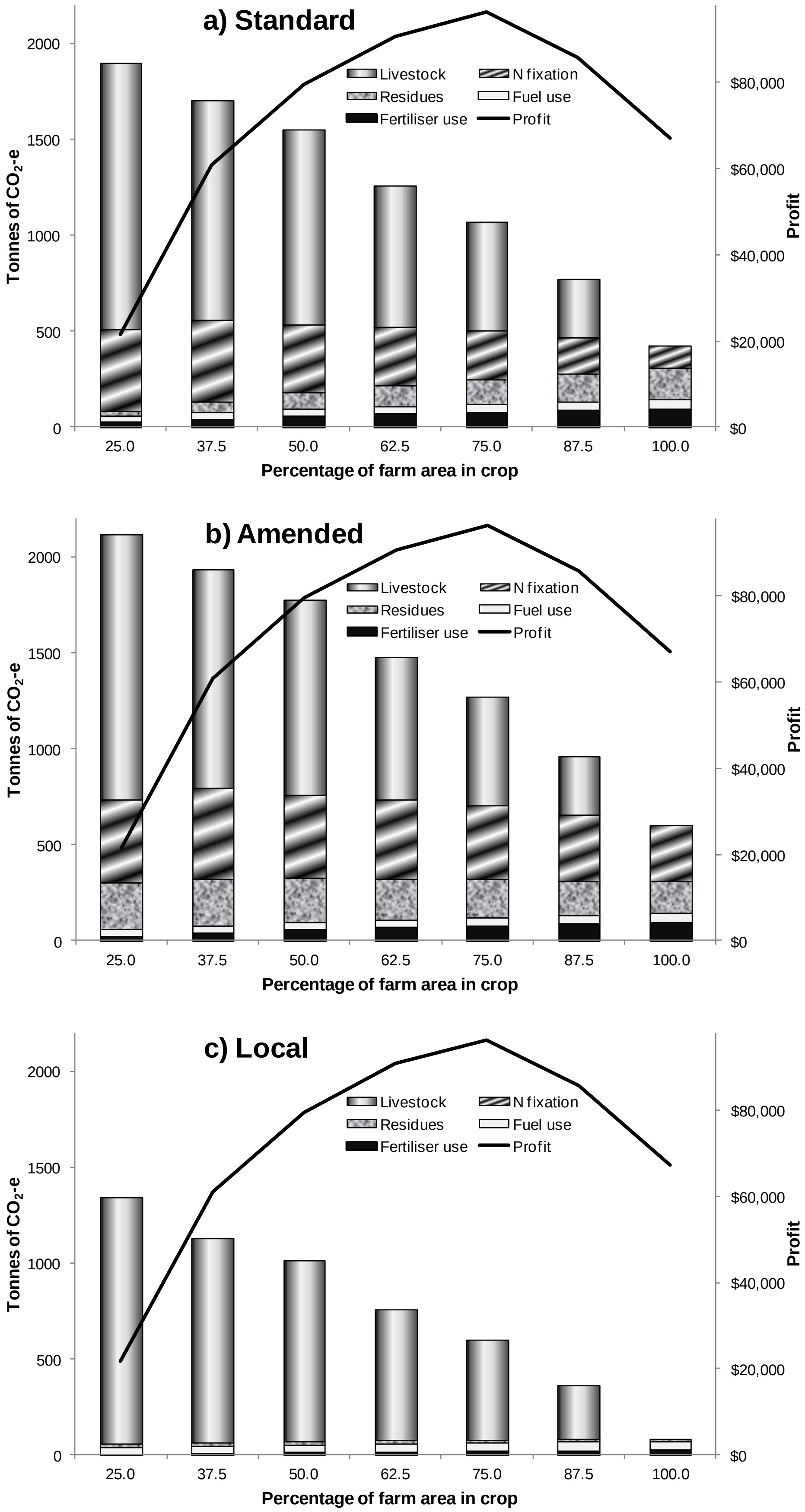

Figure 5.2. Profit and annual on-farm emissions as function of the proportion of the farm allocated to cropping in a 'business-as-usual' scenario, as estimated using **a**) Standard **b**) Amended or **c**) Local methods**.**

Alternatively, if the Local method is used, on-farm emissions are estimated at only 592 t of CO_2-e yr^{-1} for the optimal farming system (Figure 5.2c). That is 56 per cent of that estimated with the Standard method. N_2O emissions from fertiliser, residues and N-fixation are much smaller when estimated with the Local method. Such differences between methods reflect the localised characteristics of N_2O emissions (e.g., Galbally et al., 2005), a finding consistent with N_2O from agricultural soils being the most uncertain source of emissions in national inventories (Rypdal and Winiwarter, 2001). Recorded N_2O emissions in the Wheatbelt region are minimal compared to other semi-arid regions, perhaps because rainfall, soil organic matter levels, N inputs and the use of tillage that incorporates stubble all tend to be relatively low in this area (Li et al., 2011). This makes the dominance of livestock in the farm's emissions profile even greater.

In summary, compared to the Standard accounting method, the Amended method indicates that emissions are higher, due to capturing higher emissions related to N fixation and pasture residues, while under the Local method, emissions are substantially lower, mainly due to much lower emissions from cropping. The results for the Local method are specific to this region, but they highlight that reliance on standard national values will result in errors in some regions, potentially disadvantaging some farmers and advantaging others.

5.4.2 Carbon price imposed but agriculture excluded

The results in this sub-section relate to the policy scenario where there is a price on carbon, but agricultural producers are not required to pay for on-farm emissions. Because of this, the different emissions accounting methods outlined earlier do not influence farm management (or profit) in this scenario. Within this scenario, two possibilities are considered: claiming of offsets for carbon sequestration on farms may be disallowed or allowed. The later possibility represents the situation recently legislated in Australia. The sub-section is included to provide a base line for comparison with later results.

5.4.2.1 *No sequestration can be claimed*

Imposing a domestic carbon price that excludes on-farm emissions has little impact on the proportion of the farm allocated to cropping: the range of cropping percentages with high farm profits continues to be 55 to 85 per cent (Figure 5.3)[2]. Compared to the business-as-usual scenario, the profit of the optimal farming system falls by $5,700 (or 5.9 per cent) to $91,100 at a carbon price of $23/t of CO_2-e, or by $12,400 (12.8 per cent) to $84,400 at $50/t of CO_2-e. Farm profitability falls as price-taking, export-orientated farms in the study region cannot pass on the higher input costs caused by the impost of the domestic carbon price on other sectors of the economy. The higher costs are not because of charges for agricultural emissions, which are excluded in this policy scenario.

5.4.2.2 *With voluntary claiming of on-farm sequestration*

Allowing farmers to sell offsets for carbon sequestered by the voluntary revegetation of their land may reduce the impact of a carbon price on farm businesses (Flugge and Schilizzi, 2005). A high carbon price favours sequestration as it both reduces the viability of other land uses that the revegetation would displace, and also increases the price for which the stored carbon could be sold. If sequestration rates were estimated based on tree growth measured locally in the study area (Jonson, 2010), then an initial price of at least $34/t of CO_2-e is required before it is optimal to revegetate some of the farm's soil types that have a low opportunity cost (results not shown). With an initial price of $50/t of CO_2-e, farm profitability would fall by $4,500 (4.6 per cent) to $92,300 (as opposed to 12.8 per cent in the absence of sequestration) (Figure 5.3). In this scenario the impact of higher carbon prices on farm profit is less than predicted by other studies (e.g., Keogh and Thompson, 2008). As well as allowing for sequestration, other likely reasons for this difference include that this study allows for changes in farm management in response to the carbon price and for the existence of different quality soil types with differing profitability for each enterprise.

If instead of using local data, carbon sequestration is estimated using the NCAT model, income from sequestration is reduced six-fold. This means an initial price in excess of $220/t of CO_2-e is now required before sequestration appears in the optimal solution (results not shown).

[2] This differs from Kingwell (2009) who found that the viability of livestock would increase slightly relative to cropping which tended to be more input-intensive. We attribute this to the carbon price not applying to fuel used on-farm in the current study (in accordance with more recent legislation), and also the factoring in of increases in the cost of processing livestock domestically.

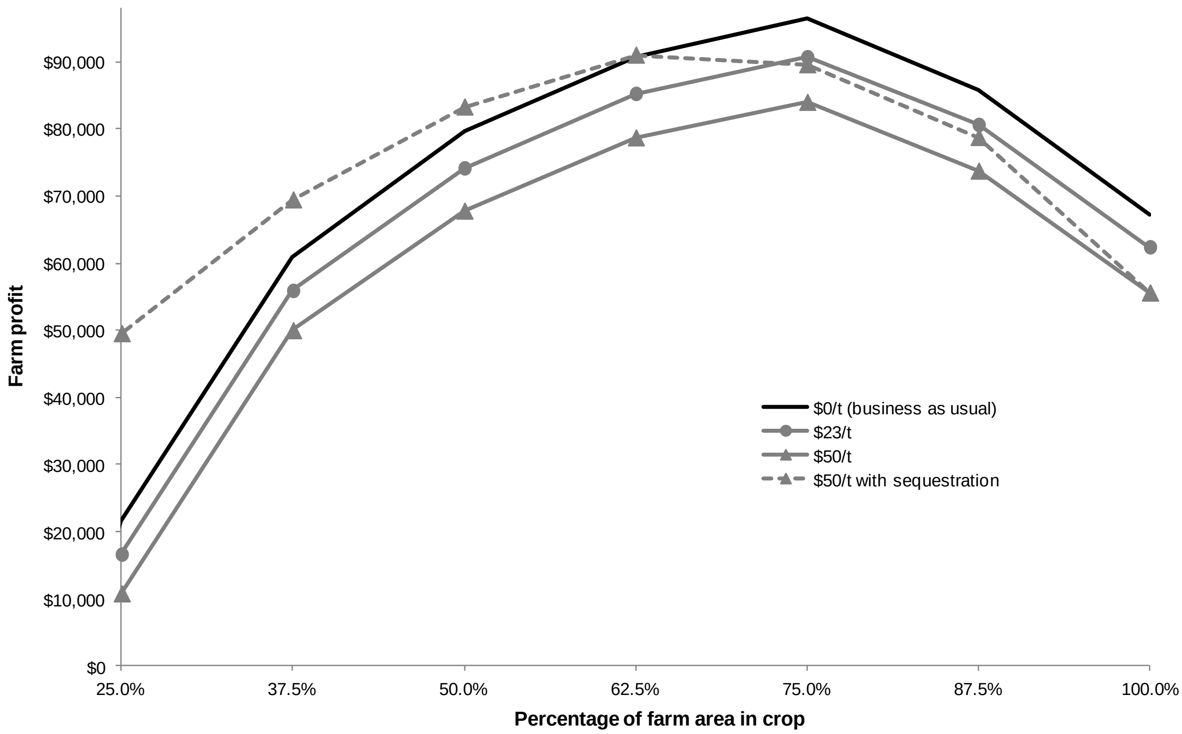

Figure 5.3. Farm profitability when agricultural emissions are not covered by an initial carbon price of either $23, or $50/t of CO_2-e with or without sequestration estimated using local data from (Jonson, 2010).

In summary, the impact of a carbon price that does not include agriculture depends on the carbon price and on whether farmers receive payments for sequestration offsets. Payments for sequestration can offset some or all of the losses due to higher costs resulting from the carbon price, but only at high carbon prices.

5.4.3 Carbon price imposed with agriculture included

This sub-section relates to the policy scenario where there is a price on carbon, and agricultural producers are required to pay directly for their emissions, as well as being affected by higher input costs. Under this scenario farmers can respond to the price for on-farm emissions through a combination of altering farm operations to reduce emissions, using sequestration to abate emissions or paying the carbon price.

5.4.3.1 *No free permits*

With the inclusion of on-farm emissions a carbon price has a substantial impact on farm profits. For instance, applying the Standard emissions accounting method and using an initial price of $23/t of CO_2-e, the profit of the optimal farming system falls to $67,600 (Table 5.3), a $23,500 (25.7 per cent) reduction compared to the scenario where agricultural emissions are excluded from the carbon price (Section 5.4.2). In Section

5.4.1, estimated emissions were the greatest with the Amended method and thus a carbon price on agricultural emissions has the greatest impact with that method (Table 5.3). With the Local method, estimates of on-farm emissions are smaller and so profit of the optimal farming system at $23/t of CO_2-e is $78,000, a reduction of $13,100 (14.4 per cent) compared to when agriculture is excluded. Clearly in this case the method used for emissions measurement at the farm-level has a substantial impact on farm profit.

For mixed farming systems the impact of a carbon price on agricultural emissions would be worse in situations conducive to livestock production (Flugge and Schilizzi, 2005), such as when livestock prices are high relative to grain prices. This is due to the large emissions of CH_4 attributable to livestock (Figure 5.2). It therefore follows that as the carbon price increases, the optimal farming system shifts further toward cropping in order to reduce on-farm emissions.

Table 5.3. Characteristics of the optimum farming-system when agricultural emissions estimated with different methods are included in the carbon price. Percentages in parentheses show the change relative to agriculture's exclusion.

Emissions method	On-farm emissions (t of CO_2-e yr^{-1})	Crop area (ha)	Sheep (DSE)	Revegetated area (ha)	Farm profit ($'000)
Initial carbon price of $23/t of $CO_{2\text{-}e}$ (Sequestration unviable)					
Standard	955 (-8.5%)	1503	2662	0	67.6 (-25.7%)
Amended	1153 (-7.7%)	1504	2656	0	63.1 (-30.8%)
Local	564 (-2.1%)	1472	2934	0	78.0 (-14.4%)
Initial carbon price of $50/t of $CO_{2\text{-}e}$ (Sequestration estimated with NCAT)					
Standard	516 (-50.5%)	1820	507	0	40.2 (-52.4%)
Amended	694 (-44.4%)	1820	507	0	31.2 (-63.1%)
Local	280 (-51.4%)	1720	1199	0	59.0 (-30.1%)
Initial carbon price of $50/t of $CO_{2\text{-}e}$ (Sequestration rate from Jonson (2010))					
Standard	407 (-59.3%)	1585	253	325	55.1 (-40.3%)
Amended	531 (-55.3%)	1600	139	325	47.6 (-48.4%)
Local	255 (-54.3%)	1462	1114	325	70.4 (-23.7%)

Estimates of sequestration are much smaller when NCAT is used compared to locally accurate data, meaning a much higher carbon price is required for afforestation to become viable (Section 5.4.2.2). Hence at $50/t of CO_2-e, no land would be revegetated with NCAT but 325 ha would be afforested if sequestration occurred at the rate reported by Jonson (2010). Hence again the measurement method used is very important in influencing land use, farm profitability and the levels of emissions and sequestration.

In summary, farmers' profits depend on the method used to measure emissions and sequestration and, without free permits, are highly sensitive to the inclusion of agriculture in the carbon price.

5.4.3.2 Agriculture is given free exemptions/permits

Under the current carbon pricing mechanism legislation some 'trade-exposed' industries whose emissions are included in the ETS receive 94.5 or 66 per cent shielding (Australian Government, 2011). We therefore analyse a situation where farm businesses are granted free exemptions/permits for 66 or 94.5 per cent of what their emissions would be at that carbon price if agriculture was excluded.

The on-farm emissions shown in the second column of Table 5.3 represent the point where the marginal opportunity cost of changing production to reduce emissions equals the marginal benefits of reducing payments for emissions. The granting of free exemptions/permits has no impact on the makeup of the optimal farming system (and hence also the level of on-farm emissions) if the quantity granted is less than these on-farm emissions, but it does counter reductions in farm profit (see the results for 66% free permits in Table 5.4). However, if the quantity granted is greater than the optimal level of on-farm emissions at that carbon price (i.e., there is an 'excess' – see results for Standard and Amended with 94.5% free permits in Table 5.4) then the effect of free permits depends on the policy settings. One possibility is that farms can sell any excess permits to emitters in other industries. Comparing Table 5.3 with Table 5.4 and Table 5.5 reveals that this would not alter the optimal farming strategy, but would provide a windfall to farmers.

Table 5.4. Characteristics of the optimal farming systems with an initial price of \$23/t of CO_2-e (where sequestration is unviable) and the granting of free permits/exemptions which could be on-sold.

Emissions method	**Free permits (t of CO_2-e yr^{-1})[a]**	**On-farm emissions (t of CO_2-e yr^{-1})[a]**	**Crop area (ha)**	**Sheep (DSE)**	**Excess free permits sold (t of CO_2-e yr^{-1})**	**Farm profit (\$'000)[a]**
Standard	689 (66%)	955 (-8.5%)	1503	2662	0	83.5 (-8.3%)
Amended	825 (66%)	1153 (-7.7%)	1504	2656	0	82.0 (-10%)
Local	380 (66%)	564 (-2.1%)	1472	2934	0	86.8 (-4.8%)
Standard	986 (94.5%)	955 (-8.5%)	1503	2662	31	90.3 (-0.8%)
Amended	1181 (94.5%)	1153 (-7.7%)	1504	2656	28	90.2 (-1.0%)
Local	544 (94.5%)	564 (-2.1%)	1472	2934	0	90.5 (-0.6%)

[a]Numbers in parentheses show per cent of emissions or profit when agriculture is excluded from the carbon price

Table 5.5. Characteristics of the optimal farming systems with sequestration estimated using either NCAT or Jonson (2010) at the initial price of $50/t of CO_2-e and the granting of free permits/exemptions which can be on-sold.

Emissions method	Free permits (t of CO_2-e yr^{-1})[a]	On-farm emissions (t of CO_2-e yr^{-1})[a]	Revegetated area (ha)	Excess free permits sold (t of CO_2-e yr^{-1})	Farm profit ($'000)[a]
Initial carbon price of $50/t of CO2-e (Sequestration estimated with NCAT)					
Standard	689 (66%)	516 (-50.5%)	0	173	74.6 (-11.7%)
Amended	824 (66%)	694 (-44.4%)	0	130	72.4 (-14.3%)
Local	380 (66%)	280 (-51.4%)	0	100	78.0 (-7.6%)
Standard	986 (94.5%)	516 (-50.5%)	0	470	89.4 (5.9%)
Amended	1180 (94.5%)	694 (-44.4%)	0	486	90.1 (6.7%)
Local	544 (94.5%)	280 (-51.4%)	0	264	86.2 (2.0%)
Initial carbon price of $50/t of CO2-e (Sequestration rate from Jonson (2010))					
Standard	660 (66%)	407 (-59.3%)	325	253	88.0 (-4.6%)
Amended	784 (66%)	531 (-55.3%)	325	253	86.7 (-6.1%)
Local	368 (66%)	255 (-54.3%)	325	113	88.8 (-3.8%)
Standard	946 (94.5%)	407 (-59.3%)	325	538	102.3 (10.8%)
Amended	1123 (94.5%)	531 (-55.3%)	325	592	103.6 (12.2%)
Local	527 (94.5%)	255 (-54.3%)	325	272	96.7 (4.8%)

[a]Numbers in parenthesis show per cent of emissions or profit when agriculture is excluded from a carbon price

Alternatively, if policy rules prohibit the sale of permits and more permits are issued than the farm would emit at that carbon price in the absence of free permits/exemptions, then it becomes optimal to increase on-farm emissions to the exact level of free permits (Table 5.6). Thus with on-selling prohibited the granting of free permits also reduces the impact of including agriculture in the ETS. Prohibiting or allowing on-selling of excess permits would not change government revenue because the same amount of permits/exemptions is issued (so the net reductions in emissions would also be equal). However the cost to society of these emissions reductions would be greater if the on-selling was prohibited because there would be less incentive for farmers to utilise any opportunities they have to reduce emissions for a lower cost per tonne than the emissions price.

There is an interaction between the effect of the different emissions accounting methods on profit and the granting of free permits. When no free permits are issued there are big differences in emission liabilities and thus, especially at high carbon prices, large differences in farm profit arise between the emissions accounting methods (Table 5.3). Likewise if on-selling occurs then profit differences between the methodologies increase as the amount of excess free permits that are on-sold increases, especially at higher C-prices (Table 5.4 and Table 5.5). However, when the level of free permits is similar to the level of on-farm emissions, or if the on-selling permits/exemptions is prohibited (Table 5.6), the profit difference between the methods narrows.

Table 5.6. Characteristics of the optimal farming systems when agriculture is included in the carbon price but shielded by the granting of free permits/exemptions which cannot be on-sold.

Emissions method	Free permits (t of CO_2-e yr^{-1})[a]	On-farm emissions (t of CO_2-e yr^{-1})[a]	Revegetated area (ha)	Farm profit ($'000)[a]
Initial carbon price of $23/t of CO_2-e (Sequestration unviable)				
Standard	689 (66%)	955 (-8.5%)	0	83.5 (-8.3%)
Amended	825 (66%)	1153 (-7.7%)	0	82 (-10.0%)
Local	380 (66%)	564 (-2.1%)	0	86.8 (-4.8%)
Standard	986 (94.5%)	986 (-5.5%)	0	90.2 (-1.0%)
Amended	1181 (94.5%)	1181 (-5.5%)	0	90.1 (-1.1%)
Local	544 (94.5%)	564 (-2.1%)	0	90.5 (-0.6%)
Initial carbon price of $50/t of CO2-e (Sequestration estimated with NCAT)				
Standard	689 (66%)	689 (-34.0%)	0	74.1 (-12.2%)
Amended	824 (66%)	824 (-34.0%)	0	72.1 (-14.6%)
Local	380 (66%)	380 (-34.0%)	0	77.7 (-8.0%)
Standard	986 (94.5%)	986 (-5.5%)	0	83.6 (-1.0%)
Amended	1180 (94.5%)	1180 (-5.5%)	0	83.5 (-1.1%)
Local	544 (94.5%)	544 (-5.5%)	0	83.8 (-0.7%)
Initial carbon price of $50/t of CO2-e (Sequestration rate from Jonson (2010))				
Standard	689 (66%)	689 (-31.2%)	325	86.8 (-5.9%)
Amended	784 (66%)	784 (-34.0%)	325	84.7 (-8.3%)
Local	368 (66%)	368 (-34.0%)	325	88.1 (-4.6%)
Standard	986 (94.5%)	986 (-1.5%)	202	92.3 (-0.1%)
Amended	1123 (94.5%)	1123 (-5.5%)	238	92.1 (-0.2%)
Local	527 (94.5%)	527 (-5.5%)	192	92.2 (-0.2%)

[a]Numbers in parentheses show per cent of emissions or profit when agriculture is excluded from the carbon price

In summary, free permits greatly reduce the financial impact of carbon pricing on agriculture, without altering the level of emissions (unless excess permits are granted to farmers and they cannot be on-sold).

5.4.3.3 Implications of inaccuracy in methods

A major use of GHG accounting is to determine a country's emissions trend. Winiwarter and Rypdal (2001) suggest that uncertainty associated with methods used in GHG accounting may have minimal impact on trend estimates if sources of error behave similarly on a yearly basis. However, when policies like a carbon price are implemented to change these trends, measurement uncertainty could become problematic. It may result in the erroneous ranking of the importance of different sources of emissions and the per-unit cost of reducing them. Our results show that an emissions policy based on incorrect estimates of emissions can result in emitters being charged for emissions that in reality are much different. Moreover, firm behaviour can be altered in ways that make the policy inefficient.

This efficiency loss is illustrated by comparing the Standard and Local accounting methods at a price of $23/t of CO_2-e with no sequestration allowed (Table 5.7). Compared to farm income and government revenue, deadweight losses of under $500 suggest that the inefficiency losses from using incorrect methods are relatively minor to society as a whole. However, the losses borne by particular groups (farmers or the government) will be much larger. In this case study, the Standard accounting method would significantly disadvantage farmers relative to a method based on more locally accurate data. There may be other parts of Australia where farmers are advantaged by the use of the Standard method.

As part of its package of legislation for the ETS the Australian government also created the ‘Carbon Farming Initiative’. This initiative allows farmers the option of claiming and selling offsets for voluntarily undertaking actions that mitigate emissions. The quantity of offsets that can be claimed for a given action is governed by a series of rules including one for ‘leakage’. Leakage is when an action that mitigates emissions indirectly causes other emissions (potentially in another location, time or different form of GHG) to increase. As leakage nullifies abatement that would otherwise result from the mitigation activity, it must be subtracted when calculating net abatement. If this leakage is in the form of on-farm emissions and is incorrectly estimated at the farm level with methodologies used in national accounting then it will either cause offsets to be more expensive than they should be, or result in a net increase in atmospheric GHGs whilst giving the false impression that no net change in emissions had occurred due to the offset.

Table 5.7. The implications of applying a $23/t of CO_2-e price to the 2000 ha farm using accurate methods versus inaccurate methods.

	Local method is accurate		**Standard method is accurate**	
	Local method is applied	**Standard method is applied**	**Standard method is applied**	**Local method is applied**
Cost to producer	-$13,086	-$23,451	-$23,451	-$13,086
Transfers to government	$12,965	$21,974	$21,974	$12,965
Benefits of abatement	$278	$1,214	$2,036	$320
Net benefit to society	$157	-$263	$558	$199
Deadweight loss	-	$421	-	$359

5.5 Conclusion

Different methods for measuring agricultural emissions can generate very different estimates of emissions. This paper has explored, for different emission policy scenarios, the economic consequences of using different emission measurement methods, focusing on consequences for farmers. If agricultural emissions are covered under a carbon pricing scheme, the emissions accounting method can significantly affect farm profit. The method for measuring carbon sequestration can also make a large difference to how much farm area is reforested and thus also affects the impact of a carbon price on farm businesses.

Even if agricultural emissions are excluded from a domestic carbon price the profit of a farm producing primarily for export markets will fall due to increased input costs. However, the reduction in profit is limited by competition from imported inputs not subject to the carbon price and/or government protection for local manufacturers and so substantial changes to the enterprise mix of the farming system is unlikely.

Sequestration may lessen the impact of a carbon price on farm businesses. However, for the farming system examined, a high carbon price is required for sequestration to be viable, especially if sequestration rates are low, or underestimated through use of an inaccurate measurement method.

If on-farm emissions are subject to a domestic carbon price then the impact on farm profit (without compensation) is large, and agricultural emissions do reduce. Grazing production is most affected as livestock are the dominant source of emissions. Mixed cropping-livestock farming systems would become more crop orientated.

If a carbon price is applied to agricultural emissions that are incorrectly estimated then the deadweight inefficiencies generated by inaccurate methods may not be large. It would, however, raise issues of equity and fairness as the impacts of inaccurate accounting methods on costs to producers and transfers to government can be large. Hence, the recent allocation of research funds under the *Filling the Research Gap* program (DAFF, 2012) that aims to provide greater accuracy in emissions measurement is likely to be an appropriate investment.

5.6 Online appendix

This online appendix/supporting information describes in detail the methodology and emissions factors (EFs) used to account for on-farm emissions from CH_4 from livestock; N_2O from animal waste, N fertiliser, biological N-fixation and crop residues; and CO_2 from urea hydrolysis for each of the three greenhouse accounting methodologies: Standard, Amended and Local.

Note unless referenced otherwise, the methodology and parameter values described here were sourced from National Inventory (2011).

5.6.1 Enteric CH_4 from livestock

To model these emissions in MIDAS Petersen et al. (2003b) used the close relationship between feed intake and enteric CH_4 production in sheep (Howden et al., 1994). Current National Inventory (2011) methodologies still account for enteric emissions in the same way:

$$\text{enteric emissions (kg of } CH_4\text{/head/day)} = I \times 0.0188 + 0.00158 \qquad \textbf{Eq 5.1}$$

where *I* is the kg/head of dry matter consumed daily by each type of livestock as estimated endogenously in MIDAS. CH_4 was converted to CO_2-e using a global warming potential of 21. Although there is uncertainty about estimates of enteric methane emissions predicted in this way, details of how (and importantly why) they may vary for the study area are scant. Therefore Eq 5.1 was used as the basis to calculate enteric emissions for all three methodologies.

5.6.2 CH_4 emissions from animal waste

These emissions were omitted from all methodological scenarios as CH_4 production from manure is likely to be negligible for free-ranging animals in Australian conditions (National Inventory, 2011).

5.6.3 N_2O emissions from Animal Waste

National Inventory (2011) calculates N_2O emissions from animal wastes based on the following:

Faeces: 0.005 Gg N_2O-N/ Gg N excreted by the animal (i.e., an EF of 0.5%) **Eq 5.2**

Urine: 0.004 Gg N_2O-N/ Gg N excreted by the animal (i.e., an EF of 0.4%) **Eq 5.3**

These EFs, which were used for both the Standard and Amended methodologies, are based on international studies and are designed to be applied universally for both sheep and cattle. There is a dearth of data on N_2O emissions from sheep, not only in the study area, but also in Australia more generally.

The lowest rates of the N_2O emissions from animal waste found in a review by Oenema et al. (1997) were from extensively grazed sheep. Given N_2O emissions have been found to be lower from sheep than cattle manure (van der Weerden et al., 2011), and the EF recommended by the IPCC for cattle is twice that from sheep (IPCC, 2006b), emissions from sheep only are likely to be lower than the composite, single-species estimates for faeces and urine above.

Trials in New Zealand have found the following EFs: cattle urine 0.29%, sheep dung 0.01%, (which was not significantly different from zero) (van der Weerden et al., 2011); sheep urine 0.1 to 0.14% (Hoogendoorn et al., 2008); sheep dung 0.0% (de Klein et al., 2004). Part of the rationale that National Inventory (2011) used for excluding the CH4 emissions associated with manure from free-ranging livestock was that the typical Australian environmental conditions of low humidity and high temperatures and solar radiation combined with the prevalence of scarab (dung) beetles reduces the likelihood of manure becoming anaerobic. Anoxic conditions favour N_2O production (Oenema et al., 1997; van der Weerden et al., 2011). Given that Cunderdin is drier and warmer than any of the trial sites from the aforementioned studies and New Zealand lacks dung beetles, it seem reasonable to expect that if anything, EFs for Cunderdin would be lower than for New Zealand. For this reason, an EF of 0.0014 (0.14%) for sheep urine and 0.0001 (0.01%) for faeces were used in the Local methodology, although the lack of field data is acknowledged.

5.6.4 N_2O emissions from nitrogenous fertiliser

For the Standard and Amended methodologies the formula of National Inventory (2011) was used:

$$\text{kg of } CO_2\text{-e per } Nfert = Nfert \times EF \times CF \times 310 \qquad \textbf{Eq 5.4}$$

where *Nfert* is the kg of elemental nitrogen applied in fertiliser per ha, *EF* is the emissions factor for the proportion of N fertiliser emitted as N_2O set at 0.003 (0.3%)

(National Inventory, 2011) and *CF* is the standard conversion factor (44/28) to convert the elemental mass of N_2O to molecular mass. 310 is the global warming potential of N_2O over 100 years. The exact same formula was used for the Local methodology, except that the *EF* was instead set to 0.0004— based on averaging the results from two local studies performed in Cunderdin (the study area) (Barton et al., 2008; Barton et al., 2010).

5.6.5 N_2O emissions from N fixation by legumes

According to National Inventory (2011) N_2O is emitted by legumes when they fix atmospheric N. Therefore for the Standard methodology estimates of the amount of N fixed are derived using the amount of biomass produced and N in that biomass which is fixed as follows:

$$CO_2\text{-e from N fixation} = M \times R \times DM \times CC \times NC \times EF \times CF \times 310 \qquad \textbf{Eq 5.5}$$

where *M* is the grain yield in the case of legume crops or biomass production per ha in the case of legume pasture (since the emissions being estimated relate to the actual process of biological N fixation, this is the mass produced before grazing). For pastures this is the mass of the legume component of the pasture and not the biomass of the whole sward, something not made explicitly clear in National Inventory (2011). *R* is the ratio of stubble or residue to grain, assumed to be 2.1 for grain legumes and 1.0 for pastures. *DM*—set at 0.8—is used to convert the stubble from legume crops into dry weight terms. However, this was set to 1.0 for pastures as their production is already expressed in the dry matter terms in MIDAS. *CC* is the proportion of the legume residues that is carbon, set at 0.4 for both crops and pastures. The nitrogen to carbon ratio of legumes (*NC*) was set at 0.05 for legume crops and 0.08 for legume pastures. The *EF* was set 0.0125 for Eq 5.5, whilst *CF* was the same as in Eq 5.4.

One of the inconsistencies of the Standard methodology of National Inventory (2011) is that to derive an estimate of the amount of N fixed by biological processes they use the amount of stubble from legume crops and the N content of that stubble. Therefore they fail to take into account the N that was fixed whilst the crop was growing but later removed in the harvested legume grain. For the Amended methodology this was taken into account by modifying Eq 5.5:

$$CO_2\text{-e from N fixation} = ((M \times R \times DM \times CC \times NC) + (Ng + Y)) \times EF \times CF \times 310 \qquad \textbf{Eq 5.6}$$

to include *Ng*—the N content of the legume grain—calculated on the basis that each kg of lupin, field pea, faba bean and chickpea yielded contained 0.053, 0.038, 0.041, 0.033 kg of N respectively (Grain Legume Handbook, 2008). The grain yield itself is represented by *Y*.

Biological N fixation is still included as an emission source in National Inventory (2011) although many studies (including that of Barton et al. (2011) in the actual Cunderdin area) have failed to demonstrate that biological N fixation actually causes appreciable N_2O emissions, such that IPCC (2006a) removed it as source from their accounting methodologies (instead N_2O associated with legume production is accounted for purely under the methodology for residue decomposition—see below). Hence no emissions from N-fixation were included in the Local scenario.

5.6.6 N_2O emissions from residues

As crop residues decompose the N they contain can be emitted as N_2O. For the Standard methodology this was predicted by:

$$CO_2\text{-e from residues} = M \times R \times DM \times CC \times NC \times (1 - FR) \times EF \times CF \times 310 \qquad \textbf{Eq 5.7}$$

Where *M* is the grain yield of the crop per ha. *R* was set at 1.5 and 2.1 for non-legumes and pulses respectively. The *DM* was assumed to be 0.9 for wheat and 0.8 for all other crops including grain legumes and NC was set at 0.008 for cereals and 0.05 for pulses. *FR* represents the fraction of the residue is removed, set at 0.09 for all crops. The *CC*, *EF* and *CF* had the same values as in Eq 5.5.

Another of the inconsistencies in National Inventory (2011) is that although they count N_2O emissions from both the N-fixing process and the break-down of residues for legume crops, they only count emissions from N-fixation for pastures. That is, they ignore that legume pastures have N-rich residues that must also decompose. Furthermore, to consider the pasture residues properly, one must consider the legume and non-legume components of the sward separately due to their different N contents. This was done in the Amended scenario by including the following in emissions accounting (in addition to Eq 5.7):

CO_2-e from pasture residues =

$$[(M \times FL \times CC \times NC_l) + (M \times (1-FL) \times CC \times NC_{nl})] \times (1-FR) \times EF \times CF \times 310 \qquad \textbf{Eq 5.8}$$

where *M* is the biomass of the whole pasture sward per ha in dry matter terms. The fraction of the sward that is made up of legumes (*FL*) was source endogenously from MIDAS, but was generally 30 –60 per cent for annual pastures and 60 per cent for lucerne. The nitrogen to carbon ratio of the pasture residue was set to 0.08 for the legume component (NC_l) and in the absence of data for the non-legume component from National Inventory (2011), NC_{nl} was set to 0.039 (Gladstones and Loneragan, 1975). To recognise that much of pasture residue maybe removed by grazing before it decomposes, FR was set at the amount of pasture residue remaining after grazing that MIDAS determines endogenously. Once again *CC*, *EF* and *CF* were set to the same values as Eq 5.5.

For the Local methodology N_2O emissions from crop residues were also estimated using Eq 5.6, except with parameters based on best available data. *R* was set to the following: wheat 2.19; barley 2.00; oats 2.40; canola 2.19; lupin 2.75; faba bean and field pea 2.57; chickpea 3.17. These were sourced from the existing values in MIDAS which were themselves based on local field data. The values of *NC* for Local were based on the mean values of local datasets (numbers in parenthesis indicate the number of studies used to calculate the mean): cereals 0.015 (22); canola 0.016 (11); and pulses 0.024 (9) (Gladstones and Loneragan, 1975; Schultz and French, 1978; unpublished field trials; Barton et al., 2011). The proportion of residues removed, or *FR*, was set to the constraint for maximum amount of stubble removal that is permitted in each MIDAS run—typically 0. 5. An *EF* of 0.001 was used for two reasons. Firstly, when used with the other local parameters, it closely predicted the post-harvest N_2O emissions from legume residues actually measured by Barton et al. (2011) in the study area[3]. Secondly, modelling done by (Li et al., 2011) suggested that an *EF* of approximately 0.001 would have been appropriate across 37 years of meteorological data at Cunderdin. The values for *DM*, *CC* and *CF* were the same as used in the Standard methodology.

[3] Barton et al. (2011) ceased their measurement just before the sowing of the next crop, so it is possible that N_2O emissions might have be greater after measurements ceased due to the breakdown of the residues the following winter. However the authors of that study thought this unlikely and, more recent as of yet unpublished trials have found no significant difference in N_2O emissions between wheat grown after leguminous lupins or wheat (with an equivalent amount of N supply) (L. Barton pers. comms.).

There have been no studies on N_2O emissions from pasture residues in the Cunderdin area. However theoretically there is no reason why N in pasture residues couldn't be emitted as N_2O, just as N in the residues of (legume) crops can be. Thus, the possibility of N_2O emissions from pasture residues were included for Local, with the same Eq 5.8 used for the Amended methodology, and also the same values for the parameters with a couple of exceptions. The NC_l for the legume component of the pasture residue was set to 0.084 (mean of seven datasets) (Gladstones et al., 1975; unpublished trials). Lastly, the local *EF* of 0.001 for the crop residues was also used for the pastures residues.

5.6.7 Urea Hydrolysis

Another source of on-farm emissions included in other LCA studies of the Wheatbelt region is the hydrolysis of urea ($CO(NH_2)_2$) fertiliser (e.g., Biswas et al., 2008).Therefore CO_2 emissions from urea hydrolysis were included using an EF of 20 per cent of the urea applied (IPCC, 2006b) for all three methodology scenarios.

5.6.8 Emissions from on-farm fuel use

To estimate costs, the amount of fuel used on-farm for every activity including crop establishment, harvest, livestock husbandry, chemical and fertiliser application were already accounted for in MIDAS. Therefore to report the emissions in all three scenarios, these estimates of fuel use were combined with EFs from NGA Factors (2010) (e.g., 2.7 kg of CO_2-e /L of diesel).

Chapter 6. Paper 5. Climate change impacts and farm-level adaptation: economic analysis of a mixed cropping-livestock system

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6.1 Preface

Readers may notice that estimates of farm profit from MIDAS reported in the next two chapters differ from those reported previously in Chapter 3 and Chapter 5. There are two reasons for this. Firstly, subsequent to publishing the analyses reported in Chapters 3 and 5, updates were made to the MIDAS model to keep pace with changes in Wheatbelt farming systems (for instance farm size was increased in the version of the model used in this Chapter 6 and Chapter 7) and prices were also updated. Secondly, there are differences in the composition of the profit figures reported. In all chapters the profit was calculated as the (pre-income tax) farm income minus variable cash costs, as well as non-cash costs like machinery depreciation, and fixed overheads like household expenses and hiring of professional services. The opportunity cost of the capital invested in the livestock flock, machinery or land was also deducted from the profit values reported in Chapter 3 and Chapter 5. However, in Chapter 6 and Chapter 7 these opportunity costs have not been deducted, meaning that the annual net return reported in these next two chapters is considerably larger, but importantly, potentially more representative of the way farmers typically consider financial outcomes.

Climate change impacts and farm-level adaptation: economic analysis of a mixed cropping-livestock system

6.2 Abstract

The effects of climate change on agricultural profitability depend not just on changes in production, but also on how farming systems are adapted to suit the new climatic conditions. We investigated the interaction between production changes, adaptation and farm profits for a mixed livestock-cropping farming system in the Western Australian Wheatbelt. Crop and pasture production were simulated for a range of plausible rainfall, temperature and CO_2 concentrations for 2030 and 2050. We incorporated the results of these simulations into a whole-farm bio-economic optimisation model. Across a range of climate scenarios, the impact on farm profit varied between −103% to +56% of current profitability in 2030, and −181% to +76% for 2050. In the majority of scenarios profitability decreased, and the magnitude of impacts in negative scenarios was greater than the upside in positive scenarios. Profit margins were much more sensitive to climate change than production levels (e.g., yields). Adaptive changes to farm production under extreme climate scenarios included reductions in crop inputs and animal numbers and, to a lesser extent, land-use change. The whole-farm benefits of these adaptations were up to $176,000/year, demonstrating that estimating the impact of climate change without allowing for adaptation can substantially inflate costs. However, even with adaptation, profit reductions under the more negative scenarios remained large. Nevertheless, except for the most extreme/adverse circumstances, relatively minor increases in yields or prices would be sufficient to counteract the financial impacts of climate change (although if these price and/or productivity increases would also have occurred without climate change then the actual cost of climate change may still be high).

Keywords: adaptation, mixed farming, dryland farming, optimization, profit, climate change impacts

Highlights

- Whole-farm analysis of climate change impacts on a mixed cropping-livestock system
- Impacts varied. In general, potential losses much greater than potential gains

- Benefits of adaptation were substantial, but in adverse scenarios costs still high
- Not allowing for adaptation inflated the cost of climate change by 15–35%
- Yield or price increases could offset much or all of the impact of climate change

6.3 Introduction

The effect that climate change has on the productivity and economic viability of agriculture will depend on how much it is possible to adapt to reduce the change's impact (Lobell, 2014). Therefore, estimates of the economic impact of climate change will likely be overstated if adaptation is not allowed for. Nonetheless, in many existing projections of climate change impacts adaptation is not considered (White et al., 2011).

We investigate the impact of climate change, allowing for adaptation, in the Wheatbelt region of Western Australia. In this region the agricultural growing season is limited by moisture availability and as the region is predicted to warm and dry with climate change (e.g., Moise and Hudson, 2008; Turner et al., 2011) the dryland agriculture practiced there is potentially vulnerable. Climate change may already be affecting the region: average growing-season rainfall (May to October) has declined by more than 10% since the 1970s (Ludwig et al., 2009). Interestingly, despite this, farms in the region experienced high yield and productivity growth in the 1980s and 1990s (Islam et al., 2014). However, more recently, average yields appear to have stabilised (Turner et al., 2011; Stephens et al., 2012).

Studies of the economic impacts of climate change that incorporate agricultural adaptation need to encompass: (a) the impacts of climate change on the production of outputs in various possible production systems, and (b) an economic assessment of the impact of these production changes and the options for adaptation that are available to the farmer. Aspect (a) is often addressed using detailed plant and/or animal simulation models, and there have been a number of studies of this type for the case-study region (van Ittersum et al., 2003; Asseng et al., 2004; Ludwig and Asseng, 2006; Ludwig et al., 2009; Farre and Foster, 2010; Asseng and Pannell, 2013; Moore and Ghahramani, 2013; Anwar et al., 2015).

Aspect (b) has been much less thoroughly researched for the study area. There are two main approaches that can be used to investigate it. The first is to identify packages of adaptations that are of interest and then simulate the economic consequences of each package (e.g., Crimp et al., 2012; Ghahramani et al., 2015). An advantage of this approach is that the modeller has complete control over which adaptations are simulated, allowing transparent analysis of particular strategies that are of interest. Deciding which packages of adaptations to simulate can be problematic though (White et al., 2011), particularly in complex mixed farming systems such as those found in the case-study region. The modeller may not be able to anticipate which of the many potential combinations of adaptations are most likely to be worth assessing.

The second approach is to use optimisation to automatically assess all of the available combinations of adaptations. The obvious advantage is avoiding the need for numerous simulations to identify the adaptations that best meet the farmers' economic objectives (Klein et al., 2013). However, the analysis may be less transparent than under the simulation approach, and the objective function used in the optimisation model may not match that of all farmers.

In this study, we utilise process-based simulation models for the first phase, and extensively modify an existing bioeconomic whole-farm optimisation model for the second. We judged that the very large number of production options available in our case-study region means that the advantages of the optimisation approach outweigh its disadvantages. Also, previous analyses of climate change impacts on the case-study region have tended to consider impacts on a solitary crop or enterprise in isolation. Our use of a whole-farm model allows the simultaneous consideration of impacts on all elements of a typical farming-system in the region. Amongst other things, this allows adaptation in the form of changing land use to be represented in our study (Reidsma et al., 2015).

Our aim is to explore potential impacts of future climate change on production and profitability in the West Australian Wheatbelt. Specifically we address the following questions: 1) What is the impact on farm production and profits under a range of realistic climate scenarios over the next 15 to 35 years?; 2) Which currently available adaptations are most effective in moderating any adverse effects or exploiting positive effects, and to what extent do they improve farm profits?; Finally, 3) What increase in prices or yields would be needed to maintain profits equivalent to the no-climate-change scenario?

6.4 Methodology

6.4.1 Study area

The Western Australian Wheatbelt region accounts for approximately 40% of the wheat and 11% of the wool exported by Australia (around 5% and 7% of the wheat and wool traded internationally—ABARES, 2013). Our study area is the central part of this Wheatbelt region, around the township of Cunderdin (Figure 6.1). This area has a Mediterranean-type climate with long, hot and dry summers and cool, moist winters. Historically annual rainfall is between 330 and 400 mm, approximately 75% of which falls during the May to October growing season.

Farms in the area are commonly 2000 –4000 hectare, of which 65 –85% is typically sown to annual crops in May and June; the remaining areas are pastured, supporting sheep for meat and wool production. Farming systems are solely rain-fed, and after harvest in December, the remaining crop residues are utilised in-situ as dry fodder. Once this feed supply is exhausted, livestock receive a grain-based supplementary ration until adequate green pasture becomes available after the onset of winter rains (Rowe et al., 1989).

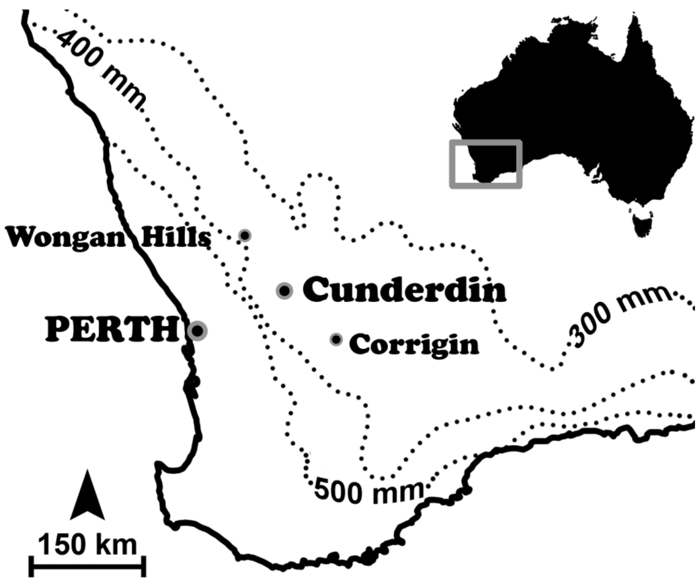

Figure 6.1. Our Central Wheatbelt study area is centred on the Cunderdin Township. Precipitation isohyets are based on historical observations.

6.4.2 Farm-level modelling

The economic impact of climate change was evaluated using MIDAS (Model of an Integrated Dry Land Agricultural System—Morrison et al., 1986; Kingwell and Pannell, 1987). MIDAS has been used extensively to explore the impacts of innovations, policy changes and environmental degradation on mixed cropping-livestock farms (e.g., Doole et al., 2009; Monjardino et al., 2010; Robertson et al., 2010; Kragt et al., 2012). MIDAS is deterministic, based on an ‘average’ weather-year in the study area (although the region’s Mediterranean-type climate is semi-arid, historically, the variability in this climate has been relatively low, making the steady-state modelling framework of MIDAS justifiable—Kingwell, 2011).

MIDAS uses a linear-programming algorithm to maximise farm net return subject to resource, environmental, and managerial constraints, including machinery capacity and the availability of land, labour and capital. MIDAS contains approximately one thousand activities, including: a range of rotations with different sequences of crops and pasture for each soil type; feed supply and utilisation by different classes of livestock; different crop sowing dates (and yield penalties for delays to sowing); cash flow recording and; machinery and overhead expenditures. MIDAS captures biological and technical relationships at the farm-level, particularly interdependencies between enterprises such as the benefits of nitrogen fixation, the yield-enhancing (e.g., disease-break) effects of crop rotation, the value of crop residues as animal feed, the effects of cropping on subsequent pasture growth and the effect of weed burdens for subsequent crops.

For this study the Central Wheatbelt MIDAS used in recent studies (Kragt et al., 2012; Thamo et al., 2013) was updated to reflect changing trends by increasing the capacity and value of machinery. Farm size was also increased to 3200 arable hectares. The MIDAS farm contains eight different soil types with varying production characteristics, as farms in the study area typically contain a mix of soil types (for descriptions of, and areas assumed for each soil type see the Supplementary Material). Land-uses represented in the model include rotations of wheat (*Triticum aestivum),* barley (*Hordeum vulgare),* oats (*Avena sativa),* lupins (*Lupinus angustifolius),* canola (*Brassica napus),* and annual legume-based pastures. The annual net return we report represents the pre-tax profit after deducting variable costs, as well as non-cash costs like depreciation, and fixed overheads like household expenses and hiring of professional services. For the present study we added the option of retiring land from production, the rationale being if climate change renders agricultural production unprofitable, a producer's optimal response may be to 'retire' from production their least productive land to minimise their losses. Unlike the temporary fallowing of land, land retirement is purely a loss-minimisation activity that neither generates income nor incurs costs (overheads associated with maintaining the farming enterprise as whole are still incurred).

The predicted impacts of changes in climate and atmospheric CO_2 levels (hereafter called 'climate scenarios') on farm production were incorporated into MIDAS. This was done by using biophysical simulation models (described in Section 6.4.4) to estimate the effect of a given climate scenario on agricultural production, and then based on these results, the growth potential of crops and pastures in MIDAS were scaled.

6.4.3 Climate projections and scenarios

In the most recent comprehensive climate projections for the study region, Hope et al. (2015) collated the results of over 40 Global Climate Models (GCMs) from the Coupled Model Intercomparison Project Phase 5 (CMIP5) ensemble of climate models (this ensemble underpins the Intergovernmental Panel on Climate Change's Fifth Assessment Report). Compared to the 1986–2005 period, Hope et al. (2015) predicted with high confidence that annual rainfall in the study region will decrease, with June to November (the agricultural growing season in the study region) rainfall in the study region projected to change by −15% to +5% by 2030, and −45 to -5% by 2090. They also predicted that average temperatures will increase (in the order of 1.2 –4.0°C by 2090, equally across all seasons). These projections are consistent with earlier studies, and indeed, decreases in rainfall and increases in temperature already observed in the study region in recent decades (Moise and Hudson, 2008; Hennessy et al., 2010; Delworth and Zeng, 2014).

Although the direction of climatic changes predicted for the study region is relatively clear, particularly in the long-run, the magnitude of these changes is less certain. This uncertainty arises because of variation between the results of different GCMs, limitations of GCMs in general, and uncertainty about future global emissions (Hennessy et al., 2010; Asseng et al., 2013).

To reflect this uncertainty we therefore considered a wide range of changes in climatic parameters and atmospheric CO_2 concentrations in our analysis. A similar factorial approach has been used in other climate change studies of the study region (e.g., van Ittersum et al., 2003; Ludwig and Asseng, 2006). In total we considered 72 climate scenarios: the factorial combination of three CO_2 levels, three temperature increases and four rainfall reductions for each of the years 2030 and 2050 (Table 6.1). The magnitude of these changes were chosen because they were consistent with the trend of projections from the literature, yet deliberately broad enough to capture a wide range of climatic possibilities, allowing us to explore the sensitivity of the agricultural system to changes.

The climate scenarios were generated by 'changing' the historic weather (herein this historic weather—meteorological data observed from 1957 –2010 at Cunderdin and a constant concentration of 390 ppm atmospheric CO_2—is referred to as the 'base-case' climate). So for instance, for the scenario of 525ppm CO_2 / 20% rainfall reduction /2.5°C increase (or '525/-20/2.50'), the atmospheric CO_2 level in the biophysical simulation models were set to 525ppm, all rainfall observations in base-case dataset were reduced by 20% (changing the intensity but not regularity of rainfall), and the maximum and minimum temperature observations were increased by 2.5°C[1].

Table 6.1. Factorial combinations of the following changes in climate and CO_2 were investigated for 2030 and 2050.

Years	CO_2 (ppm)	Rainfall reduction (%)	Temperature increase (°C)
2030	425	0	0.50
	450	-5	1.25
	475	-10	2.00
	-	-15	-
2050	475	0	1.00
	525	-10	2.50
	575	-20	4.00
	-	-30	-

[1] With the evaporation rate and vapour pressure deficit derived endogenously within each biophysical simulation model, changes in these two parameters due to the changes in temperature were also taken into account. However, this required that the vapour pressure in the meteorological dataset be recalculated exogenously (Allen et al., 1998) after the temperature was scaled.

6.4.4 Simulating the biophysical impact of climate change and incorporating the results into MIDAS

The effect of climate and CO_2 change on crop yields and pasture growth was simulated with the models Agricultural Production Systems Simulator (APSIM, ver 7.5) (Keating et al., 2003; Holzworth et al., 2014) and GrassGro® (ver 3.2.6) (Moore et al., 1997) respectively. Both of these biophysical models have been extensively applied to the study area, including for climate change research (e.g., Asseng et al., 2013; Moore and Ghahramani, 2013; Anwar et al., 2015; Ghahramani et al., 2015). These models were calibrated for the eight soil types in MIDAS under base-case climatic conditions. To incorporate the predictions of these simulation models into MIDAS, the yield of crops and growth of pastures in MIDAS were scaled based on their relative difference between the base-case scenario and the given climate scenario predicted by the biophysical simulation models. This meant the relative change in crop yield-potential or pasture growth-potential predicted by the biophysical models for each soil type, with each climate scenario was emulated in MIDAS. MIDAS was then run like normal, to select profit maximising land uses, management strategies and input levels for each scenario.

Currently, APSIM lacks the capacity to simulate the effect of elevated CO_2 on many crops other than wheat. Consequently, in our analysis the impact of CO_2 increases on barley, oats, lupins and, to a lesser degree, canola, was based on APSIM's results for wheat. Additional details on this, how we took into account the potential for climate change to impact pasture growth differently at different times of the year and/or different stocking rates, and the parameterisation of the biophysical models in general can be found in the Supplementary Material.

6.4.5 Prices

MIDAS was configured with 2013 prices, except for the more volatile fertiliser, grain and livestock prices that were instead set to five year (2009 –2013) average prices in real terms (these prices are listed in the Supplementary Material). No systematic, longer term changes in prices (and/or productivity) were implicitly considered, meaning our analysis is contingent upon the assumption that farming productivity and prices of inputs and commodities are not changed fundamentally in the future.

6.4.6 MIDAS validity

MIDAS has been extensively tested in Western Australia for around 30 years since its creation by Morrison et al. (1986). It has been frequently updated to reflect changes in prices, costs, resources, farming systems and technologies. Although, as an optimisation model, the sort of validation strategies used for simulation models are not applicable, wide exposure and critique of results by experts has established that results and behaviour of the model are realistic and well aligned with actual farms in the region. The model naturally has limitations. Perhaps most importantly for this study, it represents farming under average and deterministic weather and price conditions. This means interactions between climate change and seasonal variability/risk, such as the role of enterprise diversification in building more resilient, stable farming systems (e.g., Kandulu et al., 2012), could not be considered in the present analysis.

A comparison of profits, yields and land uses predicted by MIDAS and the results of empirical farm surveys is available in the Supplementary Material (Section 6.9.3). It shows that the proportion of the farm cropped, sheep numbers, profit and yields predicted by MIDAS under base-case climate are broadly consistent with common practice in the study area.

6.5 Results

6.5.1 Impact of climate change on profitability

The analysis indicates that farm profitability is sensitive to changes in annual rainfall, temperature and CO_2 even after allowing for the most beneficial adaptations (Figure 6.2 and Figure 6.3). Of the 36 scenarios selected to represent the range of possible circumstances for 2030 (Figure 6.2), six result in profit increasing by more than 10% relative to the base case, four give profits within 10% of the base case, and 26 result in profits falling by more than 10%. The potential for losses is much greater than the potential for gains; there are 13 scenarios in which the loss of profits is greater than 50%, generally where temperature is highest and/or rainfall is lowest.

Not all of these scenarios are equally likely. At the lowest CO_2 concentration, relatively low changes in temperature and rainfall are more likely, increasing the chance that the impact on profit will be moderate, or even positive. At the highest CO_2 concentration, more extreme changes are relatively likely. Although they are offset to some extent by the benefits of high CO_2 for plant growth, overall the more likely effects on profit at high CO_2 are highly negative.

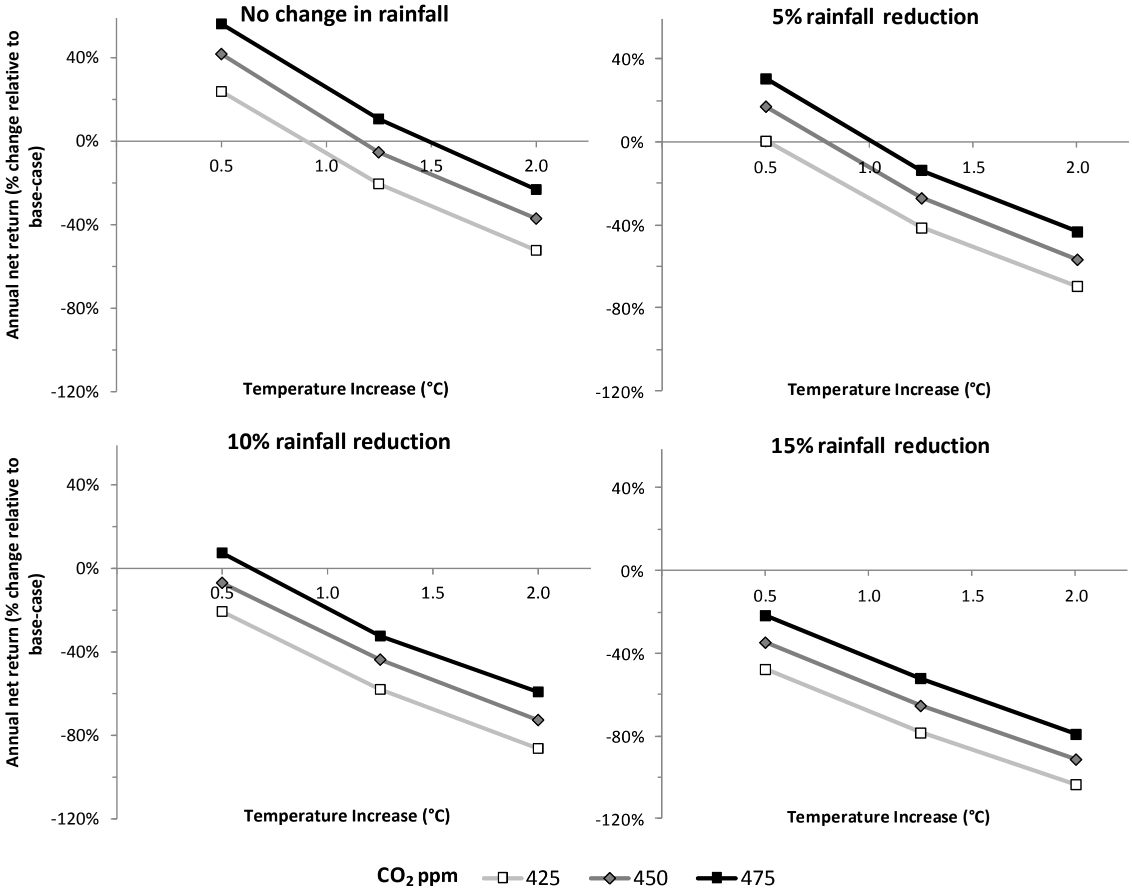

Figure 6.2. Percentage change in net return relative to base-case net return (of $208,000) for the 36 climate scenarios for 2030.

The 36 scenarios modelled for 2050 reflect the potential for larger changes in temperature, rainfall and CO_2 by that time (Figure 6.3). The possibility of a positive impact on profits is less likely than for 2030, with only four of the 36 scenarios resulting in profit increases above 10%. By contrast, there are now 30 scenarios that produce profit decreases greater than 10%, including 24 where the profit falls by more than 50%.

Figure 6.3 reveals that there are interactions between rainfall, temperature and CO_2 changes. The greater the rainfall reduction, the less responsive profits are to temperature or CO_2 concentration. Conversely, the greater the temperature increase, the less the impact of rainfall reductions or CO_2 increases.

Across all 72 scenarios, if there is either a greater than 2.5°C temperature increase or greater than 20% rainfall reduction, then regardless of what happens to the other climate parameters, farm profit falls compared to the base-case. If changes in climate are minor, then the implications for farm profit can be quite positive due to CO_2 increases and the beneficial impacts of small increases in temperatures. On the other hand, if the more extreme negative climate outcomes are realised in the 2050 scenarios, the consequences for farmers, in the absence of effective and novel adaptations, would be substantial, even after accounting for the positive effects of CO_2.

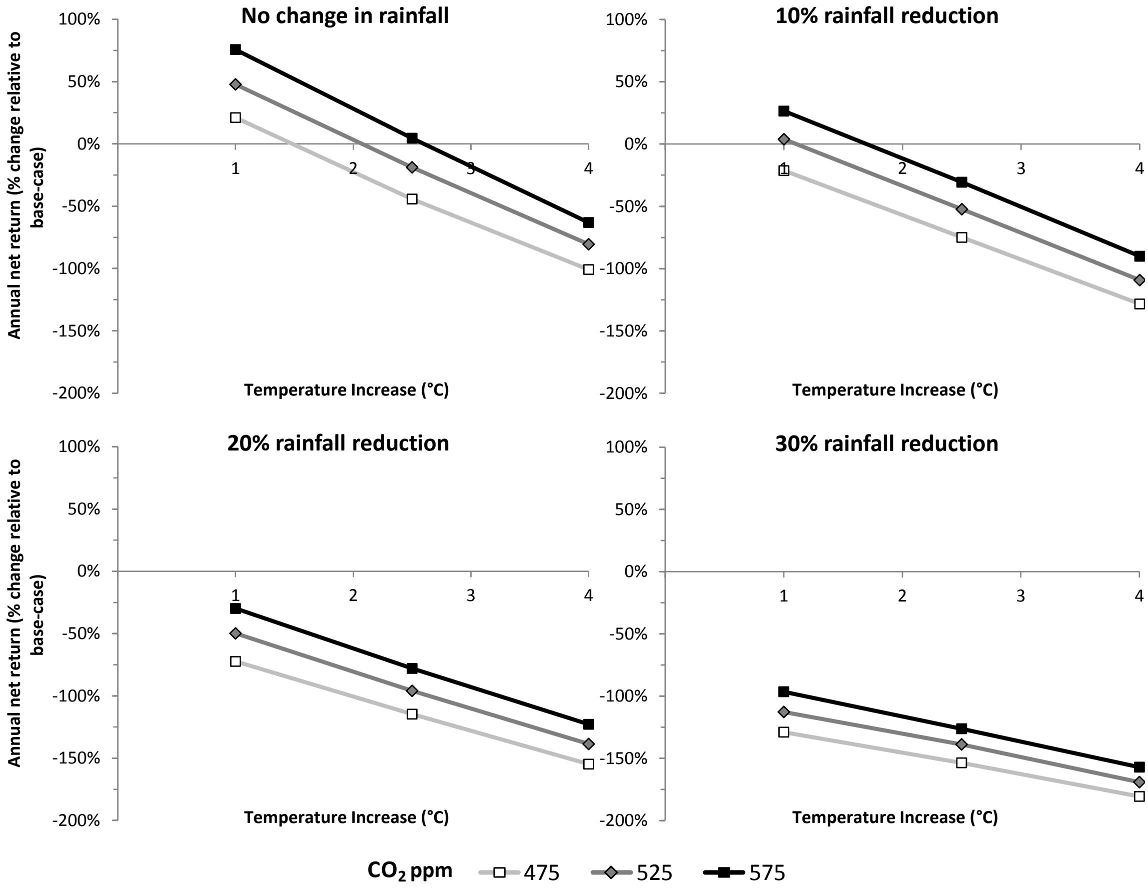

Figure 6.3. Percentage change in net return relative to base-case net return ($208,000) for the 36 climate scenarios for 2050.

6.5.2 Impacts on production versus profit

Profit margins are inherently sensitive to production levels because a certain level of production needs to occur to cover production costs. Hence the impact of climate change on profitability is proportionately larger than the impact on the amount of food and fibre produced by the farming system. To illustrate, Figure 6.4 shows changes in net production and profit for five selected climate scenarios (these scenarios were selected because they show the effect of changes ranging from small to large, as may be associated with different CO_2 levels). As the scenario becomes more severe, annual net return falls more rapidly than does production. Although severe climate change reduces the productivity of both crops and pasture, the relative reductions in the production of animal-derived outputs are disproportionately large.

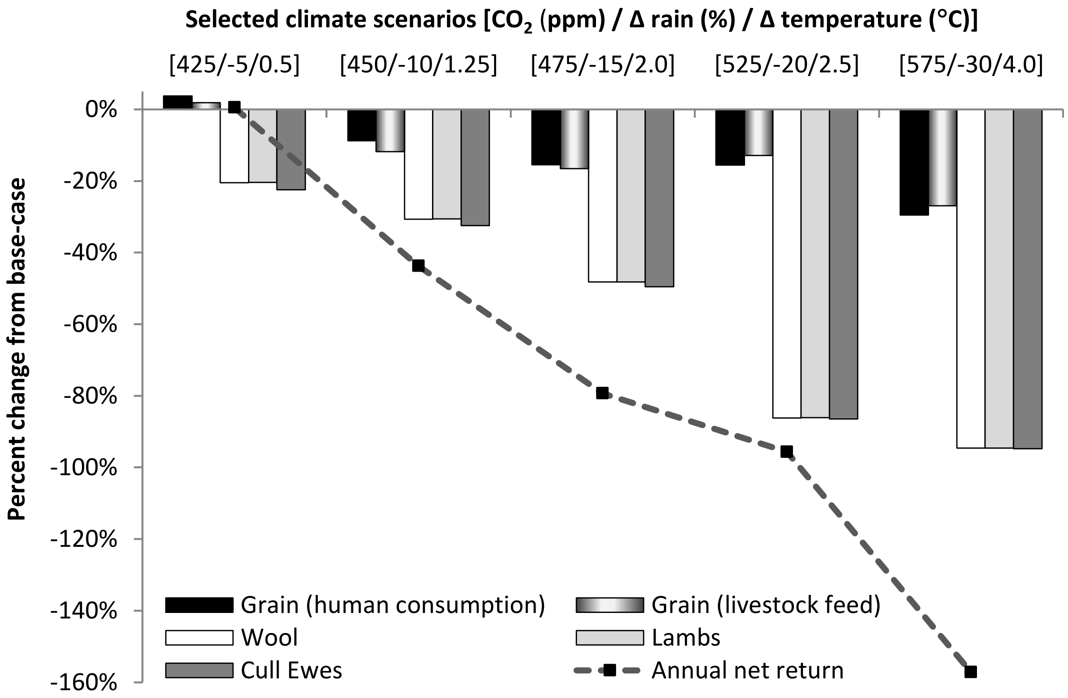

Figure 6.4. Reductions in profitability are disproportionately larger than reductions in the amount of food and fibre (tonnes of grain and wool, or in the case of livestock, number of head) produced.

6.5.3 Adaptation to climate change

For the same five climate scenarios, Table 6.2 shows the optimal set of changes or adaptations from those strategies that are presently available in the model. These strategies include altering land-uses (crop types, crop and pasture areas, rotational sequences, allocation of land uses to soil types, retiring land) and/or management (fertiliser rates, livestock numbers, and feeding strategies). The impact that each climate change scenario has on crop and pasture production (that is, the biophysical changes driving the adaptation) is difficult to show in Table 6.2 because changes in yield or growth are occurring not only due to climate change, but also due to adjustments in land use and management. However, data showing how the average yields and pasture growth would be affected by each climate scenario in the absence of confounding changes in land use and management is available in the Supplementary Material (Section 6.9.4).

Table 6.2. Optimal farm plan with average commodity prices and a base-case climate, and how it changes under selected climate scenarios.

		Base-case	Change from base-case with selected climate scenarios [CO_2 (ppm) / Δ rain (%) / Δ temperature (°C)]				
	Units	**390/0/0.0**	**425/-5/0.5**	**450/-10/1.25**	**475/-15/2.0**	**525/-20/2.5**	**575/-30/4.0**
Net return	$'000/yr	**208**	1	-91	-165	-200	-327
Crop area	ha	**2,548**	65	10	0	112	165
Pasture area	ha	**652**	-65	-230	-220	-332	-385
Retired land	ha	**0**	0	220	220	220	220
Cereal area	ha	**1,362**	28	4	0	48	261
Lupin area	ha	**545**	0	0	0	0	0
Canola area	ha	**641**	37	6	0	64	-96
N fertiliser[a]	t	**91**	5	-17	-21	-26	-50
Winter sheep	dse	**2,545**	-553	-809	-1,249	-2,198	-2,410
Extra feeding[b]	t	**132**	5	-13	-44	-108	-125
Soil Type 1	rotation	**PPPP**	PPPP	RETIRED	RETIRED	RETIRED	RETIRED
Soil Type 2	rotation	**WNWL**	WNWL	WNWL	WNWL	WNWL	WNWL
Soil Type 3	rotation	**WNWL**	WNWL	WNWL	WNWL	WNWL	WNWL
Soil Type 4	rotation	**WNWL**	WNWL	WNWL	WNWL	WNWL	WNWL
Soil Type 5	rotation	**NWBL**	NWBL	NWBL	NWBL	WNWL	WNWL
Soil Type 6	rotation	**PPPW**	42% PPPW 58% PPNWW	91% PPPW 9% PPNWW	PPPW	PPNWW	PWW
Soil Type 7	rotation	**PPNWW**	PPNWW	PPNWW	PPNWW	PPNWW	PWW
Soil Type 8	rotation	**WNWL**	WNWL	WNWL	WNWL	WNWL	WNWL

[a]Total use of synthetic nitrogen (applied to cereals & canola only) [b]Amount of supplementary grain fed to livestock
W: Wheat B: Barley N: Canola L: Lupin P: Pasture dse: dry sheep equivalents

6.5.3.1 Changes in the farming system

Across these selected scenarios, impact on farm profit ranges from a slight increase with modest changes to the climate of the study area—which may be more likely in the shorter term—to dramatic reductions in returns with more substantial changes in climate that may be more indicative of the study area in the longer term (Table 6.2). The economically-optimal set of land uses is not highly sensitive to climate change. Across the scenarios in Table 6.2 there is a slight trend toward increased cropping (usually cereals) and transition away from pasture. The main changes in land use are on Soil Types 6 and 7, which have relatively high clay contents and so are vulnerable to rainfall reductions (e.g., Ludwig and Asseng, 2006; Farre and Foster, 2010). Despite the negative returns in many scenarios, only Soil Type 1, which has low fertility, is ever retired in the optimal solution. This indicates that production is still covering variable costs on most of the soil types, albeit by a small margin in some cases. Therefore the negative net returns reflect that income from production is not sufficient to cover fixed costs and the farmer's personal expenses. Clearly this would not be sustainable in the long run.

Within a given land use, adaptation through changes in management is important—more so than changes in land use. The optimal size of the sheep flock is reduced substantially—by up to 95% in the most extreme climate scenario (Table 6.2). This leads to reductions in the amount of extra feeding required to sustain livestock in the period prior to the commencement of the growing season. For cropping land-uses, reduced nitrogen applications are the main management response to climate change, reflecting the reduced yield response to fertiliser under less favourable growing conditions. Although enhanced CO_2 can increase the nitrogen fertiliser requirements of crops (e.g., Howden et al., 2010b), in the scenarios where yields are substantially reduced (despite elevated CO_2), the optimal rate of fertiliser instead declines.

Results from a sensitivity analysis on grain and livestock prices (available in the Supplementary Material) show that the general pattern of results is not altered. At higher or lower prices, the most favourable adaptations to climate change are adjustments in livestock numbers and fertiliser rates, rather than changes in land use. Land retirement is more prevalent under low crop/high livestock prices, but under all price scenarios only soils less suited to cropping are retired from production.

6.5.3.2 Benefits of adaptation

To illustrate how much difference the adaptations described in the previous section make, Table 6.3 shows the change in annual farm profit (relative to the base-case) under different climate scenarios. The first column shows results when the model is free to adapt in any way that increases returns (the profit change is the same as in Table 6.2 as ‘full adaptation’ is the default setting used in our analysis). Moving across the columns from left to right involves progressively greater restrictions on which adaptations are allowed, and economic returns accordingly decrease. Results show that in this case study, land-use change (including land retirement) makes a relatively minor contribution to profit. On the other hand, comparison between the last two columns suggests that adaptations in management (livestock stocking rate and fertiliser rates) are much more beneficial, especially under the most extreme climate scenario. However, even under full adaptation the costs of climate change remain high.

Table 6.3. The effect of varying levels of adaptation to climate change.

Selected scenarios	Change in net return ($'000/year) compared to base-case			
CO_2 (ppm)/ Δ rain (%)/ Δ temperature (°C)	Full adaptation (changes to both management & land-uses)	Full adaptation minus the ability to retire land	Ability to adapt management but not land-uses	No adaptation (of management nor land-uses)
425/-5/0.50	1	1	1	-1
450/-10/1.25	-91	-95	-95	-107
475/-15/2.00	-165	-169	-169	-195
525/-20/2.50	-200	-203	-204	-252
575/-30/4.00	-327	-331	-344	-503

6.5.4 Maintaining profitability

In the results presented above, it is assumed that current output prices and farming technologies remain unchanged in the future. However, climate change that resulted in changed levels of agricultural production around the world would inevitably lead to altered prices. Also, ongoing agricultural research has the potential to increase production levels under any given climate scenario. Because these potential future changes are highly uncertain, we take a break-even approach, asking the question: what percentage change in either output prices or production levels would be required to return the farm to base-case profitability? For the purpose of this analysis, all output prices or production levels are assumed to change by the same percentage, with the model allowed to select profit-maximising management in response. Table 6.4 shows that with the exception of the most extreme climate scenario, the price or production increases required to maintain profitability are less than 18%, and of a magnitude which could plausibly occur as a result of market adjustments or successful research.

Table 6.4. Changes in either output prices or production levels required for the farm to maintain the same annual net return as the base-case.

Selected scenarios CO_2 (ppm)/Δ rain (%)/Δ temperature (°C)	Change required in all output prices or output levels to restore base-case profitability[a]
425/-5/0.50	-0.1%[b]
450/-10/1.25	7.1%
475/-15/2.00	13.9%
525/-20/2.50	17.3%
575/-30/4.00	35.7%

[a]Assuming no changes in input costs [b]Negative result because net returns increase in this climate scenario

6.6 Discussion

Given the high level of uncertainty about the details of future climate change, the plausible range of financial outcomes for farmers in the case-study area is very wide. In both the medium term (2030) and the longer term (2050), financial outcomes from the modelled scenarios range from moderately positive to highly negative. Results suggest the more extreme climate scenarios would likely see sizeable reductions in the economic activity generated by agriculture in the study area. Though adaptation with existing strategies was beneficial in these scenarios, the impacts of climate change remained substantial.

There are, however, grounds to hope that, at least some of the negative financial outcomes indicated for the more adverse climate scenarios could be offset by conceivable changes in price or technology. In relation to prices, population growth and increases in wealth are expected, particularly in developing countries, contributing to increased demand for food (e.g., Spence, 2011). On the supply side, depending on the spatial pattern and severity of climate change globally, there may be reductions or increases in supply. Results in Table 6.4 suggest that a modest rise in agricultural prices resulting from these factors would offset much or all of the impact of climate change (assuming the costs of inputs did not change).

Further, crop producers in the region have a strong record of innovation and adoption of superior technologies as they become available (Asseng and Pannell, 2013). Consequently, since the mid-1980s, average yields have approximately doubled, from around one to two tonnes per hectare (Turner, 2011). Table 6.4 indicates that much smaller percentage yield increases than that would be sufficient to offset the adverse effects of climate change as modelled here. Also, as superior adaptations become available, we may see more extensive changes in land use than indicated in Table 6.2, as farmers adapt to take advantage of new opportunities.

Of course, price or productivity increases could also enhance profitability without climate change. Therefore whilst price or productivity increases may offset or counterbalance the effects of climate change, they will only reduce the actual true cost of climate change to the extent that they would not have occurred without climate change (Lobell, 2014). It must also be noted that whilst a climate-induced rise in agricultural prices could benefit producers, it will be to the detriment of consumers.

In this study we defined a range of future climate scenarios to explore the consequences of uncertainty about the extent of change in rainfall and temperature. However, uncertainty about the resulting production levels of crops and pastures is even greater, because of uncertainty about the timing of changes within a year. We assumed temperatures would increase by the same amount for every day of the year, and we changed all historical rainfall observations by the same percentage across the year. In the study area, crop yields are much more responsive to rainfall in May or August than in June or July (Stephens and Lyons, 1998; Ludwig et al., 2009). Hence the rainfall reductions already experienced in the study region since the 1970s have had negligible impacts on yields because they have been concentrated in June and July (Ludwig et al., 2009; Asseng and Pannell, 2013). Assuming uniform changes across an entire growing season has therefore been criticised as likely to overstate the impacts of climate change (Ludwig et al., 2009). On the other hand, it is possible that future rainfall reductions might occur disproportionately in the most sensitive periods.

The results of this study can be compared to several other modelling studies that have been conducted for this same region. Moderate temperature increases had a more adverse effect in our study than in some previous studies (e.g., Asseng et al., 2004; Ludwig and Asseng, 2006). In Figure 6.2 and Figure 6.3, CO_2 increases only improved farm profitability if changes in temperature (and/or rainfall) were minor. In contrast, Ludwig and Asseng (2006) who, like us, also assumed any changes in climate would be distributed uniformly across the entire year, found that the positive effect of elevated CO_2 would generally compensate for the negative effects of increased temperatures. Although both studies used APSIM, the versions of the model varied. Specifically, unlike Ludwig and Asseng (2006) we did not represent the possibility of reduced waterlogging following climate change. However, such benefits—which are difficult to predict—are more applicable to areas with higher rainfall than the study area (Stephens and Lyons, 1998).

On the other hand, there are other APSIM-based analyses of the Western Australian Wheatbelt that are more consistent with our results: Farre and Foster (2010) found that increased CO_2 often failed to adequately compensate for reductions in rainfall and increases in temperature, and Crimp et al. (2012) also found negligible benefits from increased temperature. It is worth noting that ambiguity about the response of crops to high temperatures (potentially in interaction with CO_2) is a leading source of uncertainty when modelling climate change impacts (White et al., 2011; Asseng et al., 2013; Boote et al., 2013; Yin, 2013).

Our economic, farming-systems approach could also have contributed to our predictions of potentially more severe impacts than other analyses for several reasons. Firstly, other studies for the study region have tended to consider impacts on single enterprises in isolation, ignoring interactions between enterprises. These interactions can affect the viability of the farm-business too: changes in crop growth will also alter the amount of crop residue available post-harvest for livestock fodder, and reductions in the growth of legume crops and pastures will reduce the amount of nitrogen they provide for subsequent non-legume crops. By using the MIDAS model we captured such farming-system changes. Secondly, wheat production—the sole enterprise that other studies have typically considered—is potentially less sensitive to climate change than other cropping enterprises (Anwar et al., 2015). Lastly, previous analyses tend to be biophysical, whereas profit is disproportionately sensitive to yield changes.

There is one comparable economic analysis of climate change for the study region (John et al., 2005). They employed the whole-farm economic optimisation model MUDAS. MUDAS differs from MIDAS primarily in representing a probability distribution of season types, rather than a single weather-year with average conditions. They found climate change could potentially reduce farm profit by more than 50%. Whilst severe, this reduction is less than we found for many scenarios in Figure 6.2 and Figure 6.3. However, John et al. (2005) used less-sophisticated biophysical models to simulate the effects of climate change on plant growth. Also, the farming-system portrayed in their version of MUDAS is somewhat dated compared to present-day conditions (Kingwell and Payne, 2015). For instance, they did not capture recent advances in cropping agronomy (e.g., the breakcrop canola was omitted from their model) and machinery technology. Likewise, labour cost (and availability) has become an increasingly challenging issue in modern times, particularly for animal production (Doole et al., 2009; Kingwell, 2011).

This study did not concentrate on exhaustively representing all possible adaptation options and further work to parameterise a greater range of adaptation options would lead to improved results. Nonetheless, Table 6.3 shows that adaptation with existing strategies (relatively simple management and land-use changes) can moderate adverse impacts. These strategies are true ‘adaptations’ in the strict sense of the term (Lobell, 2014) because, whilst they are impact-reducing, they offer no benefit with a base-case climate. Had we not allowed for these adaptations we would have overstated the impacts of climate change by 15 –35%. However, many studies fail to allow for any form of adaptation when projecting climate change impacts (White et al., 2011). In some cases this may be because those studies rely on simulation models, for which each adaptation option must be manually specified and solved. Conveniently, optimisation models such as MIDAS adapt endogenously. In reality, farmers may not fully identify the optimal adaptations, or may delay their adaptive responses, in which case the losses due to climate change would be increased relative to an optimal set of adaptations. Further, like any whole-farm model, MIDAS is not a perfect representation of any particular farm, so the results should be treated as indicative rather than precise.

For two reasons, our results suggest that there is not a clear case for strong pre-emptive adaptation. Firstly, there is a wide range of possible outcomes, given the diverse climate scenarios modelled. Secondly, there are relatively small benefits from adaptation when changes in climate are less substantial (i.e., more representative of changes likely in the near term). Therefore, farmers may be wise to wait and see how climate change unfolds before committing to any particular adaptation. In the meantime, research focused on improving the ability of farmers to adapt in the future and on developing resilient adaptation strategies suitable for a wide range of climatic situations may be advisable (Howden et al., 2007; Hayman et al., 2012; Asseng and Pannell, 2013).

Land retirement was included in the model as a simple strategy for loss-minimisation. In reality, more-nuanced responses may occur. For instance, rather than being fully-retired, land could be planted to hardy perennial shrubs and grazed occasionally on a strategic basis (Monjardino et al., 2010). However, given the relatively low levels of land retirement observed in the results, and that whilst generating more returns than land retirement, more-nuanced approaches would also incur more costs, there is little to suggest that these more-nuanced approaches would substantially improve farm returns.

In none of the climate scenarios we examined in detail did the optimal adaptation strategy involve an increase in the area of pastoral enterprises. In many cases, it fell. This is consistent with empirical evidence that a greater dependence on crop production has been a successful strategy adopted by many farms during the number of challenging years experienced across the study region this century (Lawes and Kingwell, 2012; Kingwell et al., 2014; Kingwell and Payne, 2015; Kingwell et al., 2016). For instance, a longitudinal study of the West Australian Wheatbelt found recent periods of consecutive drought had prompted farmers to move out of livestock production into additional grain production, as cropping was generally more profitable strategy in these conditions (Kingwell and Xayavong, 2016). Nonetheless, a recent analysis of the most profitable farming systems in the lower rainfall zone directly east of our study area revealed 80% still included a livestock component, even though livestock generated only a small proportion of farm income (Kirk, 2014).

Moore and Ghahramani (2013) attributed large/disproportionate reductions in stocking rates with climate change to the need to guard against soil erosion. In the study region not all of the pasture biomass grown can be grazed; some must remain unconsumed as groundcover, protecting against erosion. Consequently, any reduction in pasture production results in a relatively larger percentage reduction in the amount of grazable biomass, and therefore, a disproportionally large reduction in livestock profitability. As MIDAS contains constraints for minimum levels of soil cover this is probably the explanation for the reduction in pasture area under adverse climate change. On the other hand, our results also showed that if pasture area was not reduced in response to climate change, profit only decreased slightly. Given that some farmers in the region perceive that livestock production is less risky than cropping, they may consider that the trade-off between risk and return favours retaining livestock in the system, although probably at a reduced stocking rate.

This analysis of climate adaptation is unusual in its integration of simulated results for several crops and pastures within an optimisation framework. There were, however, some limitations encountered when doing this. Crop yields are susceptible to the occurrence of relatively short periods of frost during anthesis or to desiccating events during grain-fill (Teixeira et al., 2013; Barlow et al., 2015). Although the frequency of hot days during grain-fill has increased in the study region (Asseng and Pannell, 2013), the ability of many crop models to capture the impacts of temperature extremes (both spring frost and heat shocks) is currently limited (Barlow et al., 2015). Also, APSIM lacked the capacity to represent the impact of elevated CO_2 on canola and lupin yield. Similar problems with crop models not being as developed for intermediate crops compared to principal crops have been encountered by others (e.g., Kollas et al., 2015); our response was to assumed equivalent-percentage responses to wheat for these crops, as outlined in the Methods and Supplementary Material. Furthermore, when simulating pasture growth, the seed dormancy/germination characteristics assumed in GrassGro reflected those typically exhibited by current pasture populations, and were especially sensitive to rainfall reductions. In reality, evolution (and/or breeding) may result in pasture systems with germination characteristics more suited to future conditions. Rather than relying solely on GrassGro and APSIM, a superior approach would be to utilise the combined predictions of an ensemble of biophysical simulation models (e.g., Asseng et al., 2015). However, the limited range of models—locally-calibrated and capable of simulating the more intermediary crops and annual, self-regenerating pastures—precluded this.

6.7 Conclusions

Our estimation of climate-change impacts at the system/whole-farm level is unlike most analyses that instead focus on a single crop or enterprise, and thereby ignore the interactions between the various enterprises that can have a large impact on the performance and make-up of a farming system. Unlike some studies, we also allowed for adaptation with existing management strategies when projecting climate impacts, showing that failing to allow for this adaptation would exaggerate estimates of the financial cost of climate change by 15 –35%. Of the existing adaptation strategies we represented in our analysis, changes in cropping inputs and livestock stocking rate were predominate, with land–use change playing a more minor role.

Across the climate change scenarios considered for the study region, the uncertainty about profit impacts is high, ranging from moderately positive to highly negative. However, the potential for loss appears much greater than the potential for gain. Although increasing atmospheric CO_2 concentration has a positive impact, under most scenarios it is not nearly enough to offset rainfall and/or temperature changes. Further, an increase in temperature or a decrease in rainfall by itself can still have severe adverse impacts without the other.

Small changes in production caused much larger changes in profitability. Amongst other things, this means that in all but the most extremely adverse climate scenarios, plausible increases in productivity or prices would be sufficient to restore profitability to pre-climate-change levels. However, if these price and/or productivity increases would have occurred in the absence of climate change then, compared to what otherwise would have happened, the cost of climate change may still be high.

6.8 Acknowledgements

We are grateful to John Finlayson for his contribution at the early stages of this analysis, particularly model development.

6.9 Supplementary Material

6.9.1 Soil types

The 3200 hectare farming system comprised the soil types outlined in Table 6.5.

Table 6.5. Description and areas of the eight soil types in the MIDAS model.

Soil Type	Description	Area of farming system (ha)
S1	Deep, pale poor sands	220
S2	Deep, yellow average sand-plain	340
S3	Good quality, yellow loamy sand	560
S4	Shallow sandy loam over clay (duplex)	340
S5	Medium to heavy rocky red/brown or grey loamy sand/sandy loam	320
S6	Red/brown sandy loam over clay; Red/grey clay heavy valley floors	320
S7	Deep/shallow sandy-surfaced valley floor	480
S8	Loamy sand with clay at depth (deep duplex)	620

6.9.2 Biophysical modelling

6.9.2.1 Parameterisation and incorporation of predictions into MIDAS

The eight soils in MIDAS were parameterised in the biophysical models. To fine-tune this parameterisation, predictions of crop and pasture production were derived using meteorological data from 1957 –2010 for Cunderdin and assuming a constant atmospheric CO_2 concentration of 390 ppm to represent ‘current’/’base-case’ climatic conditions. This process was repeated iteratively until the relative differences in crop yield and pasture productivity between soil types predicted by APSIM or GrassGro were aligned with the relative differences between the soil types in MIDAS (the predictions of the biophysical models for the 53 different ‘weather years’ (i.e., 1957 – 2010) were averaged to produce a mean estimate compatible with the ‘average weather year’ assumed in MIDAS).

However, the yields and pasture growth predicted by the simulation models for a given soil still differed in absolute terms to those specified in MIDAS (because MIDAS allows for rotational effects, weeds, pests and diseases). Therefore, the impact of the climate scenarios on the yield and growth of crops and pastures in MIDAS were based on scaling the base MIDAS yields by the relative difference between the base-case scenario and the given climate scenario predicted by the biophysical simulation models (meaning the rotational effects of nitrogen, weeds and disease had the same relative values between climate scenarios). By repeating this process for all land uses, soil types and climate scenarios, we were able to emulate in MIDAS, the relative change in pasture productivity or crop yield suggested by the biophysical models for each land-use, on each soil type, with each climate scenario.

6.9.2.2 Crop simulations in APSIM

For the crops requiring nitrogen fertiliser (i.e., non-legume crops), the APSIM simulations were conducted with applications of 40, 90 & 140 kgN/ha, with the highest yield obtainable with any of these fertiliser rates used when calculating the relative differences between the climate change and base-case simulations; the idea being to estimate the effect of climate change on yield potential, independent of nutritional constraints. The capability for elevated CO_2 to increase a crop's nitrogen requirement was then accounted for with MIDAS's endogenous nitrogen response curves that allow for the interrelationship between a crop's nutrition, yield potential, and grain quality.

The interactive effects of rainfall-temperature-CO_2 changes on barley and oats have not been studied as extensively as wheat. Consequently, the relative changes in the wheat yield predicted for by APSIM were also used for these two cereals.

The effect of CO_2 is also not well studied for the two main non-cereals grown in the study region—canola and lupin—and again APSIM currently lacks calibration for the effects of elevated CO_2 on these two crops. For canola, this problem was overcome by adding the code for the response of wheat to CO_2 to the existing APSIM canola module, thereby allowing APSIM to predict the interactive response of canola to changes in temperature-rainfall-CO_2, albeit with the response to CO_2 based on wheat. When this same approach was trialled with APSIM's lupin module, response to elevated temperature appeared inconsistent. Therefore instead, the relative change in wheat yield for a given climate scenario was also assumed for lupin. Whilst this was not ideal as it meant wheat and lupin shared the same relative response to climate change, it did nonetheless allow the interactive/rotational relationships between wheat and lupin under climate change to be explored in MIDAS.

6.9.2.3 Pasture simulations in GrassGro

MIDAS divides the growth of annual pasture over a growing season into five different temporal phases or stages, each with their own growth rate. The GrassGro predictions of pasture production were also analysed down to the level of these five phases of the growing season. Therefore not only was the relative change in pasture production predicted by GrassGro emulated in MIDAS for each soil type with each climate scenario, but furthermore also dynamically, for these five different sub-divisions of the growing season.

Whilst it is possible the effect of climate change on pasture production could also differ with different stocking rates, preliminary simulations conducted in GrassGro suggested the relative effect of a given climate change scenario remained reasonably consistent across a wide range of stocking rates. Others have also found stocking rate to have minimal influence on the relative effect of climate change, particularly for more severe scenarios (A. Moore pers. comm.). Nonetheless, the average between simulations grazed at two, four and eight wethers/ha was used when calculating climate scalars, with MIDAS then endogenously selecting the optimal stocking rate, given the changes in pasture productivity.

The pasture modelled in GrassGro was mixed-sward comprised of *Trifolium subterraneum* cv Dalkeith, *Medicago truncatula* cv Paraggio, *Lolium rigidum* (annual ryegrass) cv. Wimmera and *Arctotheca calendula* (capeweed). Climate-change-induced changes in sward composition or phenological development were only considered to the extent that they affected pasture production but not pasture quality (we also did not consider possible changes in feed quality as a result of earlier (or later) senescence with climate change). Although instances of improved pasture quality due to increased legume dominance have been predicted for southern Australia, the impact of such changes in feed quality were insignificant compared to changes in productivity (Moore and Ghahramani, 2013). Furthermore, no systematic increases in legume composition were observable in the GrassGro simulations for the study area. The evolution of annual pasture populations in response to changed climate (for example, altered seed dormancy) could have implications for both pasture productivity and quality but cannot be simulated in GrassGro and was accordingly also not considered in this analysis.

6.9.3 Comparison of MIDAS with actual farm performance

Table 6.6 shows how key results from the base-case MIDAS solution compare with empirical survey data for the central Wheatbelt study area. Overall, MIDAS's predictions accord well with the empirical data, although there are differences in some cases. Specifically:

- Annual profit in MIDAS is 84% of average farm profit reported by Planfarm for the study area between 2011 –2015, which we judge to be a reasonable alignment given the significant year-to-year variation.
- Farm area in MIDAS tends to be slightly smaller than farm surveys report (a small number of very large farms skew the average size reported by surveys upwards).
- Although the total areas of crop and pasture are very similar between MIDAS and the empirical data, for particular crops (cereals, lupins and canola) areas vary by 10 to 20 percentage points between MIDAS and the empirical data. This variation makes a little difference to predicted profit. Constraining MIDAS to the empirical areas for each crop reduces profits only very slightly (Robertson et al., 2010).
- MIDAS crop yields represent the empirical values very accurately.
- The stocking rate for sheep in MIDAS is 87% of the Planfarm estimate.

Table 6.6. Key parameters of the profit-maximising mixed farming system with a base-case climate predicted by MIDAS as compared to the average results of empirical surveys of the central Wheatbelt study area.

		Data source [year/s data collected]		
Parameter	*MIDAS*	Robertson et al. (2010) [2004-06]	Planfarm[a] [2011-15]	Lacoste et al.[b] [2014]
Average net return ($'000/year)	*$316*[c]		$377	
Average farm size (arable ha)	*3,200*	3,039	3,641	4,500
Percentage of farm occupied by:				
Cropping	*80%*		75%	85%
Cereals	*43%*		63%	65%
Lupin	*17%*		5%	8%
Canola	*20%*		13%	12%
Pasture	*20%*		25%	15%
Average yield (t/ha):				
Cereals	*2.1*	2.0	2.2	2.0
Lupin	*1.3*	1.2	1.2	
Canola	*1.0*	1.0	1.0	1.0
Stocking rate (dse[d] per winter-grazed ha)	*3.9*		4.5	

[a](Planfarm, 2012; 2013; 2014; 2015; 2016). Values shown are averages over the five years.
[b]Lacoste et al. (2015) plus unpublished data shared with authors.
[c]To facilitate comparison with empirical data, this figure does not include depreciation (fixed & variable depreciation has been deducted from the net returns we present in the main paper).
[d]dse: dry sheep equivalents

6.9.4 Biophysical impacts of selected climate scenarios

Table 6.7 shows the impact of five selected climate scenarios on yields and annual pasture growth had no adaptation occurred, and instead the optimal farming system with a base-case climate had been maintained for the climate change scenarios. The purpose of this is to show the biophysical impact of climate change as best as possible without confounding changes in management or land use[2]. Values for a given land use in Table 6.7 are based on the average value for all rotations with that land use, over all soil types and for the entire growing season. Therefore the differences in impacts between different crops may reflect differences in the susceptibility of the soil type on which that crop is predominately grown, rather than susceptibility of that crop type *per se* to climate change. Also, in the case of pasture, changes in growth were not always uniformly-distributed throughout the growing season.

Compared to crop yields, the annual pasture growth is relatively low in Table 6.7. There are several reasons for this. Compared to crops, pasture tends to be grown on the less productive soil types (like Soil Type 1), and on other soil types, pasture is often grown in rotations with long crop phases. Germination/initial production is lower in the first year after a long crop phases. Lastly, unlike fertilised crops, the pasture is fixing its own nitrogen.

Table 6.7. Average yields or annual growth in t/ha (percentage change from base-case in parenthesis) for five selected climate scenarios with no adaptation.

	Base-case	**Selected climate scenarios: [CO_2 (ppm) / Δ rain (%) / Δ temperature (°C)]**				
Land use	**390/0/0.0**	**425/-5/0.5**	**450/-10/1.25**	**475/-15/2.0**	**525/-20/2.5**	**575/-30/4.0**
Cereal	**2.1**	2.2 (1%)	2.0 (-8%)	1.8 (-15%)	1.8 (-17%)	1.4 (-34%)
Lupin	**1.3**	1.3 (1%)	1.2 (-9%)	1.1 (-16%)	1.0 (-19%)	0.8 (-34%)
Canola	**1.0**	1.0 (-2%)	0.9 (-5%)	0.9 (-10%)	0.8 (-15%)	0.7 (-30%)
Pasture	**2.1**	2.1 (-1%)	2.0 (-6%)	1.9 (-9%)	1.7 (-18%)	1.3 (-37%)

6.9.5 Prices used in analysis

Fertiliser, grain, wool and livestock prices used in the analysis were based on the average real prices from 2009 –2013. Values for the latter three are outlined in Table 6.8. Although only results derived with these average commodity prices are presented in the main text, we did also conduct some sensitivity analysis of two alternative price

[2] The quantity of supplementary feeding was allowed to change because it would have been infeasible for the model to maintain the same head of livestock without changing the amount of supplementary feeding in some scenarios

scenarios: (**i**) high grain and low livestock prices (+20% and −20% of average prices respectively) (**ii**) low grain and high livestock prices (−20% and +20% of average prices respectively) (Table 6.8).

Table 6.9 and Table 6.10 show the profitability and characteristics of the optimal farming system with these alternative price scenarios and a base-case climate, and how they change under selected climate scenarios.

Table 6.8. Grain, wool and livestock prices in the three price scenarios (expressed as 2013 prices).

		Standard Price:	Other prices used in sensitivity analysis:	
	Units	Average commodity prices	High grain (+20%) & low livestock prices (-20%)	Low grain (-20%) & high livestock prices (+20%)
Grain (FOB) prices				
Wheat	$/t	300	360	240
Barley	$/t	295	354	236
Oats	$/t	235	282	188
Lupin	$/t	305	366	244
Canola	$/t	540	648	443
Wool and sheep prices				
Wool	c/kg	1060	848	1272
Shippers	$/head	63	50	76
Prime lamb	$/kg	4.0	3.2	4.8
Cast for age ewes	$/head	54	43	65

Table 6.9. Optimal farm plan with high (+20%) crop and low (−20%) livestock prices and a base-case climate, and how it changes with selected climate scenarios.

		Base-case	**Change from base-case with selected climate scenarios [CO_2 (ppm) / Δ rain (%) / Δ temperature (°C)]**				
	Units	**390/0/0.0**	**425/-5/0.5**	**450/-10/1.25**	**475/-15/2.0**	**525/-20/2.5**	**575/-30/4.0**
Net return	$'000/yr	**473**	8	-111	-199	-237	-415
Crop area	ha	**2,880**	32	0	0	128	-199
Pasture area	ha	**320**	-32	0	0	-128	-21
Retired land	ha	**0**	0	0	0	0	220
Cereal area	ha	**1,557**	128	0	-37	155	-61
Lupin area	ha	**618**	0	0	37	37	-73
Canola area	ha	**705**	-96	0	0	-64	-64
N fertiliser[a]	t	**106**	3	-13	-29	-34	-62
Winter sheep	dse	**619**	-172	-182	-340	-509	-484
Extra feeding[b]	t	**18**	-5	-5	-10	-15	-13
Soil Type 1	rotation	**WWL**	WWL	WWL	WL	WL	RETIRED
Soil Type 2	rotation	**WNWL**	WNWL	WNWL	WNWL	WNWL	WNWL
Soil Type 3	rotation	**WNWL**	WNWL	WNWL	WNWL	WNWL	WNWL
Soil Type 4	rotation	**WNWL**	WNWL	WNWL	WNWL	WNWL	WNWL
Soil Type 5	rotation	**NWBL**	NWBL	NWBL	NWBL	NWBL	WNWL
Soil Type 6	rotation	**PPNWW**	PPNWW	PPNWW	PPNWW	WWWW	PWW
Soil Type 7	rotation	**PPNWW**	PWW	PPNWW	PPNWW	PPNWW	PPNWW
Soil Type 8	rotation	**WNWL**	WNWL	WNWL	WNWL	WNWL	WNWL

[a]Total use of synthetic nitrogen (applied to cereals & canola only) [b]Amount of supplementary grain fed to livestock
W: Wheat B: Barley N: Canola L: Lupin P: Pasture dse: dry sheep equivalents

Table 6.10. Optimal farm plan with low (−20%) crop and high (+20%) livestock prices and a base-case climate, and how it changes with selected climate scenarios.

		Base-case	**Change from base-case with selected climate scenarios [CO_2 (ppm) / Δ rain (%) / Δ temperature (°C)]**				
	Units	**390/0/0.0**	**425/-5/0.5**	**450/-10/1.25**	**475/-15/2.0**	**525/-20/2.5**	**575/-30/4.0**
Net return	$'000/yr	**151**	-12	-120	-221	-262	-414
Crop area	ha	**1,538**	-18	60	276	322	-18
Pasture area	ha	**1,662**	18	-280	-496	-542	-1,342
Retired land	ha	**0**	0	220	220	220	1,360
Cereal area	ha	**769**	-9	30	138	161	-9
Lupin area	ha	**385**	-5	100	154	165	-5
Canola area	ha	**385**	-5	-70	-16	-5	-5
N fertiliser[a]	t	**54**	0	-14	-17	-19	-30
Winter sheep	dse	**12,152**	-209	-2,152	-6,346	-6,936	-11,325
Extra feeding[b]	t	**963**	19	-111	-520	-469	-862
Soil Type 1	rotation	**PPPP**	PPPP	RETIRED	RETIRED	RETIRED	RETIRED
Soil Type 2	rotation	**95% PPPP 5% WNWL**	PPPP	WL	WL	WL	RETIRED
Soil Type 3	rotation	**WNWL**	WNWL	53% WNWL 47% PPPP	92% WNWL 8% PPPP	WNWL	WNWL
Soil Type 4	rotation	**WNWL**	WNWL	WNWL	WNWL	WNWL	WNWL
Soil Type 5	rotation	**PPPP**	PPPP	PPPP	PPPP	PPPP	RETIRED
Soil Type 6	rotation	**PPPP**	PPPP	PPPP	PPPP	PPPP	PPPP
Soil Type 7	rotation	**PPPP**	PPPP	PPPP	PPPP	PPPP	RETIRED
Soil Type 8	rotation	**WNWL**	WNWL	WNWL	WNWL	WNWL	WNWL

[a]Total use of synthetic nitrogen (applied to cereals & canola only) [b]Amount of supplementary grain fed to livestock
W: Wheat B: Barley N: Canola L: Lupin P: Pasture dse: dry sheep equivalents

Chapter 7. Paper 6. Climate change reduces the abatement obtainable from sequestration in an Australian farming system

This paper will be submitted for peer review in near future; most likely to *Agricultural Economics*, as:

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The candidate's overall contribution to the published paper was approximately 75%, as certified in the Statement of Student Contribution.

Climate change reduces the abatement obtainable from sequestration in an Australian farming system

7.1 Abstract

Agricultural research on climate change generally follows two themes: (i) impact and adaptation or (ii) mitigation and emissions. Despite both being simultaneously relevant to future agricultural systems, the two are usually studied separately. By contrast, this study jointly compares the potential impacts of climate change and the effects of mitigation policy on farming systems in the central region of Western Australia's grainbelt, using the results of several biophysical models integrated into a whole-farm bioeconomic model. In particular, we focus on the potential for interactions between climate impacts and mitigation activities. Results suggest that, in the study area, farm profitability is much more sensitive to changes in climate than to a mitigation policy involving a carbon price on agricultural emissions. Climate change reduced the profitability of agricultural production and, as a result, reduced the opportunity cost of reforesting land for carbon sequestration. Despite this, the financial attractiveness of reforestation did not necessarily increase because climate change also reduced tree growth, and therefore the income from sequestration. Consequently, at least for the study area, climate change has the potential to reduce the amount of abatement obtainable from sequestration—a result potentially relevant to the debate about the desirability of sequestration as a mitigation option.

Keywords: Climate Change Impacts, Adaptation, Mitigation, Emissions, Agriculture, Interaction, Economics, Sequestration

7.2 Introduction

In studies of climate change and agriculture the topics of (i) impact and adaptation and (ii) mitigation and emissions are typically addressed in isolation. For instance, a review of 221 studies modelling the impact of climate change on agriculture found that only 2% of studies extended their projections to also consider the potential impacts of climate change on soil carbon levels, and only 3% considered the potential impacts on farm emissions (White et al., 2011). However, the reality is that both aspects are simultaneously relevant to future agriculture and may interact with each other.

It is possible that practices that mitigate emissions could also help with adaptation to climate change. For example, farmers may choose to adapt to climate change by switching from the production of agricultural crops to trees for carbon sequestration (Howden et al., 2010a). The long timeframes associated with sequestration makes the impact of climate change on sequestering practices a potentially important consideration (Goetz et al., 2013). It is also possible that changes in climate could reduce the efficacy of strategies to mitigate emissions. For example, in Australia biophysical analyses have suggested the ability to increase and/or maintain carbon already sequestered in agricultural soils may decline under projected changes to climate (Grace et al., 2006; Hoyle et al., 2013; Liu et al., 2014; Conyers et al., 2015).

Smith and Olesen (2010) suggested that there is potential for large synergies between the adaptation of agriculture to climate change and mitigation. However, they also noted that the relationships between the two require further research. In particular, they called for economic analysis of the effect of climate change impacts and adaptation strategies on the cost of policies aimed at mitigating emissions in the agriculture/land sectors. However, of the analyses that have considered the effect of climatic changes on mitigation strategies/policies, many are purely biophysical (Liu et al., 2014; Xiong et al., 2014; Hobbs et al., 2016). Whilst biophysical models are useful for predicting how biophysical rates of emissions or sequestration from a given enterprise or land uses may alter under a changed climate, climate change may also affect the financial performance of a farming enterprise (and therefore the cost of incorporating into a farming system those practices or land uses that mitigate emissions). To provide this combination of information on physical and financial impacts, bioeconomic analysis is required (e.g., Reidsma et al., 2015).

In Australia, interactions between climate change and mitigation policy could be particularly important. Given Australia's land resources, there is potential for mitigation with sequestration (e.g., Harper et al., 2007). Indeed, under its 'Direct Action' climate policy, the Australian Government intends to procure large quantities of sequestration from agricultural land with its Emissions Reduction Fund (e.g., Neales, 2014). Yet, as a dry continent, Australia also has a high level of exposure and sensitivity to climatic change (Garnaut, 2008).

Recently several bioeconomic integrated assessments have considered interactions between climate change and mitigation policy in their projections about future land uses across Australia (Bryan et al., 2014; Connor et al., 2015; Bryan et al., 2016a; Bryan et al., 2016b; Grundy et al., 2016). Although these are methodologically-complex studies, they did not use process-based models to directly predict the impact of climate change on agricultural production, nor did they account for the possibility of elevated CO_2 enhancing agricultural production. Furthermore, these analyses use a grid-based spatial approach, so do not capture how climate change and climate policy will affect the interrelationships between different enterprises and business performance at the farm-level. They also did not endogenously allow for the management of existing agricultural land-uses to be adapted in response to changes in climate, even just with currently available technology/options.

Conversely, a number of other studies (Petersen et al., 2003a; Petersen et al., 2003b; Flugge and Schilizzi, 2005; Flugge and Abadi, 2006; Kingwell, 2009; Kragt et al., 2012; Thamo et al., 2013) have considered the effect of climate change policy with detailed, farm-level analysis. However, none of these analyses simultaneously considered the impact of future climate changes in climate, and how they these changes may interact with mitigation policy, for example by affecting the viability of sequestration.

We aim to fill these gaps by conducting a farm-level bioeconomic analysis comparing the prospective impacts and interactions between climate change and mitigation policy. We use process-based models to simulate the impact of climatic changes (including elevated CO_2) on both agricultural production and reforestation for sequestration. We then incorporate these predictions into a detailed, whole-farm model that explicitly represents the interrelationships between different components of farm businesses and furthermore, utilises optimisation techniques to adapt land-uses and management practices to changes in climate and policy.

Our study area is the central area of Western Australia’s Wheatbelt. Although this is Australia’s largest grain-growing region, reforestation of farmland could provide sequestration under suitable policy regimes. However, in coming decades the climate of the region is predicted to dry and warm, potentially substantially (e.g., Delworth and Zeng, 2014). Consequently, our results suggest that for typical farming systems in this area, climate change could substantially reduce the amount of abatement provided by sequestration, and that farm profitability is much more sensitive to changes in climate than to a mitigation policy involving a carbon price on agricultural emissions.

7.3 Methodology

To quantify the impacts and interactions between climate change and mitigation policy we first used simulation models to predict the biophysical impacts of a number of climate scenarios, and then incorporated these predictions into the whole-farm bioeconomic Model of an Integrated Dry Land Agricultural System (or MIDAS—Kingwell and Pannell, 1987), allowing the financial impact of these climate scenarios to be analysed simultaneously with different scenarios for mitigation policy.

7.3.1 Study area and MIDAS model

MIDAS is a steady-state deterministic model in which farm profitability is maximised (subject to various managerial, resource and financial constraints) by selecting the optimal combination of land uses and management practices for a 'typical' or average weather-year (Morrison et al., 1986). MIDAS is a complex model. It captures interdependencies and relationships between various aspects of the farming system. These include: the benefits of rotating crop types (nitrogen fixation and disease management); the impact of cropping phases on pasture regeneration; the influence of weed populations on crop yields and; crop residues being a fodder for livestock.

The MIDAS model we use was developed for central area of the Western Australian Wheatbelt (Figure 7.1) where it has been employed extensively (e.g., Petersen et al., 2003b; Kingwell, 2009; Kragt et al., 2012; Thamo et al., 2013). Validation of the latest version of MIDAS (which we use) is presented in (Thamo et al., 2017). This area's Mediterranean climate is characterised by dry, hot summers and cool winters during which the majority of precipitation occurs (the town of Cunderdin, at the epicentre of the study area has as average rainfall of 360mm). The farming systems are rain-fed, with no irrigation. Based on the characteristics typical of contemporary farms in this area, we assume a 3200ha farm with eight different soil types (these soils are described in more detail in Thamo et al., 2017).

Land uses in MIDAS include different rotational combinations of wheat, canola, lupin, barley and oats, and legume-based annual pastures. Being a mixed-enterprise farming system, sheep are grazed on pastures in winter/spring and on crop residues in summer, producing wool and meat. In autumn, livestock require a grain-based supplementary ration. To allow for the possibility of climate change making agriculture unprofitable on some soil types, the model includes the option to retire land from production.

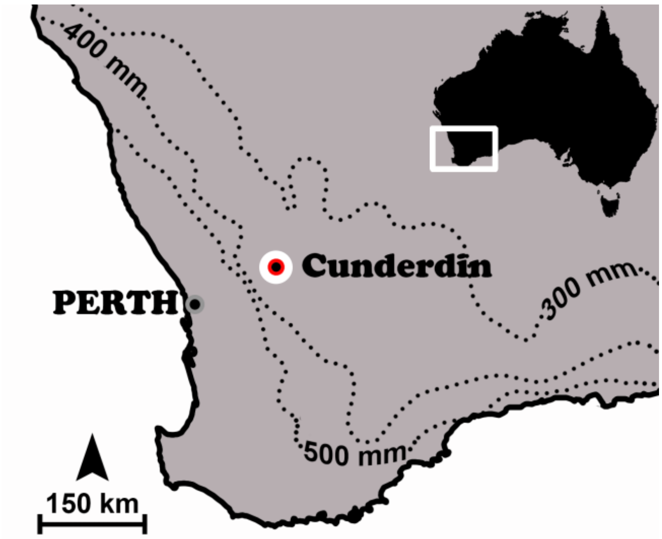

Figure 7.1. Cunderdin, in the centre of the Western Australia's Wheatbelt, represents the middle of the study area. Isohyets show the historically-observed annual rainfall.

For the analysis, grain, livestock and fertiliser prices were based on the average real prices from 2009–2013. For inputs other than fertiliser we used 2013 prices. Thus, we did not implicitly consider any long-term changes in the price of agricultural commodities (either due to structural changes in demand, technological change or the effect of climate change and/or mitigation policy). We express all monetary values in Australian dollars (on average from 2001–2015, AU$1 was equal to €0.64 and US$0.80).

7.3.2 Climate change impacts

7.3.2.1 Projections for study region

With climate change, south-western Australia is predicted to get hotter and drier, particularly in the longer term. An analysis of the predictions of more than 40 Global Climate Models that underlie the Fifth Assessment Report of the Intergovernmental Panel on Climate Change (Hope et al., 2015) indicated with strong confidence that, across all seasons, temperatures will increase by an average of 0.5–1.1°C by 2030, and by 1.2–4.0°C by 2090 compared to 1986–2005. In addition, June –November rainfall (effectively the growing season) will change by +5% to −15% by 2030, and by −5% to

−45% by 2090. These predicted changes in temperature and precipitation accord with those from other studies and with changes already observed in the study region's climate (e.g., Hughes, 2003; Hope et al., 2006; CSIRO and BoM, 2007; Bates et al., 2008; Potgieter et al., 2013; Delworth and Zeng, 2014). Indeed, the study region is one of the few instances where regional changes in rainfall have been conclusively attributed to climate change (Karoly, 2014).

7.3.2.2 Climate change scenarios

The 'base-case' climate used in this study consisted of metrological observations for Cunderdin from 1957–2010, and an assumed atmospheric CO_2 concentration of 390 ppm. Five climate-change scenarios were considered (Table 7.1). These scenarios reflect the near unanimous predictions in the literature that the study region will become hotter and drier, conditional on the future trajectory of global emissions (i.e., CO_2 levels). The scenarios were created by adjusting all observations in the base-case dataset by the amounts shown in Table 7.1. This approach to the development of climate scenarios—also used by van Ittersum et al. (2003); Ludwig and Asseng (2006); Bryan et al. (2010); Bryan et al. (2011); Paudel and Hatch (2012); Thamo et al. (2017)—changes the minimum and maximum temperatures, and the intensity but not frequency of precipitation, therefore creating climatic datasets with a different average but the same variability as historically experienced.

Table 7.1. The five climate scenarios analysed.

Scenario	**CO_2 (ppm)**	**Precipitation reduction (%)**	**Temperature increase (°C)**
Small change	425	-5	0.50
Small-Medium change	450	-10	1.25
Medium change	475	-15	2.00
Large change	525	-20	2.50
Extensive change	575	-30	4.00

7.3.2.3 Modelling biophysical impact on agricultural production

For each climate scenario, we simulated pasture growth using GrassGro® (version 3.2.6) (Moore et al., 1997) and crop yields using APSIM (Agricultural Production Systems Simulator, version 7.5) (Keating et al., 2003). These process-based models have the capacity to model the effect of climate change, including elevated CO_2, on plant growth (Moore, 2012; Holzworth et al., 2014), and have been used for this purpose in the study region (e.g., Asseng et al., 2004; Ludwig et al., 2009; Ghahramani and Moore, 2013).

The predictions of agricultural production under each climate scenario by GrassGro and APSIM were incorporated into MIDAS as follows: First, the simulation models were parameterised for each of MIDAS's eight soil types, such that the relative differences in yields and pasture growth between the different soil types predicted by the simulation models (on average, across the 53 years (1957 –2010) of the base-case climate dataset) matched those specified in MIDAS for an 'average weather year'. The simulation models were then run for each climate scenario in Table 7.1. The relative difference between the result for each climate scenario and result with the base-case climate (when averaged over 53 years) was then used to scale the growth and yield potential of pastures and crops in MIDAS. This process was repeated for all soil types and land-uses, allowing the relative change in pasture or crop growth projected by the simulation models for every climate scenario, on every soil type, to be represented in MIDAS. In the case of pasture, the growing-season was divided into five phases, with pasture growth in each climate scenario assessed separately for each phase. This allowed for the possibility of climate change impacting pasture production differently across the growing season.

The APSIM model we used had not been calibrated to model the impact of elevated CO_2 on lupin, oats and barley. Therefore our estimates of the effect of changes in precipitation-temperature-CO_2 on the yields of these crops were based on APSIM's predictions for wheat (for which the model has been extensively calibrated). We also assumed that canola would display the same response to elevated CO_2 as wheat. For more details about the process used to simulate the impact of climate change on agricultural enterprises see (Thamo et al., 2017).

7.3.3 Mitigation policy

We considered two ways in which mitigation policy could affect agriculture: (i) agriculture could provide emission offsets by sequestering carbon and; (ii) landholders could be required to buy permits or pay a tax for emissions that occur on-farm (with the cost of this permit or tax being set by the 'carbon price').

7.3.3.1 Sequestration

Tree planting is a much-discussed option for sequestering carbon on agricultural lands (e.g., Harper et al., 2007; Polglase et al., 2013; Bryan et al., 2014 etc.). Indeed, under the Australian Government's Emission Reduction Fund, landholders can voluntarily re-afforest their land and then claim and sell (at the carbon price) 'credits' for the resulting sequestration (ComLaw, 2013).

To represent this mitigation opportunity we included in the MIDAS model the option for farmers to reforest their land with mallee trees (*Eucalyptus loxophleba* subsp. *Lissophloia*) planted in block configuration. The amount of sequestration that could be claimed from these plantings was estimated with FullCAM (version 3.55) (Richards and Evans, 2004), which is the model used to estimate sequestration in the Australian Government's Emissions Reduction Fund. Because FullCAM does not readily differentiate between soil type at a paddock/farm-scale (Hobbs et al., 2016), we adjusted FullCAM's estimates to match each of MIDAS's eight soil types based on predictions by Farquharson et al. (2013), and the forestry model 3PG (Landsberg and Waring, 1997).

To represent the effects of climate change, the estimates of sequestration were scaled based on the 3PG model's prediction of the effect of each climate scenario (CO_2 fertilisation, rainfall and temperature changes) on sequestration rates for each soil type. For simplicity, we assumed re-afforestation would occur once the climate had 'changed' to that scenario, and not whilst it was in the process of changing. Carbon prices ranging from \$5 –\$100/tCO_2-e were tested for each climate scenario. We assumed that this carbon price, the opportunity cost of foregone agricultural production, and transaction costs would all remain constant through time in real terms. Values assumed for transaction costs and additional details about the modelling of sequestration are described in the Supplementary Material.

7.3.3.2 Agricultural emissions

While sequestration opportunities in agriculture have been lauded as a mitigation option, agriculture is also a significant source of emissions in some countries (~16% of emissions in Australia—Department of the Environment, 2015b). If, in time, other sectors of the economy manage to reduce their emissions, there is likely to be increasing focus on agricultural emissions. It has been suggested that in the longer-term, the best policy response would therefore be to impose a mandatory carbon price to agricultural (i.e., on-farm) emissions (Garnaut, 2011).

To represent this scenario in MIDAS, we used the emission factors employed in Australia's national greenhouse gas accounting (National Inventory, 2011)[1] to quantify emissions from the following sources:

- CO_2 from fuel combustion and urea hydrolysis;
- N_2O from fertiliser, animal wastes, biological N-fixation and crop residues, and;
- CH_4 from enteric fermentation.

Although climate change could affect the biophysical processes behind some of these emissions, we did not consider this because: (a) these complex processes are multifaceted and interact with other factors such as moisture level, stage of crop growth etc. and; (b) these changes would also have to be recognised in the actual emission factors used when pricing emissions. Consequently, climate-change-induced changes in emissions were limited to those resulting from structural adjustments to the farming system (changes in input use, land use, animal numbers, etc.).

7.4 Results

7.4.1 Climate change impacts on the farming system

With a base-case climate, the financially-optimal strategy for a typical farm in the study area involves cropping 2,548 hectares (~80% of the farm area) annually, with the remainder of the farming system under pasture (Table 7.2). Under the mildest climate scenario farm returns increase by 1% compared to base-case. This is because of the positive effect of elevated CO_2 and small increases in temperatures during the winter months. However, with the larger rainfall reductions and temperature increases assumed under other climate scenarios, this positive effect is overwhelmed[2] and returns substantially decrease. Note that the annual net returns we report are the returns after deducting variable costs, fixed overheads and non-cash expenses (like depreciation) but not the opportunity cost of the capital invested in land, livestock and machinery. Including these opportunity costs would lower returns, which would have had the effect of making relative changes in returns more pronounced.

[1] The exact formulas and assumptions used when applying these emissions factors in MIDAS are detailed in Thamo et al. (2013).

[2] Overwhelmed in the sense the overall *net* effect of the climate scenario is negative. However, without 'CO_2 fertilisation' the net effect would be even more negative (this is covered in detail in Thamo et al., 2017).

Adaptations to climate change include a large decrease in livestock numbers (Table 7.2). This is because under climate change, pastured land is less productive and the optimisation model responds by allocating less area to pasture in the farming system. Land converted out of pasture is either placed under crop or retired from production. The 220 hectares of land that is retired from production under most climate scenarios represents the most infertile, sandy soil in the farming system. Another adaptive change is to reduce applications of nitrogen fertiliser (despite increases in the area cropped). This is because crops tend to have lower yield potential under most of the climate scenarios, making it economically optimal to apply less fertiliser.

Table 7.2. Key characteristics of the typical, economically-optimal farming system in the study area and how they may change with climate change (percent change compared to base-case climate shown in parentheses).

Climate Scenario	Crop area (ha)	Pasture area (ha)	Sheep flock (DSE[a])	Fertiliser (tonnes)[b]	Retired Land (ha)	Farm annual net return ($ '000)
Base-case	*2,548*	*652*	*2,545*	*91*	-	*$208*
Small change	2,613 (3%)	587 (-10%)	1,992 (-22%)	96 (6%)	-	$209 (1%)
Sml-Med change	2,558 (0%)	422 (-35%)	1,736 (-32%)	74 (-18%)	220	$117 (-44%)
Medium change	2,548 (0%)	432 (-34%)	1,296 (-49%)	70 (-24%)	220	$43 (-79%)
Large change	2,660 (4%)	320 (-51%)	347 (-86%)	65 (-28%)	220	$9 (-96%)
Extensive change	2,713 (6%)	267 (-59%)	135 (-95%)	41 (-55%)	220	$-119 (-157%)

[a]'dry sheep equivalents' [b]Total tonnes of synthetic elemental nitrogen (applied to cereals & canola only)

Overall, on-farm emissions tend to fall as the severity of climate change increases (Figure 7.2). This is consistent with reductions in emissions observed when drier years are experienced in the study region under 'current' climatic conditions (Kingwell et al., 2016), and is primarily driven by a reduction in methane emissions from livestock (due to the decrease in sheep numbers shown in Table 7.2). Reduced emissions from fertiliser use, crop residues and nitrogen fixation play a smaller role.

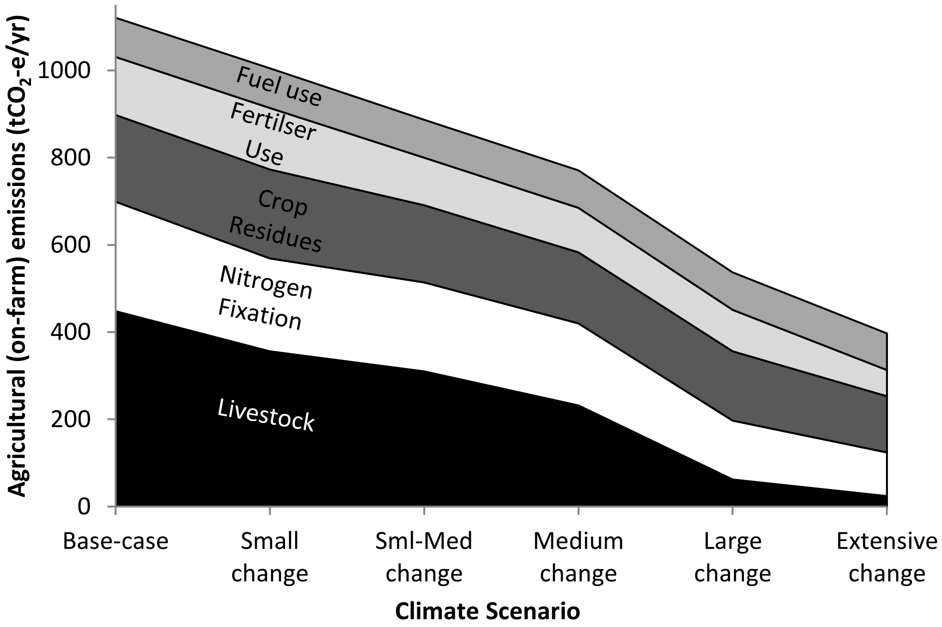

Figure 7.2. Emissions profile of the farming system under each climate scenario.

7.4.2 Mitigation policy and interactions with climate impacts

We now consider the implications of different options for mitigation policy, and how they may alter if the climate changes, starting with a policy where land can be voluntarily reforested to sequester carbon.

7.4.2.1 Sequestration

Table 7.3 shows how much of the farm would be reforested under various carbon prices and climate scenarios. For reforestation to become part of the optimal farming system with a base-case climate, more than \$40/tCO_2-e would need to be received from selling carbon credits. Whilst this may seem like a relatively high carbon price, it is broadly consistent with other Australian analyses (e.g., Flugge and Abadi, 2006; Polglase et al., 2013; Thamo et al., 2013; Bryan et al., 2014; Hatfield-Dodds et al., 2015; Grundy et al., 2016).

Values of 7%, 33% and 43% appear repeatedly in Table 7.3 because they correspond to proportions of the farm occupied by given soil types. For instance, the 7% of the farming system that requires the lowest carbon price to be reforested represents the same infertile soil which is retired from production in the absence of sequestration policy in Table 7.2. For it to be optimal to reforest more area than this 7% requires a carbon price of around \$80/tCO_2-e. Although Table 7.3 indicates 43% of the farm would be reforested at \$100/tCO_2-e, in reality this percentage is likely overestimated because with wide-scale reforestation, agricultural commodity prices would increase

Table 7.3. The optimal percentage of the farming system to reforest for sequestration under various carbon price and climate scenarios.

Carbon Price (\$/tCO_2-e)	Climate Scenario:					
	Base-case	Small change	Sml-Med change	Medium change	Large change	Extensive change
\$40	-	-	-	-	-	-
\$50	*7%*	7%	7%	7%	-	-
\$60	*7%*	7%	7%	7%	7%	-
\$70	*7%*	7%	7%	7%	7%	-
\$80	*33%*	18%	13%	17%	7%	7%
\$90	*33%*	33%	33%	33%	33%	7%
\$100	*43%*	43%	43%	43%	33%	33%

Reforesting for sequestration does not become more attractive under the climate scenarios modelled. Instead, Table 7.3 shows that at many carbon prices, it is optimal to re-vegetate less land under the five climate scenarios than with the base-case climate. Given traditional agricultural pursuits are less profitable under climate change (Table 7.2), it may seem counter-intuitive that reforesting agricultural land for sequestration does not therefore become more attractive. However, climate change also affects tree growth, reducing the amount of carbon sequestered by the reforestation of a given amount of the farming system (Figure 7.3). Lower sequestration rates mean less income from reforestation. Importantly, if, under such circumstances where revenue is reduced, there is less capacity to adapt and alter the cost structure of reforestation than agriculture, then the profitability of sequestration can fall more than agriculture, and it can be optimal to undertake less reforestation under a changed climate.

If the rate of sequestration is lower (per unit of area reforested), and it is economically-optimal to reforest less area, the combined results is that the amount of sequestration obtainable for a given carbon price will decrease (Figure 7.4). In other words, in this environment, climate change reduces the cost-effectiveness of a mitigation policy based on sequestration. This result appears to be relatively robust; a sensitivity analysis of the effect of climate change on the supply of sequestration for a given carbon price is provided in the Supplementary Material. It shows that, even if sequestration rates were 20% less sensitive to climatic change than the biophysical modelling predicts, the abatement obtainable for a given carbon price is still lower with climatic change.

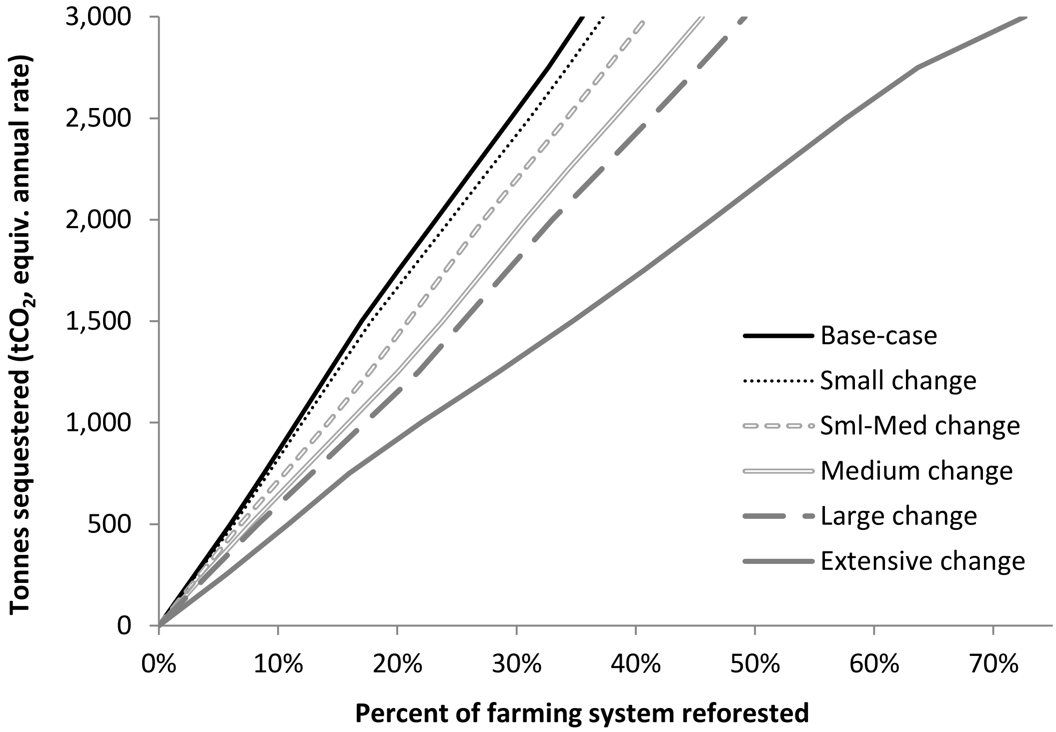

Figure 7.3. As the extent of the changes in climate increases, the amount of sequestration obtainable by reforesting a given amount of farm land declines.

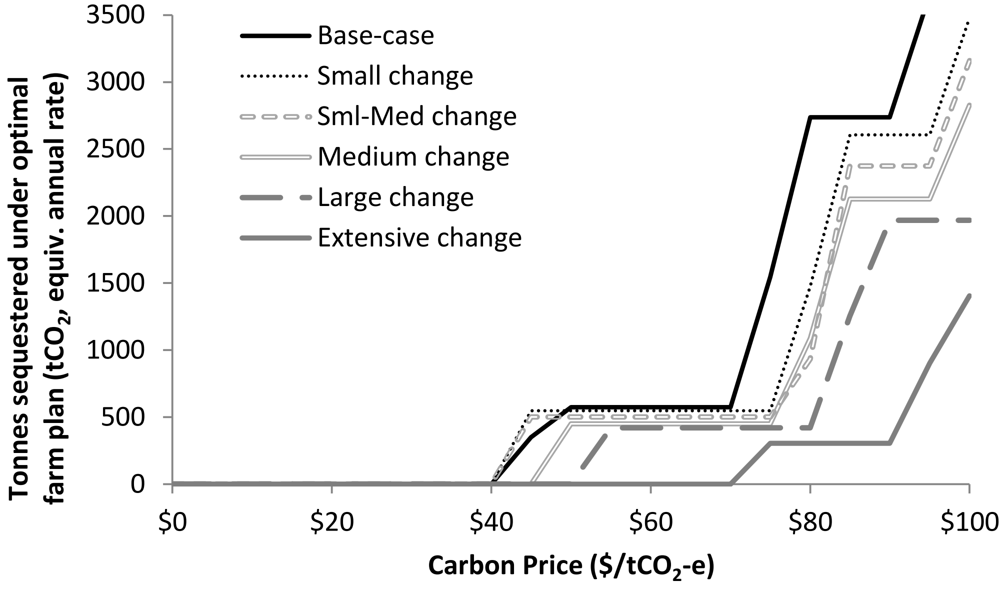

Figure 7.4. The carbon price required for the production of a given quantity of sequestration to be economically attractive tends to increase as the extent of climate change increases.

7.4.2.2 Emissions price but no sequestration

We now consider the application of a mandatory carbon price to on-farm emissions, without a sequestration policy. With the 'stick' of the emissions price, yet no 'carrot' in the form of potential sequestration-income, this would be the worst-case policy scenario in terms of impact on farm profit. It is therefore interesting to see how, compared to climate change, such a policy might impact profit. Table 7.4 shows the price on agricultural emissions (but with no sequestration policy) required to produce the same impact on farm returns as predicted under the climate scenarios (with no mitigation policy at all). It reveals that climate change has an impact equivalent to charging a high price on agricultural emissions. For instance, the ‘Small-Medium’ climate scenario (a 10% reduction in precipitation, 1.25°C increase in temperatures and 450 ppm atmospheric CO_2) has an impact on profit equivalent to a \$104/tCO_2-e carbon price on agricultural emissions with the base-case climate. These results suggest it is very likely that the negative impact of climate change on farm profit would exceed the negative impact of this emissions pricing policy.

Table 7.4. The carbon price on agricultural emissions, but with no sequestration policy, that achieves the equivalent impact on net returns with a base-case climate, as would occur under each climate scenario (without any mitigation policy).

Climate scenario	Impact on returns equivalent to base-case climate but a carbon price (\$/tCO_2-e) on agricultural emissions (and no ability to sequester) of:
Small change	-\$1.1[a]
Small-Medium change	\$104
Medium change	\$213
Large change	\$268
Extensive change	\$473

[a]Price is negative because net returns increase with this climate scenario

7.4.2.3 Both sequestration and a carbon price on agricultural emissions

In reality, if a carbon price were imposed on on-farm emissions it would likely be accompanied by a sequestration-credit scheme. Figure 7.5 shows the impact of an increasing carbon price under this policy situation, for each climate scenario. In Figure 7.5, annual returns are curvilinear. Returns initially decrease as the carbon price increases because the cost of agricultural emissions becomes more expensive, reaching a nadir with a carbon price of about \$75/tCO_2-e. With further increases in the carbon price the income from sequestration becomes greater than the cost of agricultural emissions and returns begin to increase. This in turn leads to more land being converted to trees for sequestration. The larger the area of trees, the more rapidly profit increases with an increase in carbon price. The increase in profit at high carbon prices is less pronounced under more severe climate scenarios. This is partly because on-farm emissions tend to be lower under these scenarios (Figure 7.2), which means that emission charges have less impact, and partly because climate change reduces sequestration so there is less potential to capitalise on high carbon prices with reforestation.

The relative flatness of the curves in Figure 7.5 and the distance between them indicate that, with one exception, differences in the climate scenarios have much bigger effects on farm returns than a carbon price of $0 –$100/tCO2-e.

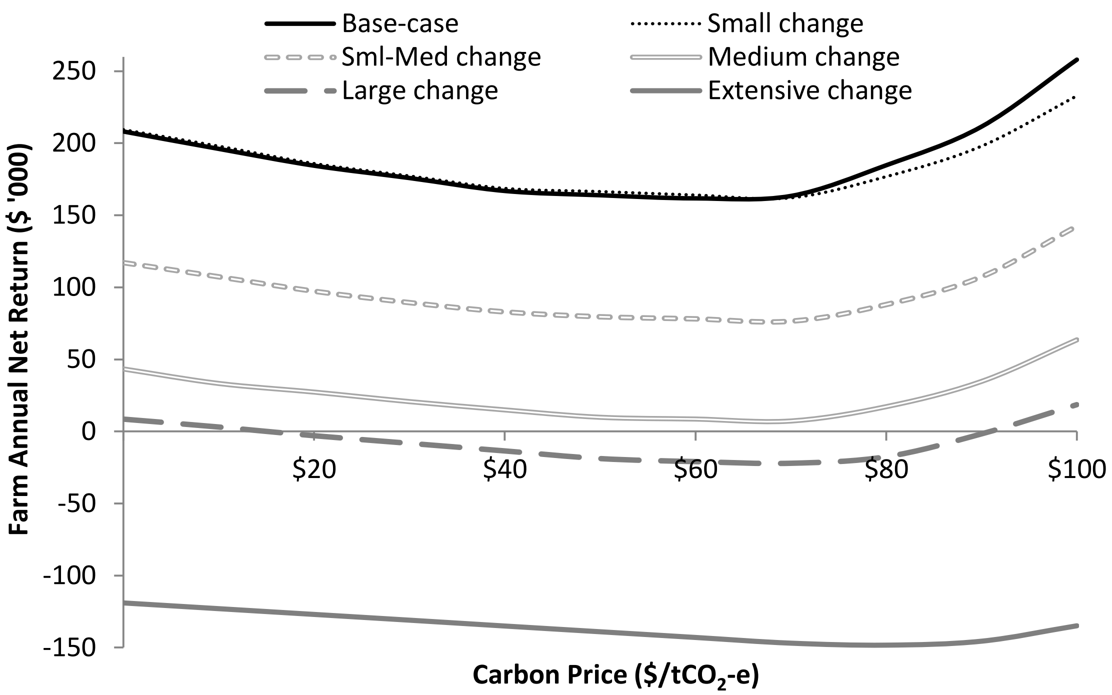

Figure 7.5. The impact of a carbon price on agricultural emissions (when there is also the option of participating in sequestration policy) on farm profitability under different climate scenarios.

Figure 7.5 and Table 7.4 are intended to provide some insight into the scale of the relative effects of climate change versus mitigation policy. They are, however, not sufficient to identify the most efficient response to climate change. That would require comparing the discounted cost of climate-related losses in the future against abatement costs incurred in the present.

7.5 Discussion

Two key findings emerge from the results: (i) in this study area, climate change has the potential to reduce financial-attractiveness of participating in a policy that aims to encourage sequestration through reforestation, and; (ii) farm profitability is likely to be more sensitive to climatic changes than to the implementation of a mitigation policy involving a carbon price on agricultural emissions. We discuss each of these in turn.

7.5.1 Impact of climate change on the cost-effectiveness of sequestration

With larger changes in climate, the quantity of sequestration obtained from the farming system is reduced. This suggests that, at least for the study region, there could be risks for policies that rely heavily on sequestration to meet mitigation targets. Under a changed climate, the abatement obtainable from reforestation declines (Figure 7.4) for both economic and biophysical reasons. Biophysically, as the climate warms and dries, the amount that would be sequestered by growing trees on a given area of land reduces (Figure 7.3). This means that with a warming and drying climate, if the same amount of area was reforested, then *ceteris paribus* sequestration would likely provide less abatement in the study region. Of course the area reforested could also change: Table 7.3 showed fewer hectares being allocated to sequestration under a changed climate. How much area is reforested does not depend (directly) on whether, biophysically, tree growth is more affected by climate change than crop and pasture growth, but rather economically, on whether the land use provide the best returns under climate change. Hence, a reduction in the area deforested, by and of itself, should not be interpreted as indicating that the growth of trees is more affected than crop and pasture growth but rather as indicating that the economic performance of reforestation is more affected.

Contributing to the economic performance of sequestration being more impacted than agriculture was that agricultural production methods were adapted under climate change (e.g., management was changed, lowering input costs). In contrast, the costs associated with reforestation—planting/establishment and ongoing monitoring and auditing—were assumed fixed in this analysis. It is possible that adaptations or technical developments could reduce these costs and/or increase productivity of reforestation in the future. In terms of climate change impacting the efficacy of mitigation strategies, such developments are, however, only relevant if they are more advantageous in the presence of climate impacts than in the absence of climate impacts. The adaptation of agricultural production that occurred in this analysis, like changes in fertiliser applications or livestock stocking rate, were not only impact-reducing, but were also not advantageous in base-case conditions. Although not captured in this analysis, another consideration about adaptability is that trees, once planted, are not easily adapted thereafter. In contrast, the annual cyclic nature of agriculture lends itself to the progressive adoption of adaptation strategies as they become available.

Our results suggest that previous studies that did not consider the impact of climate change (Petersen et al., 2003a; Petersen et al., 2003b; Flugge and Schilizzi, 2005; Flugge and Abadi, 2006; Kingwell, 2009; Thamo et al., 2013) may have overstated the mitigation potential offered by reforestation in the study area. Our results also suggest that assuming that currently-realistic sequestration rates will persist may be erroneous (Hobbs et al., 2016). Rather the pricing of sequestration contracts with landholders may need to account for the long-run impacts of climate change. Another consideration is that the upfront investment to establish trees is considerable; if climate change's impact on farm returns reduces the cash-flow then obtaining the capital to make this land-use change could become an issue for farmers (John et al., 2005).

A recent series of Australian land-use studies (Bryan et al., 2014; Connor et al., 2015; Bryan et al., 2016a; Bryan et al., 2016b; Grundy et al., 2016) did consider interactions between climate change and mitigation policy. A common thread uniting these studies is that they all employ fundamentally the same methodology, utilising equilibrium modelling to project economic growth, demand for agricultural commodities and energy, and carbon prices until 2050. Broadly-speaking, these studies predict that substantial areas of agricultural land across Australia will be reforested for sequestration in the future, despite the impacts of climate change. However, their integrated assessments do not separate out the effects just of climate change, independent of changes in other economic factors predicted by the equilibrium modelling, meaning direct comparison of their findings with ours is difficult. For instance, their scenarios with highest carbon price (which made sequestration attractive) were also the scenarios in which the amount of climatic change experienced was the smallest.

We note that for our study region (West Australian Wheatbelt), Connor et al. (2015) predicted that the level of reforestation would be considerably less than in the agricultural regions in eastern Australia they analysed in detail, and less than the average level reforested across the continent. Also, as raised in the Introduction, this collection of studies did not allow for the management of agricultural land uses to be adapted, nor the possible beneficial effect for 'CO_2 fertilisation' on agricultural production.

Lastly, their most bullish predictions of land use change from agriculture to sequestration were associated with carbon prices in 2050 exceeding AU$110 t$CO_2$-e, and as high as AU$200/tCO_2-e (in 2010 dollars) (Bryan et al., 2016a). Whilst some believe that high prices will be required in the future if climate change is to be successfully addressed (e.g., Garnaut, 2008), these prices are much higher than those featured in contemporary policies. For example: Australia's 'carbon tax' was $24.15/t$CO_2$-e in 2014; at the conclusion of 2015, prices in the EU and South Korean Emission Trading Systems were equivalent to about AU$14/t$CO_2$-e and AU$15/tCO_2-e respectively (ICAP, 2016); and $12.10/t$CO_2$-e is the average price in the Australian Government's Emission Reduction Fund, as of mid-2016 (CER, 2016).

Several factors could potentially change the economic viability of sequestration compared to what we modelled. We assumed that the opportunity cost of foregone agricultural production and the carbon price would both remain constant in real terms. Under different assumptions, the viability of sequestration would obviously differ. For example, if opportunity costs decreased (increased) and/or carbon prices increased (decreased) in real terms, then sequestration would become more (less) attractive. Our estimates of sequestration rates were based on methodologies specified for the Australian Government's Emission Reduction Fund, which tend to be conservative. Different species of woody vegetation, and/or planting configurations other than mallee 'block' plantings may offer greater sequestration (Paul et al., 2015). Given our analysis is focused on interactions with climate change rather than the financial attractiveness of sequestration *per se*, the above considerations only matter to the extent that their effect on the attractiveness of sequestration differs between the base-case and other climate scenarios.

The MIDAS model we used portrays a single year with 'average' weather. Consequently, we considered only changes in 'average' weather, not changes in extremes or variability. Such changes in the riskiness of farming may modify farmers' decision in two ways: most farmers in the region are averse to risk and will seek to manage their farm to limit risk, and/or farmers will modify the year-to-year tactical decisions that they make in response to current weather conditions (Pannell et al., 2000). However, in the context of interactions between climate change and mitigation policy, the omission of risk is only a limitation to the extent that with climate change there is an increase in the variability of income from traditional agricultural pursuits relative to the variability of sequestration income, and not the potential for sequestration to be a steadier source of income *per se*.

There is increasing pressure to keep global temperature increases below 1.5°C, particularly since the Paris COP21 Conference. Many believe that achieving this will require emissions to be ‘negative’ in the future, through the wide-scale deployment of strategies to actively remove carbon from the atmosphere, including sequestration (e.g., Smith, 2016). However, our study indicates that the prospects for carbon sequestration from trees in this farming environment may not be strong, and could worsen as climate change occurs. Another form of sequestration, the sequestration of carbon in agricultural soils was not part of our analysis. Nonetheless, under all but the most benign of the climate scenarios we considered the growth of crops and pastures decreased. Lower plant production tends to adversely affect soil sequestration because it means less inputs of organic material into the soil, and therefore ultimately lower levels of carbon in the soil (Baldock et al., 2012). Indeed, elsewhere in Australia the main factor found to be influencing carbon levels in agricultural soils is climate, with agricultural management much less influential (Robertson et al., 2016).

In Australia, there has been considerable government interest in using the agricultural sectors as a major and key component of mitigation efforts. Hence our finding that in one of Australia’s major agricultural regions climate change may interact negatively with sequestering activities, is a potentially important insight for policymakers. Worldwide, in other situations and different regions, the performance of sequestration is likely to vary, including its response to climate change. A more universal take-home message therefore is the need for greater recognition of the potential for interactions between future changes in climate and the cost-effectiveness of mitigation activities.

7.5.2 Relative impacts of climate change and mitigation policy

With the exception of the most benign scenario, climate change appears to have a greater effect on farm profitability in the study area than mitigation policy involving a carbon price on on-farm emissions. Nevertheless, a carbon price on agricultural emissions is an unlikely prospect in Australia. This is partly because of the transaction costs that would likely be involved, and partly because of political concerns about the impact of such a price on the profitability of farmers, who tend to be price takers in international markets. Indeed, because of agriculture’s trade-exposure, if a carbon price were applied to agriculture then farms would likely be protected from its full impost (e.g., provided with a quantity of ‘free permits’) (Thamo et al., 2013). In this situation, compared to the effect of climate change, the effect of a carbon price would likely be even lower.

In reality, costs of climate change under each climate scenario would be less than we have estimated, because of adaptations involving yet-to-be-developed strategies or technologies. However, future technological developments or other breakthroughs that enable on-farm emissions to be more cost-efficiently reduced could equally also reduce the impact of a carbon price to agriculture emissions.

Whether climate change would also likely have a greater effect on farm profitability the implementation of a carbon price to on-farm emissions in other study areas in unclear. The impact of climate change will obviously differ geographically, with agricultural production in cooler areas potentially benefiting from climate change (Challinor et al., 2014). Farming systems in other study areas may also have different emissions profiles.

Limitations of our study include assuming that changes to climate would occur equally across all months of the year. In reality, changes may be distributed unevenly, and crop yields in the study area are more sensitive to precipitation and/or temperature changes at particular times of the year (Ludwig et al., 2009). We also did not consider the possibility of ‘feedback’ changes in agricultural prices (in response to climatic change and/or mitigation policy) (e.g., Connor et al., 2015). Should feedbacks push commodity prices upwards then the cost-effectiveness of mitigation through the agricultural sector would likely decline. The impact of climate change on farm profits would also be moderated.

7.6 Conclusion

Changes in climate predicted for the Wheatbelt region of Western Australia appear likely to have a negative impact on farm profitability. A policy to impose a carbon price on on-farm emissions would also reduce farm profitability, although to a substantially lower extent than the impact of climate change. Projected climatic changes also reduce the cost-effectiveness of reforestation to sequester carbon, by reducing the rates of sequestration per land-area and/or lessening the desirability of using land for sequestration. The extent to which these outcomes would extend to other regions is unclear. Elsewhere, climate change could potentially positively impact upon mitigation strategies like sequestration. Therefore, the more globally-relevant conclusion is that in order to develop successful agriculture/land-based mitigation policy, it is prudent to consider the potential impacts of future climate change on the management actions promoted by the policy. Analysis of the impacts of climate change and mitigation policy in isolation, as has typically occurred in much research to date, may hinder the development of effective policy responses to climate change.

7.7 Acknowledgements

We thank Jenny Carter for her assistance with the 3PG modelling.

7.8 Supplementary Material

7.8.1 Reforestation for sequestration—further details

7.8.1.1 Sequestration estimate, emissions and timeframes

Consistent with Emission Reduction Fund methodologies, carbon in above- and below-ground tree biomass plus litter and debris was included in our estimate of sequestration and the estimate was reduced by 5% to allow a 'risk-of-reversal buffer' (ComLaw, 2013). Greenhouse gases emitted during the reforestation process (e.g., diesel fuel use) were also deducted the estimates of sequestration. In the Emissions Reduction Fund, increases in sequestration can be claimed indefinitely as the vegetation grows. However, permanence provisions require the sequestering land use to be maintained for at least either 100 or 25 years after the first year sequestration is claimed, but landholders opting for the 25 year period can only claim credits for 80% of their total sequestration owing to the potentially more impermanent nature of their abatement (House of Representatives, 2014). As preliminary analysis revealed that the 100 year option was likely to be less financially attractive to a landowner, we assumed sequestration would be claimed for the first 25 years of tree growth but accordingly, that credits could only be claimed for 80% of the sequestration.

7.8.1.2 Sequestration economics

A sequestering land-use returning dynamically-varying income over a long time period is not directly compatible with MIDAS, which portrays a single year of production, in a perpetual cycle. Therefore, the net present value of sequestration revenue was annualised over 25 years (using a real discount rate of 5%), yielding an equivalent annual revenue from sequestration suitable for inclusion within the MIDAS framework (Thamo et al., In Press).

Based on Sudmeyer et al. (2014) we assumed establishment costs of $1503/ha plus transaction costs (monitoring and auditing fees) of $25/ha/year. Compared to Summers et al. (2015) and Bryan et al. (2016a) these costs are at the lower end of the scale.

7.8.2 Sensitivity analysis

7.8.2.1 Testing the robustness of findings about sequestration

The stability of our findings about the effect of climate change on sequestration was tested with sensitivity analysis. This sensitivity analysis was conducted not for the effect of uncertainty about the extent of changes to the climate in the future *per se* (as this uncertainty may affect both agricultural production and tree growth), but rather to examine how the supply of sequestration might vary if the response of *just* tree growth to a given climatic change differed. The middle graph in Figure 7.6 shows the supply of sequestration based on the predictions of the 3PG forestry model. This is the same graph that is presented in Figure 7.4 of the main paper. The top and bottom graphs show the supply of sequestration if a given climate scenario's impact on tree growth (i.e., sequestration) was 20% less or 20% more than the forestry model predicts respectively. The impact on agricultural production is held constant and unchanged for all three graphs, being exactly as predicted by the agricultural simulation models (APSIM and GrassGro).

Results suggest our finding about the supply of sequestration under climate change are relatively robust. Figure 7.6a) shows that even if 3PG has overestimated the impact that a given climate change scenario will have on tree growth by 20%, but the impact on agricultural production is exactly as predicted by the APSIM and GrassGro simulation models, then the amount of sequestration obtainable for a given carbon price will still decrease with climate change (although at prices between $40/tCO_2$-e and $50/tCO_2$-e there is a slight increase in the supply of sequestration for the milder climate scenarios, this increase is also predicted in Figure 7.6b)). Nothing in these results suggests that in the study area, the biophysical impacts of climate change will stimulate a notable increase in the supply of sequestration from agricultural land.

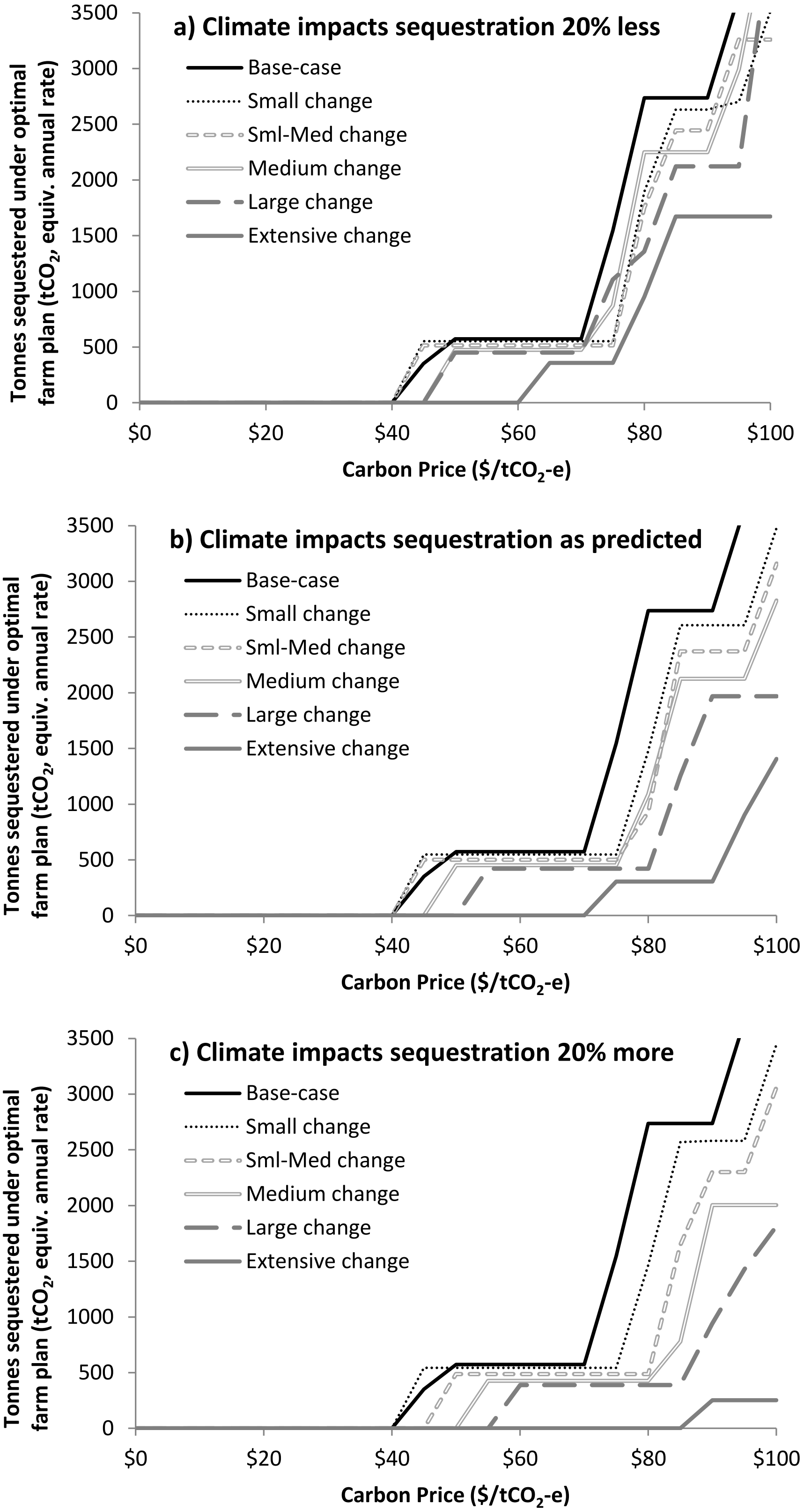

Figure 7.6. Supply of sequestration if tree growth was **a)** 20% less, **b)** the same, or **c)** 20% more sensitive to changes in climate than the 3PG forestry simulation model predicts.

Chapter 8. Discussion

8.1 Discussion

In this thesis, a number of bioeconomic analyses were conducted to investigate how climate change (and the implementation of policies designed to mitigate it) may affect agriculture in the Western Australian Wheatbelt. Results suggest that the profitability of farming systems in this region are potentially quite sensitive to changes in climate, with substantial reductions in agricultural profitability a possibility if the warming and drying trend predicted for the region translate into large temperature increases and/or rainfall reductions. Conversely, the potential for agricultural land in the Wheatbelt to act as a low-cost carbon sink seems limited, particularly for soil carbon. To incentivise wide scale land-use change to sequester carbon is likely to require a relatively high carbon price. Furthermore, climatic change may limit the amount of abatement obtainable from sequestration.

The thesis findings are discussed in more detail in this chapter. This discussion is structured around the thesis' two aims: (i) examine the potential for agriculture to provide cost-effective abatement in general, and in the study region specifically; and (ii) investigate the potential bioeconomic impact of climate change on the Wheatbelt region. Lastly, limitations of the analyses underpinning the findings are considered, and where applicable, avenues for further research identified.

8.1.1 Thesis aim #1: Mitigation opportunities and policy

8.1.1.1 Sequestration

Based on the analyses presented in this thesis, it appears that (at least for the study region) carbon sequestration is unlikely to provide a low-cost abatement option or an additional income source to revitalise rural economies. Rather, opportunities for sequestration are relatively costly, are difficult to exploit (in terms of policy design), and may be affected by climate change in the future.

For reforestation to become attractive, the carbon price needed to exceed \$35 –40/tCO_2-e (Chapter 5 and Chapter 7). This is relatively consistent with other estimates of \$40 – 70/tCO_2-e being required for reforestation to be viable in study region (Petersen et al., 2003a; Flugge and Schilizzi, 2005; Flugge and Abadi, 2006; Kingwell, 2009; Jonson, 2010), but higher than any carbon price mentioned in the contemporary political debate in Australia, or emissions markets/carbon trading schemes globally. Some have suggested that if climate targets are to be met much higher prices (exceeding

\$100/tCO_2-e in real terms) will be required, particularly in the long term (e.g., Garnaut, 2008). However, in 2016, prices of this magnitude seem unlikely, at least in the short to medium term. If the significant reductions in the cost of renewable technology recently is a guide—in the five years between 2010 and 2015, the cost of wind energy fell by 30% and large-scale solar photovoltaic generation by two thirds (IEA, 2015)—it could even be questioned whether such high prices will eventuate in the long term.

Estimates of the carbon price required to compensate for a change in land use to sequester more carbon in the soil (by increasing the amount of pasture phases in the farming system) varied between \$35 and \$240t/CO_2-e, depending on the balance between livestock and grain prices (Chapter 3). Using average prices for both, an incentive of \$87/tCO_2-e was required to compensate for any increase in pastured land and thus sequestration. As highlighted in Chapter 4, these estimates of the 'break-even' carbon price can be sensitive to assumptions about the dynamic fate of the price of carbon and the opportunity cost incurred by adopting sequestration practices, as well as whether the incentive to compensate for sequestration is received based on a dynamically-varying sequestration rate or a constant (average) rate.

Unlike the analysis of reforestation (Chapter 5 and Chapter 7), the analysis of soil carbon in Chapter 3 did not take into account the time-value of money, meaning implicit in the analysis were assumptions that: (i) the carbon price and opportunity cost of changing land use both remained constant in nominal terms or in real terms; (ii) incentives for sequestration were received based on the constant average rate of sequestration and; (iii) there were no permanence obligations. Chapter 4 explored the consequences of making implicit assumptions such as these. For instance, in Chapter 4 it is argued that it is preferable to base incentives for sequestration on the dynamic rate of sequestration because they capture the rate at which the abatement actually occurs over time, which (because of the time-value of money) therefore facilitates more equitable and market-efficient trade-off between different mitigation options. Dynamic rates will, *ceteris paribus*, also make sequestration more financially attractive to land managers. Despite this, no follow-up analysis of soil carbon, specifically taking into account the time-value of money, was performed. This is because it was felt that the main finding of Chapter 3—in Wheatbelt farming-systems storing carbon in soil does not appear to be a low-cost way of mitigating greenhouse gas emissions—would not fundamentally change in any new analysis, for several reasons.

Firstly, the results of Chapter 3 would have to change substantially before they would suggest that soil carbon sequestration is likely be attractive to farmers in the study region.

Secondly, when estimating the incentive/carbon price required to compensate for increased sequestration in Chapter 3 a 30 year timeframe was used. Because sequestration rates decrease with time, if timeframes longer than 30 years were considered (for example a more ‘permanent’ 100 year timeframe), then a higher carbon price would be required to breakeven (though, as demonstrated in Chapter 4, the impact of the increased timeframe would be somewhat moderated by the effect of discounting to take into account the time-value of money).

Thirdly, transaction costs were omitted from the analysis in Chapter 3. The analysis is not alone in this regard: ignoring or assuming no transaction costs is very common in analyses of sequestration (Capon et al., 2013). However, the transaction costs associated with the inclusion of sequestration in an offset policy may in fact be particularly high, due to the inherent characteristics of sequestration (Cacho et al., 2013). These transaction costs will reduce the financial attractiveness of sequestration, with transaction costs incurred upfront especially onerous (Chapter 4).

Lastly, the analysis in Chapter 3 did not take into account potential changes in emissions caused by adoption of a sequestering practice, such as an increase in livestock emissions if the amount of pasture in the farming-system increases. Subsequent to the publication of the paper in Chapter 3, some other analyses have examined this topic. For sheep pastures in Victoria, in the most-favourable scenario Meyer et al. (2016) studied, the net abatement by sequestration decreased by 43% once increased emissions from grazing livestock were deducted. In the worst-case scenario, livestock emissions exceed sequestration more than tenfold. In another analysis, for the high rainfall, southern fringe of the Wheatbelt region, increased livestock emissions neutralised nearly a quarter of the emissions sequestered by the adoption for perennial pasture (Thomas et al., 2012). However, Thomas et al. (2012) only considered the first 16 years after conversion to perennial pasture in their analysis—a relatively short timeframe in the context of sequestration. As sequestration rates tend to progressively slow with time, but emissions by grazing animals do not, the net result (shown in Figure 2.4, page 45) is that over time the livestock emissions neutralise an ever-increasing proportion of the abatement from sequestration.

The paper presented in Chapter 3 represents the second published economic analysis of soil sequestration in Australia (the results of the first analysis, by Grace et al. (2010) are discussed in the introductory chapter). Only one other analysis has since been published (as of mid-2016). In a New South Wales-based study, White and Davidson (2015) also concluded that sequestering carbon by converting from cropping to pasture was unlikely to provide meaningful mitigation. They also found that the methane emissions due to the increase in grazing livestock associated with the land-use change would, in most cases, completely negate the mitigation from the sequestration.

Theoretical analysis in this thesis supports a conclusion that the potential to use agricultural land as a low-cost carbon sink seem limited. The results of the Emissions Reduction Fund (ERF) provide opportunity to check the theoretical against the empirical. Under the ERF the Australian Government has, as of June 2016, entered into $1.73 billion worth of contracts to purchase abatement equivalent to 143 million tonnes of CO_2-e from 348 different projects across Australia at an average price of $12.10/t$CO_2$-e (CER, 2016). Soil carbon accounts for just 2.2% of all projects and 5.5% of the total abatement contracted (7.8 million tonnes of CO_2 over 10 years). The 0.78 million tonnes annually that this equates to is a long way from the “at least 150 million tonnes of CO_2” that the Coalition (2010) suggested their Direct Action (i.e., ERF) policy would see sequestered annually in agricultural soils by 2020 (and for a price of $10/t$CO_2$). So far no soil carbon projects have been established in Western Australia. Contracts for reforestation by planting trees on farmland stand at 6.2 million tonnes of CO_2 over 9.9 years on average (or 4.3% of the total abatement purchased). Four reforestation projects have been established in the Wheatbelt region; combined, they are contracted to sequester less than one million tonnes of CO_2.

8.1.1.2 Mitigation policy

In reality, the sequestration that has been secured under ERF may not be as effective as it appears on the contract. Non-additionality, leakage and impermanence will reduce the efficiency of mitigation policy and/or reduce the extent to which the abatement theoretically secured by the policy actually contributes to climate change mitigation. Much has been written about additionality, permanence and leakage (e.g., Gustavsson et al., 2000; Dutschke, 2002; Herzog et al., 2003; Lee et al., 2007; Murray et al., 2007; Kim et al., 2008; Passey et al., 2008; Sun and Sohngen, 2009; Gillenwater and Seres, 2011; Cacho et al., 2013; Horowitz and Just, 2013; Barnes et al., 2014); these issues are also discussed in Chapter 2, though with an Australian focus, using the Carbon Farming Initiative (CFI) as a case-study.

Assorted mechanisms for ensuring additionality have been proposed, or are in use in various programs around the world (Kollmuss et al., 2010). Nevertheless, the intractable nature of the counter-factual issue that lies at the core of the question of additionality is such that “There is no bulletproof way to ascertain the additionality of most offset projects” (Kollmuss and Lazarus, 2011, p.542). The so-called ‘common-practice’ approach to additionality used in the CFI was no exception to this. It suffered from subjectivity (exactly what constitutes an activity being ‘common’?), and, in the form it operated in the CFI, it treated adoption simply as a binary yes/no question, thereby ignoring increases in scale-of-adoption. Its primary advantage—reducing transaction costs—only applied at the start of a scheme. Whilst the need to update additionality is recognised in the literature (e.g., Gustavsson et al., 2000), Chapter 2 points out that theoretically, updates to both the sequestering activity and the counterfactual business-as-usual practice it replaces should apply retrospectively. This had not been recognised previously, and is a relevant issue not only to the CFI, but for any sequestration policy.

Permanence is usually defined based on an arbitrary time period (e.g., 100 years in the CFI), after which sequestered carbon can be freely re-released to the atmosphere. Chapter 2 demonstrates that the release of this sequestered carbon is likely to raise atmospheric CO_2 levels to higher level than they would have been if the sequestration had never occurred. Therefore, such re-release may be undesirable if it occurs when atmospheric greenhouse gas levels are still at dangerous levels. If sequestration is to be regarded legitimately as being of *equal* effectiveness as a reduction in emissions then there should be no free release; rather it should have to be maintained until replacement abatement is purchased.

An alternative to permanent sequestration that has attracted attention in the literature is temporary sequestration or ‘carbon rental’ (e.g., Dutschke, 2002; Feng et al., 2002; Lewandrowski et al., 2004; Keeler, 2005; Murray et al., 2007; Kim et al., 2008; Cacho et al., 2013). Historically, international acceptance of temporary sequestration has been poor, with fungibility and how to appropriately value/scale temporary offsets relative to permanent abatement being sources of contention (e.g., World Bank, 2011). Nonetheless, when the CFI was reengineered to create the ERF, an alternative option of a shorter, 'temporary' 25 year permanence period was added to supplement the 100 year permanency rule. At end of the 25 years, the sequestered carbon can be freely re-released, with the government, as the purchaser, liable for this (Macintosh, 2013). To compensate for this risk of earlier re-release, land managers choosing the 25 year option have the amount of credits that they can claim scaled down by 20% (House of Representatives 2014).

Whether this 20% scaling is adequate comes down to a financial argument about discounting: is the net present value of the cost of purchasing the temporary sequestration and then the cost of potentially replacing it in 25 years, less than the net present value of the cost of purchasing permanent abatement outright? The answer is dependent on a number of uncertainties: the price of carbon in the future, appropriate discount rates, how immediately after the expiry of the permanence period will the temporary sequestration be released, etc. Nonetheless, across a range of reasonable assumptions about these unknowns, it could be concluded that the 20% deduction is not a sufficient discount. However, that conclusion applies to temporary sequestration that is being replaced after at least 25 years; in reality sequestration is purchased by the government incrementally, and it is only the *first* incremental claim that the ERF's permanency rules require to be stored for 25 years. Accordingly, subsequent sequestration claimed for the same project need only be stored for progressively shorter time. Temporary sequestration purchased in the 25th year could even be released and have to be replaced within 12 months. Despite this, the 20% deduction applies to all sequestration, whether it be claimed in the first or 25th year of a project. Once this is taken into consideration, the rate of deduction is indeed far too low. In the ERF to date, all soil carbon sequestration, and more than 91% of the reforestation projects are of this 25 year, temporary type. Whilst allowing free release and/or under-discounting might make temporary sequestration more attractive—and policies appear more successful—in the end sequestration is only a means to an end; its pursuit should not come at cost to the main objective of preventing dangerous climate change.

Leakage remains an on-going issue. As mentioned, in Chapter 2, 'indirect leakage'—an increase in emissions due to substitutions or market adjustments potentially in other countries occurring in response to the sequestration—can be significant (e.g., Gan and McCarl, 2007; Meyfroidt et al., 2013), yet is essentially overlooked in the CFI and the ERF. Internationally, leakage is ignored on the basis that those increases in emissions will show up in the other country's emissions. On the one hand, that seems reasonably justifiable, especially since accounting for international leakage in a domestic scheme would be difficult. It does, however, rely on the premise that negative externalities (emissions) will be internalised with equal veracity in foreign countries. In Chapter 2 it was shown that even when leakage is accounted for, timing discrepancies between the receipt of financial benefits from abatement and the payment of the penalty for leakage, could still result in situations that are financially attractive to landholders despite an increase in final net emissions.

Dealing with additionality, permanency and leakage ultimately requires a balance between legitimacy (abatement integrity), transaction costs and participation. The higher transaction costs associated with a stringent approach to sequestration to ensure veracity would reduce participation. A more relaxed approach, with lower transaction costs, would increase participation, but the lower quality of the resultant abatement would also reduce the efficiency of mitigation. While there may be strategies to optimise this balance, any approach will nonetheless remain a trade-off, the effect of which will be to reduce the efficacy of abatement obtained from sequestration. Because of this, the cost-effectiveness of sequestration estimated theoretically in the analyses in this thesis, would, effectively, be lower in reality.

How the balance between legitimacy and participation has played out in the ERF is worthy of discussion. Most of the abatement the Government has contracted under ERF is to occur in the future (over the next 8.8 years on average). In fact, as of May 2016, only 3.6% of the 143 million tonnes of CO_2-e committed under contract to the Government has actually been delivered. Therefore it is too early to ascertain how much of the delivered abatement has actually occurred and is genuine, and how much exists on paper only (to the extent that it is possible to ascertain this anyway, given additionality can only ever be determined relative to an unobservable, counterfactual scenario). However, what is known is that when the CFI was modified to create the ERF in late 2014 (Section 1.3.1) the permanence and additionality rules from the original CFI were 'streamlined'. These changes to permanency have been discussed above; in the case of additionality, the so-called 'common-practice' criteria used in the CFI was replaced with the requirement that abatement projects simply be new and unlikely to be occurring as a result of another/different government program. Though logically just because an activity or project that happens to lower emissions is new, does not also mean that it would not necessarily be occurring as part of the normal course of business. Indeed, it is exactly these types of activities or projects that are most likely to undertake 'mitigation' for the lowest price (Burke, 2016). Even though the common-practice approach was criticised above, an approach that instead lacks any meaningful mechanism for ensuring additionality greatly increases the risk of adverse selection (Akerlof, 1970), and the risk of creating a market for lemon sequestration that it entails.

The 'streamlining' of the additionality and permanence rules during the creation of the ERF represent a shift in the balance between the legitimacy of an offset scheme and participation, in favour of the latter (Macintosh, 2013). However, it would appear that broad-scale participation of a type that may be consistent with widespread free-riding by non-additional participants has not occurred, despite the absence of a mechanism to prevent such a thing. This suggests that other factors such as transaction costs, uncertainty about climate policy and the future of the scheme (Dumbrell et al., 2016), and perhaps a fear of losing future flexibility are obstacles to farmer participation, or at least are obstacles when the carbon price averages \$12.10/tCO_2-e. Even if these obstacles could be overcome, then the insights of Akerlof (1970) would suggest that those most amenable to supplying sequestration to the scheme would be those already undertaking sequestering practices.

In this thesis, the policy challenges associated with developing mitigation policy for agriculture have been explored through an Australian lens. The challenges are, however, fundamentally universal in nature. Therefore, policy challenges discussed in this thesis are likely to be applicable beyond the study region of the thesis.

8.1.1.3 Agricultural emissions

Results in Chapter 5 suggest that, for a typical farm in the central part of the Wheatbelt region, the application of a mandatory carbon price to on-farm emissions could help mitigate emissions but would have a substantial impact on profitability. For instance, a \$50/tCO_2-e carbon price reduced on-farm emissions by about 50%, but also decreased farm profit by 30 –60%, depending on the method used to account for emissions and sequestration (in circumstances where sequestration was viable it moderated the impact of the carbon price on emissions). If, on account of a lack of commensurate action on agricultural emissions at the international level, free permits or exemptions were granted to trade-exposed farm businesses then it would seem advisable that these are on-tradeable.

As flagged in the introductory chapter, rather than being subject to a mandatory carbon price (a 'stick' approach), a 'carrot' approach could be taken to agricultural emissions. In a 'carrot' policy approach, farmers claim credits for voluntarily reductions in emissions, which they could then on-sell to buyers such as the government or a polluter needing to offset their emissions. This is how agricultural emissions have so far being approached in Australia, through the CFI/ERF policies[1]. However, in this thesis no analysis of this policy approach to agricultural emissions was formally conducted. That said, the study presented in the appendix chapter of the thesis was conducted under the scenario of there being financial incentives for voluntary reductions in emissions (results showed that the emissions savings were small compared to their opportunity cost, meaning a large carbon price would be required to incentivise them.

Options for reducing agricultural emissions essentially fall into two categories: management-type options and technological options (Cooper et al., 2013). Management-type options involve changes in a production system to change the type or amount of output/s produced, use less of an input, or improve the emissions-intensity of production. Typically the mitigation potential of these management-type options are modest (Henry and Eckard, 2009) and, with the exception of efficiency improvements, they involve changes in output which are likely to evoke counteracting price feedback. In the analyses of Chapter 5 and Chapter 7, only these management-type options were represented and available to respond to a mandatory price on agricultural emissions. Technological solutions that, for example, alter the microbial populations in the digestive systems of ruminants or inhibit nitrification to reduce N_2O emissions from soils (Henry et al., 2012), may yield more significant emissions reductions. However,

[1] As of June 2016, methodologies for accounting for reductions in the emissions associated with irrigated cotton, beef, pork, and dairy production have been approved for use in the ERF (Department of the Environment, 2016). Because none of these methodologies are applicable to the dryland crop/sheep farming systems of the Wheatbelt, there is no empirical evidence to draw upon in regard to voluntary reductions in emissions in the study region. It perhaps worth noting though that analysis of the data from CER (2016) reveals that, at a national level, reductions in agricultural emissions (accounted for under the aforementioned methodologies) represent just 0.67% of the total abatement contracted in the ERF thus far.

more research is required before they are commercially available (Henry et al., 2012; Cooper et al., 2013).

From a policy perspective, permanence is not an issue for offsets issued for reducing agricultural emissions (Cacho et al., 2008). However, the instantaneous nature of emissions reductions presents a difficulty in another way: it is challenging to ensure that an emission reducing strategy is actually being implemented over the entire area/number of livestock and/or duration that it is claimed to be. In contrast to sequestration, evidence of a reduction in emissions may be less durable. For instance, verifying that an emissions-reducing supplement was fed to livestock two years or even two months ago may not be possible (by definition, if a practice is additional, the farmer should have a motivation to avoid doing it). Therefore the development of techniques for verifying emissions reductions should not be overlooked as part of any research into agricultural emissions. This is of course only applicable to mitigation options that are not attractive for farmers to adopt as part of business-as-usual.

One way to avoid the many policy challenges discussed above is to have mitigation options that reduce emissions whilst simultaneously improving profitability (without incentivisation from policy) and are easy to adopt. Such options circumvent the need for verification, the intractable problem of additionality, and even the need for any supportive policy framework, all while providing mitigation at zero cost. An important distinction needs to be made here. Whilst *policy* that supports non-additional mitigation is ineffective, *research* to develop abatement options readily adoptable as part of business-as-usual may be highly beneficial. Some sources of agricultural emissions may indeed be profitable to address. Methane production by ruminant livestock—the source of about one third of all global agricultural emissions and two thirds of Australian agricultural emissions—entails the loss of 2 –12% of dietary energy, so reducing these emissions would improve feed conversion efficiency (e.g., Martin et al., 2010). It may be questioned whether some of the $1.73 billion (with further $816 million budgeted) of public funds paid out purchasing abatement of questionable quality through the ERF may have been better spent researching technological options for reducing agricultural emissions.

8.1.2 Thesis aim #2: Effect of climatic change

8.1.2.1 Impacts and adaptation

The relationships between production changes, adaptation and profitability under varying climate scenarios were investigated in Chapter 6. Depending on the climate scenario, the impact of climate change on farm profit varied from moderately positive to decidedly negative, but in the majority of scenarios profitability decreased. Results indicate the potential for sizeable reductions in the economic performance of agriculture under severe climate change. The beneficial effect of increased atmospheric CO_2 was moderated by warming and/or drying. For instance, with no change in precipitation and just a 1°C temperature increase, then an increase in CO_2 from 475 to 575 ppm boosted profit by $113,000. But with a 30% reduction in rainfall and 4°C temperature increase, the difference in profit between 475 or 575 ppm was only $49,000. Warming and drying were not both required for profit to be negatively affected: across the 72 climate scenarios examined in Chapter 6, if either temperature increased by more than 2.5°C *or* rainfall reduction exceed 20% then profit declined, irrespective of what happened to the other climate parameters.

Previous analysis of climate change impacts on agriculture in the Wheatbelt region has tended to consider impacts on only biophysical parameters, namely production/yield. In the bioeconomic analysis in Chapter 6 the impact on profit was disproportionally larger than changes in production. This is because changes to production, whilst not so large in relative terms, affect the portion of production that forms much of profit margin once fixed costs and the farmer's personal expenses have been deducted. The flipside of this is that in most climate scenarios, relatively minor increases in yields (e.g. due to an improved crop variety) or prices would be sufficient to counteract the financial impacts of climate change.

The existing (biophysically-focused) literature for the study area has also tended to consider impacts on just isolated components of the farming-systems (usually wheat production), and done so in a simulation framework. In contrast, in Chapter 6, impacts were investigated at the farming-system level and in an optimisation framework, meaning the analysis endogenously adapted land use and management in response to climate change. Under a warming and drying climate the main changes to the farming system's management were a lessening of fertiliser applications and a considerable reduction in the sheep flock. Due to the need to retain a fixed amount of un-grazed pasture for soil conservation, reductions in livestock numbers were disproportionately large relative to reductions in pasture growth. Consistent with this, there was a slight trend away from pasture toward cropping in the farming system, although farm returns were not especially sensitive to whether this or other changes in land-use occurred. Had these adaptative changes in management and land use not been allowed for—as has tended to be the case in the existing literature for the study area—then the impact of climate change on farm profit would have been 15 –35% greater. The scope of this adaptation was of course limited to just those options that are currently known/available and specified in MIDAS; no doubt other options will be developed in the future. However, given that agricultural production in the region has traditionally tended to be limited by moisture availability, adaptation options developed in the future may also have been beneficial under a counterfactual, no-climate-change scenario. Therefore, whilst future adaptation and productivity growth may help to offset the economic impacts of the climate change in the study region, the actual true impacts may still be large.

8.1.2.2 Impacts and mitigation interacting

Just as it impacted agricultural production, climate change also impacted sequestration. Results in Chapter 7 indicate that under scenarios involving more substantive changes to the Wheatbelt's climate, the quantity of sequestration from reforestation obtainable for a given carbon price will decline. There were both biophysical and economic reasons for this. Biophysically, as the extent of climate change increases, the amount that would be sequestered by trees growing on a given hectare of land reduces. Economically, as the extent of climate change increases, the financial attractiveness of sequestration as a land use lessens, meaning fewer hectares were allocated to sequestration. Together, these dual negative impacts suggest that under a changing climate, there could be risks to pursuing reforestation-based climate mitigation in the Wheatbelt of Western Australia. The biophysical effect alone means that, even if the area of trees were instead maintained following climate change, there would still be less abatement realisable from sequestration than might otherwise be anticipated based on the climatic conditions historically experienced in the study area. This result appeared relatively robust under sensitivity analysis.

A recent study (Hobbs et al., 2016) clearly demonstrated that climate change could substantially reduce the sequestration that the reforestation of given hectare of agricultural land could deliver in South Australia. Whilst this is consistent with the results of Chapter 7, it is not clear whether this biophysical result would also translate into a reduction in the economic competitiveness of reforestation, as found in Chapter 7, nor if these findings will hold for other regions of Australia. In a recent integrated assessment of the dual effects of climate change and mitigation policy on land use in Australia to 2050, the proportion of farmland that Connor et al. (2015) predicted would be reforested for sequestration in Wheatbelt study region was much lower than the amount they predicted would be reforested in the other agricultural regions in eastern Australia they analysed in detail, or indeed the aggregated, national average. Given spatial variability in the results of that study it is clearly inappropriate to offer any national- or globally-relevant conclusions from Chapter 7, other than to say that in order to develop effective land-based mitigation policy, it is prudent to consider the potential impacts of future climate change on the mitigation actions targeted by the policy. Analysis of the impacts of climate change and mitigation policy in isolation, as characterised by much of the literature to date, may constrain the development of effective policy.

Results also suggest that farm profitability is much more sensitive to climatic change than to a mitigation policy involving a carbon price on agricultural emissions. Even without any free permits and/or income from sequestration (i.e., even under the scenario where a carbon price has the greatest impact on farm profitability), this impact is still less than the effect of even some of the milder changes in climate projected for the study region. For instance, a 10% reduction in precipitation accompanied by a 1.25°C temperature increase and 450 ppm atmospheric CO_2 would have an impact on farm profit equivalent to a \$104/tCO_2-e price on agricultural emissions. Note that this is when agricultural emissions are estimated using national greenhouse gas accounting methodologies. As explored in Chapter 5 these methodologies may overestimate emissions in Wheatbelt conditions. Climate change itself also reduced on-farm emissions, principally due to a reduction in livestock numbers when the farming system was subject to a changed climate (although the reduction in livestock will be less if price feedback occurs).

8.2 Limitations and further research

This thesis strives to investigate how agriculture in the Wheatbelt region may be influenced by both physical changes in climate and the implementation of policies to mitigate those emissions responsible. However, in reality climate change is such a broad and dynamically-evolving issue that canvassing all aspects of it in one thesis, even just for a specific study area, is not possible.

8.2.1 Mitigation options

In the work in this thesis potential co-benefits from sequestration such as ecosystem services were not considered (e.g., Elbakidze and McCarl, 2007). For instance, reforestation could have the co-benefit of providing habitat and resources for native fauna (Bradshaw et al., 2013). To maximise the provision of such co-benefits from sequestration will likely require additional incentivisation or regulation (e.g., Bryan et al., 2016b). In the ERF, where there is little incentivisation for the provision of ecosystem services, the relatively low uptake of reforestation so far suggests additional policy support may be required before provision of ecosystem services significantly increases the attractiveness of reforestation.

In Chapter 5 and Chapter 7 reforestation was assumed to occur as plantation-style, block plantings. An alternative planting configuration is agroforestry-style belt plantings, where agricultural production is continued in the 'alleys' between the belts. Whilst belt plantings do not sequester more per *overall* hectare, they sequester more per hectare of *planted area* (which in this case is just the area of the belt). This is because the trees gather additional moisture and nutrients from the alleys adjacent to the planted belt. However, this is offset by the fact that the opportunity cost for the agricultural production that is lost due to reforestation is now also incurred over more hectares than just the planted area, because the competition for resources also reduces agricultural production in the alleys adjacent to the belts, especially if the trees are unharvested (Sudmeyer and Flugge, 2005; Sudmeyer et al., 2012). Once this is allowed for, it is not clear whether the results would significantly change if the research were extended to account for belt-style plantings.

Reforestation plantings for bioenergy (and/or biofuels) rather than directly for sequestration is potentially relevant to the study region, but was not analysed. The potential for bioenergy to impact food prices is well-documented, but much less an issue for so-called 'second generation' bioenergy that utilises non-food feedstock, like crop residues (Herr et al., 2011; Kingwell and Abadi, 2014) or woody-biomass from dedicated tree crops (Baker et al., 1999; Bartle and Abadi, 2009) (in this latter case, the impact on food production would be similar to reforesting to sequester carbon). When it comes to climate mitigation in the agricultural sector, bioenergy stands out from other options in terms of the level of policy support it requires. In fact, theoretically, the only policy required is a carbon price on the fossil fuels competing with bioenergy[2]. There are no issues of additionality and permanence (Dutschke, 2002), assuming the carbon price/disincentive on fossil fuel use alone is sufficient to render bioenergy profitable. Nor is there a need for any farm-level measurement (an inherently difficult proposition—Chapter 5) and the associated transaction costs. Leakage is likely minimal from second generation bioenergy: ‘direct leakage’ (as defined in Chapter 2) like emissions from fossil fuels used in the bioenergy production (e.g., harvesting and transport) would be covered by the carbon price; ‘indirect leakage’ is not applicable for straw (a byproduct), and in the case of dedicated woody energy crops leading to land clearing for agriculture elsewhere, the risk would be similar to sequestration plantings.

This relative policy-efficiency, apart from anything else, means bioenergy may warrant further research. The size of the incentive for bioenergy/disincentive on fossil fuel consumption required to render bioenergy profitable looms as the key question. Fundamental to the answer is likely to be the impact of technological innovation, both on processing technologies for second generation bioenergy, and on the cost of competing energy sources, including other renewables (Ajanovic and Haas, 2014). Another consideration is what impact climate change may have on feedstock production (Bryan et al., 2010; Kingwell and Abadi, 2014; Bryan et al., 2016a). Based on Chapter 6 and Chapter 7, under climate change feedstock yields may potentially fall in the study region.

Another mitigation option not examined in the thesis is biochar, which represents a nexus between bioenergy and soil sequestration. It involves the pyrolysis of feedstock material—such as crop residues or woody-biomass—to produce (at varying ratios) renewable bioenergy, and biochar (e.g., Marris, 2006; McHenry, 2009). The latter is charcoal-like, carbon-rich substance which, if applied as a soil ameliorant, can purportedly increase crop yields (Sohi et al., 2010). As biochar is relatively stable and

[2] So long as sequestration is not claimed for carbon stored in the feedstock before it is harvested, then this carbon price need not apply to the bioenergy (although the combustion of biofuels emits carbon, unlike fossil fuels, this carbon has only recently been removed photosynthetically from the atmosphere, and further, will be removed once again by the next bioenergy crop, meaning it causes no net increase in atmospheric CO_2.

resistant to decay, it offers a resilient form of soil sequestration (e.g., Schneider et al., 2011), circumventing the issue of permanence. Indeed, similar to bioenergy, biochar production is relatively simple in terms of the demands it makes on supporting policy.

In terms of future research, there are two pertinent questions about biochar. The first is obvious: can biochar be a cost-effective mitigation strategy in the Wheatbelt region? The second is perhaps more obscure: is biochar production the *most* efficient use of a given feedstock? Even if the answer to the first question is affirmative, the answer to the second question can be negative. Depending on how the feedstock is pyrolysed, more renewable energy can be produced from it at the expense of reduced biochar output (McCarl et al., 2009). At the extreme of this trade-off, biochar production can completely be foregone and the bioenergy yield maximised. If this bioenergy displaces fossil fuels, the resultant reduction in emissions will be permanent. Therefore the answer to the second question will depend on whether the soil fertility benefits conferred by applying biochar (if there are benefits at all e.g., Blackwell et al., 2010; Van Zwieten et al., 2010; Jeffery et al., 2011), minus the cost of transporting the biochar post-pyrolysis and then incorporating it into agricultural soil, exceed the benefit generated by instead simply maximising the amount of renewable bioenergy produced from the feedstock.

In this thesis, the cost-effectiveness of sequestration was largely estimated using discounted cash-flow analysis (i.e., net present values (NPV)), as explored in detail in Chapter 4. When derived based on NPV, estimates of the incentive required to persuade land managers to adopt a sequestering land-use or practice tend to be lower than the actual incentive required empirically (Plantinga et al., 1999; Kurkalova et al., 2006; Nielsen et al., 2014). This is because discounted NPV analysis has a propensity to underestimate the impact that uncertainty and irreversibility have on decision-makers/investors (Dixit and Pindyck, 1994; Hertzler, 2006; Reeson et al., 2015; Regan et al., 2015). Uncertainty is large because of the long time frames involved with sequestration (often with high upfront costs), and because income can only be derived from sequestration on account of political decisions and not due to any fundamental, underlying demand for this ‘product’. Furthermore, permanency requirements mean landholders lose flexibility to change management in the future and, even if it is possible, changing management can be prohibitively expensive (e.g., re-clearing reforested land).

One simple way to adjust the results of a discounting NPV analysis for this effect is to assume a premium or ‘hurdle’ by which the profitability of the new practice must exceed the old practice for it to be adopted (e.g., Bryan et al., 2014; Bryan et al., 2016a). Another approach, where computationally possible, is to instead estimate the size of the incentive (carbon price) required to facilitate practice change with real option analysis, to explicitly account for the effect of uncertainty and irreversibility on farmer behaviour (Regan et al., 2015). A real options analysis of sequestration in the Wheatbelt region presents as an avenue for further research, although the cost-effectiveness of sequestration already appears to be low when estimated with discounted NPV analysis, so the results of any real options analysis are likely to be confirmatory rather than conflictual. Options analysis may also be useful to the investigation of bioenergy/biofuel production (more so from short-rotation woody crops than crop residues).

Transaction costs are thoroughly relevant to this thesis. It must be acknowledged that whilst they are touched upon regularly in this thesis, at no point are they considered in detail. Chapter 5 examined the implications of the accuracy of emissions and sequestration measurement. A natural extension would be to overlay this with different transaction costs for different levels of accuracy (e.g., Antle et al., 2003). Different assumptions about the temporal distribution, and fate over time, of transaction costs would have added richness to the investigation of the effect of different dynamic assumptions in Chapter 4.

Another limitation is overlooking possible agronomic/productivity benefits of soil carbon. These benefits are wide-ranging but include increased water-holding capacity, improved soil structure and water infiltration, better nutrient retention, reduced erosion and buffering of the soil against pH changes (Sanderman et al., 2010; Meyer et al., 2015; Murphy, 2015; Petersen and Hoyle, 2016). These benefits can be complex and are difficult to quantify, especially in terms of how they translate into economic gains for a landholder and/or society (e.g., Murphy, 2015). For this reason they were omitted from the analyses in this thesis. Likely for the same reason, studies attempting to economically quantify the agronomic benefits of soil carbon are scarce in the global literature (Petersen and Hoyle, 2016). One very recent attempt, and as it happens, for the West Australian Wheatbelt, estimated a tonne of soil carbon to be worth in the vicinity of $1 to $2 per hectare annually in agronomic benefits (Petersen and Hoyle, 2016). They also found that these benefits fell in value with decreasing rainfall.

The omission of the benefits of increased soil fertility may be offset by another omission, this time on the cost side of the equation. When sequestered in soil, carbon is physically stored within humus. Along with carbon, humus contains large amounts of nitrogen, phosphorus and sulphur in its chemical structure. Without these elements, humus cannot form (Kirkby et al., 2014). The elements must be sourced from either the background nutrients contained in the soil, or fertiliser (Richardson et al., 2014); the former has an opportunity cost, the latter an outright cost (plus there are the emissions associated with the manufacture of fertiliser). If the elements required to build humus are all supplied from fertiliser, then their cost would be in the range of $60 –$75 per tonne of CO_2 sequestered (Kirkby et al., 2011; Lam et al., 2013).

8.2.2 Climate impacts

Essentially there are four main sources of uncertainty when projecting future changes in climate: (i) there is doubt about the trajectory of global emissions, particular the further into the future one projects; (ii) there is uncertainty about the relationship between emissions of greenhouse gases and their subsequent atmospheric concentrations; (iii) for a given atmospheric concentration there is uncertainty about exactly how climate system will respond, and; (iv) there is ambiguity about how these changes will then manifest themselves at the regional level (Hennessy et al., 2008; Hennessy et al., 2010; Hope et al., 2015). In light of this uncertainty, a very broad range of future climate scenarios were defined when analysing the impact of climate change. A more limited set of scenarios were then selected and evaluated in more detail. An alternative approach, especially if only a limited set of scenarios are to be considered, is to statistically downscale the output of a Global Climate Model (GCM) or set of GCMs (ideally as many GCMs as possible—Burke et al., 2015) when run for a select set of future emissions trajectories.

An added advantage of this approach is that it would facilitate consideration of different-sized changes to the climate at different times of the year (a limitation of the approach in this thesis was that changes in climate were assumed to apply equally throughout the year). The within-year distribution of changes in climate is undoubtedly an important consideration: that agricultural production in the Wheatbelt is more sensitive to the weather at certain parts of the growing season is well established (Stephens and Lyons, 1998). The effect of timing can be quite nuanced—a change in monthly rainfall's effect can differ considerably depending on whether that change is experienced in July or August (Ludwig et al., 2009). However, whilst results are likely to be quite sensitive to timing, it might be questioned whether the precision embodied in them is realistic. Climatically, the most critical months are packed in to the coolest five months of the year (May –September). Across this shorter, concentrated period, changes in climate may be harder to delineate reliably in projections. That is, it may be easier to separate out how the extent of drying in 2050 may differ between two months if those months are July and January (i.e., summer and winter rainfall), than if they are July and August.

Changes in the frequency of extreme weather events were not considered, but are potentially important: the occurrence of even relatively short periods of frost during anthesis or desiccating events during grain-fill can greatly reduce crop yields (Zheng et al., 2012). However, for the short-term at least, research on the economic impacts of changes in the occurrence of extreme weather events may prove difficult. Changes in the intensity and frequency of extreme events are generally not well predicted, particularly for events like frost that are often affected by land cover and topography, localised factors that not well represented at the resolution of GCMs (Weeks et al., 2010; Zheng et al., 2012; Crimp et al., 2013; Andrys et al., 2015). To compound this, even if the occurrence of extremes was well predicted (or even if changes in their occurrence was simply assumed), the capacity to model the yield impact of spring frost and heat shocks is currently limited in most crop models, including the overwhelmingly preferred model in Australia, APSIM (Barlow et al., 2015).

In regards to simulating the biophysical impacts of climate change more generally, APSIM, GrassGro and 3PG are the best-tested and calibrated models for simulating crop, pasture and tree growth respectively, in the study region. Nonetheless, limitations were encountered with them, particularly as the farming-system nature of the analysis required them to perform for a relatively wide range of enterprises (e.g., crop types). In some cases the capacity of the model is under-developed, in other cases even if the will was there to improve the model, the underlying data to do so is missing. For instance, APSIM's lupin-crop module is considered to suffer from having been parameterised on too narrow a dataset, without enough diversity of soil types and enough high-yielding situations (B. Bowden and M. Robertson, pers. coms.). Also, data on the response of lupin to elevated CO_2 is simply lacking; in this study assumed responses were based on the responses of other crops for which there is evidence. As these simulation models are popular, progressive improvements to them are likely with time.

The MIDAS model endogenously adapted the farming system to climate change, using the options available and specified in the model. Adaptations included changes in land use, rotations, input use, labour, and livestock management strategies. Whilst these include many of the strategies typically available in the study area, to enumerate every possible adaptation option that is currently known and available within a modelling framework like MIDAS is impractical. A solution is to instead investigate climate change impacts through the statistical analysis of cross-sectional data, using hedonics; an early, seminal example of this approach is Mendelsohn et al. (1994). Theoretically, adaptation with any option currently practiced is implicit and endogenous in the results of such statistical studies.

A downside of statistical approaches is that they typically do not allow segregation of the impact of climate change and the impact of adaptative responses in their results (Antle and Capalbo, 2010), meaning they do not permit the benefits of adaptation to be valued (as was done in Chapter 6). Also, with future elevated CO_2 levels not represented in the cross-sectional data, future CO_2 ‘fertilisation’ is not accounted for (Lewandrowski and Schimmelpfennig, 1999). Modelling approaches have been devised to overcome these limitations (e.g., Antle et al., 2004). Whilst the approach used in this thesis has limitations, it is still a significant improvement on most previous climate change analyses of the Wheatbelt region that have not allowed for any adaptation at all.

The benefits of reduced weed, pest and disease pressure conferred by land-use rotation were assumed to stay the same in relative terms for all climate scenarios (i.e., no changes in weed, pest and disease pressure under different climate scenarios were assumed). Whilst such changes are probable, the nature of future climate-pest/disease/weed interactions is not well understood, with both negative and positive outcomes possible (Juroszek and von Tiedemann, 2013). The potential to adaptively respond to any emerging threats (e.g., by breeding disease resistance into crop varieties) is also uncertain. If improved information about future changes in weed, pest and disease pressure became available, it would be comparatively easy to incorporate into MIDAS.

This thesis used a comparative static approach, in which the climate changed from one steady-state (the base-case scenario) to another (climate scenario). Put differently, the impact of a *changed*, rather than *changing* climate was investigated. Thus no consideration was given to the dynamic aspects of transitioning from one climate state to another. This is a limitation because it hides the effect of uncertainty about the nature of changes to climate itself. In this thesis (and indeed many analyses of climate change in the literature too—Burke et al., 2015), changes to the climate are treated as ‘known’ and certain, in that they are specified with exactness in each climate scenario. In reality, farmers face the challenge of having to see through inevitable and on-going seasonal and cyclic variability to correctly identify and interpret changing climate trends as they occur. Adaptation hastily undertaken in response to what turns out to be a random fluctuation may be costly, but potentially so too is perseverance in the face of permanent change. Either type of mistake may erode a farm businesses’ underlying financial position, reducing their ability to adapt in the future. Another way of conceptualising this is to think of climate change as reducing the quality of the information available to investors, especially when making long-term decisions (Quiggin and Horowitz, 2003).

When climate change is modelled as a change from one climate steady-state to another, the costs of actually adapting (e.g., changing infrastructure and farm machinery) are also not captured. There could also be learning costs, for instance, from having to change production packages (like adopting a new crop type) or the adoption of new adaptation technologies. The extent of these costs is likely to be affected by the speed at which the climate changes (Quiggin and Horowitz, 2003). Slower rates-of-change afford more time for information diffusion, gradual uptake, and the incorporation of infrastructure change into the normal cycle of investment and replacement. It also affords more time for the research and development of adaptation options.

Lastly, logistical and supply-chain impacts of climate change were also not considered. These could include the increased logistical challenge of supplying drinking water to livestock if conditions are drier. As well as putting farm animals under more frequent thermal stress, warmer temperatures could have animal welfare implications for livestock transport (Howden et al., 2008). High summer temperatures can prevent or slow the movement of grain on freight trains (OTSI, undated). If stored grain becomes too warm, grain quality can be reduced and damage by storage pests also becomes more likely (GRDC, 2014). Changes in production volumes may necessitate the relocation of grain handling facilities and/or lead to the inefficient utilisation of existing facilities (Quiggin and Horowitz, 2003; Antle and Capalbo, 2010). The cost of these supply chain issues will ultimately be borne by farmers.

8.2.3 General limitations and further research

This section discusses issues which, broadly speaking, were limitations in both the analysis of mitigation options and the analysis of climate impacts.

The MIDAS model formed the backbone of several analyses in this thesis. MIDAS is a static, deterministic model, based on a single year with average weather (for the analyses conducted in this thesis, it was parameterised based-on average prices over the short-medium term). It therefore does not allow thorough consideration of variability and uncertainty, nor the dynamic aspects of adaptation (as discussed above) or adjustment, such as within-season tactical changes in management, and how this can change the make-up of the optimal farming system (Kingwell et al., 1992).

As a consequence of MIDAS's structure, only shifts in average climate could be considered and not changes in climate variability. Droughts, dry years and heatwaves are all predicted to become more frequent and intense in the south west of Australia in the future. However, this increase in exceptional weather may not necessarily imply an increase in climate variability for the study region *per se*. Under a warming and drying trend (a 'shift in the distribution curve'), an increase in the frequency and intensity of events that *currently* sit at the 'tails of the curve' is inevitable (Donat and Alexander, 2012).

Nonetheless, the failure to account for seasonal variability, even with current levels of variance, is a limitation. It is, for example, potentially a deficiency when investigating the attractiveness to farmers of participating in sequestration programs. If payments for sequestration can provide a source of income that is independent of, and decoupled from season type, then sequestration could provide an attractively safe source of income diversity for risk-averse farmers (e.g., Parks, 1995). Farmers in the study region do tend to display risk-averse preferences (Bardsley and Harris, 1987; Abadi Ghadim and Pannell, 2003), but the degree to which sequestration payments help manage seasonal variability may depend on the design and operation of the program (e.g., depending on if payments are based on dynamic or constant average rates of sequestration—see Chapter 4). Options for including seasonal variability in future analysis include, amongst others, multi-year cashflow analysis, and stochastic programing, possibly with Monte Carlo simulation (e.g., Kingwell et al., 1992; Kingwell, 1994; Kandulu et al., 2012; Scott et al., 2013). Unfortunately, in its current format, MIDAS does not readily lend itself to modification to enable the research to be extended in this direction (it is written in Excel and uses Visual Basic for Applications code to call an external solver). If, in the future, a version of MIDAS was developed in a more flexible environment, like General Algebraic Modelling Systems, then research in this direction should be more feasible, although the complexity of the model would continue to present a challenge.

Similar techniques could be used to represent variability in commodity prices in future research. It should also be remembered that in terms of investigating the cost-effectiveness of abatement options, the non-consideration of price and seasonal variability is not a limitation to the extent that the overall attractiveness of an option may differ in the results of future research that did not have these limitations. Rather it is a limitation only to the extent that their inclusion would change ability of a carbon price to incentivise *additional* uptake of the abatement option.

Instabilities in supply and demand for agricultural commodities caused by climate change, mitigation policy or both were regarded as being beyond the scope of this thesis and not considered explicitly and endogenously. However, Chapter 6 did show how the financial impact of climate change could be neutralised by relatively minor movements in prices. That result suggests that climate-induced fluctuations in the global balance between supply/demand could have a large bearing on how agriculture in the study area is ultimately affected by climate change and, therefore, accordingly, the omission of potential price changes from the analyses in this thesis is an important caveat.

The series of related studies investigating future land-uses in Australian by Bryan et al. (2014), Bryan et al. (2016a), Bryan et al. (2016b), Connor et al. (2015), and Grundy et al. (2016) (herein collectively termed ‘Bryan et al.’) provide an example of how the work in this thesis could possibly be extended to explicitly account for future changes in commodity prices. Bryan et al. conducted an integrated assessment that utilised equilibrium modelling (amongst other techniques) to project the future demand and supply for agricultural commodities, taking into account the effects of global population and economic growth, different scenarios for mitigation policy (corresponding to different levels of global action on climate change), the likely changes to climate these scenarios might entail, and how, on a spatial basis across the Australian continent, the changes in climate may in turn affect the productivity of agriculture (including an allowance for productivity growth). Economic components of their framework could be used to inform assumptions about possible future input and output prices for MIDAS. The projected change in the study region's climate associated with these price situations could also be 'mined' from their analysis framework and used to provide inputs into the APSIM, GrassGro, and 3PG biophysical simulation models. From there, the analyses in the Chapter 6 and Chapter 7 could be repeated.

A more complex approach might involve including representative farm models like MIDAS endogenously into Bryan et al.'s modelling framework. This has been attempted for other study regions (e.g., Europe by Wolf et al., 2015), but would be made difficult by—apart from anything else—the lack of a suitable whole-farm model/s calibrated for use right across Australia (directly-coupling just the version of MIDAS for the central Wheatbelt used in this thesis would be of little advantage). A benefit of even the simple 'data' mining approach would be the additional/complimentary insights that might be gained as a result of the differing approach used in this thesis, compared to the Bryan et al. studies. This would include the more direct employment of biophysical simulation models (including capturing the effect of CO_2 fertilisation) and whole-farm analysis of a farm business that captures interactions between different elements of the farming system, and allows it to autonomously adapt (with existing, specified options) to the new price, policy and climatic conditions.

Integrated analyses of the potential effect of future climate and economic forces on agricultural production and commodity prices at a global-level have been notable for their contradictory results (Müller and Robertson, 2014; Nelson et al., 2014a; Nelson et al., 2014b; von Lampe et al., 2014; Wiebe et al., 2015). This is not surprising, for such exercises are fundamentally speculative. Reasons for differing results are numerous and include different predictions of the extent of climate change (not simply choice of

GCM/s, but all four of the factors contributing to uncertainty about climate change listed at the beginning of Section 8.2.2) and its impact on food and fibre production (including availability of irrigation water and oceanic food sources); and different assumptions about adaptation/productivity growth, population and economic growth, and substitutability (e.g., will supply and demand be more inelastic in the short term than long term?). Because of this, if any future research were to be undertaken, it would be prudent, for interpretation and comparison purposes, to attempt to dissect the influence of these different factors.

Of course, future changes to commodity prices could also affect the cost-effectiveness of agriculture-based climate mitigation—a factor not dealt with in the investigation of climate change and mitigation policy interactions in Chapter 7. Prices of soft commodities could increase either due to the impact of climate change, and/or 'leakage'-type price feedback (Meyfroidt et al., 2013), in response to changes in agricultural output brought about by the implementation of mitigation policies (for example, due to the reforestation of farmland, reduced livestock production due to the imposition of a carbon price, etc.). Connor et al. (2015) investigated the potential importance of accounting for such price feedback (importance of feedback caused by just the reforestation of agricultural land in Australia, and not of similar action internationally too). At an aggregated national-level, these feedbacks did not drastically change the attractiveness of different land uses in their analysis. But at a more local level, in regions and scenarios where sequestering and agricultural land uses were of nearly equal attractiveness, price feedbacks had a more significant impact on the apparent attractiveness of different land uses. Whilst this suggests that the failure to consider changes in commodity prices when evaluating sequestration is a limitation of this thesis, any increase in commodity prices would only serve to strengthen the findings in Chapter 7.

Lastly, the analyses in this thesis are based on the central area of the Wheatbelt region, around the township of Cunderdin. As explained in the introductory chapter, this area's climate represents the approximate 'mid-point' of the range of temperatures and rainfall currently experienced across the region. However, any analysis based on a typical, 'representative' farm has the potential to produce misleading results for farms with different resources or characteristics. Therefore, time and budget permitting (representative farm models like MIDAS are very data intensive), ideally at least two more locations would be investigated: a location with higher rainfall and cooler temperatures, to the south and west of the present Cunderdin study area, and another with drier, warmer conditions, to the east and north. This would allow more nuanced differences in the impact of climate change and mitigation policy to be explored. For instance, livestock production tends be more prevalent in farming-systems on the southern and western fringes of the Wheatbelt and, accordingly, livestock emissions are much higher, and a carbon price on on-farm emissions may have a greater impact on these farming systems (Flugge and Schilizzi, 2005). Conversely, given the colder and moister nature of the environment in these locations, the potential for upsides from changes to climate are greater: warming may enhance growth during the cooler winter months whilst drying could also reduce waterlogging during the winter (Ludwig and Asseng, 2006).

Appendix: Chapter 9. Paper 7. Does growing grain legumes or applying lime cost effectively lower greenhouse gas emissions from wheat production in a semi-arid climate?

This paper has been published as:

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The candidate's overall contribution to the published paper was approximately 20%, as certified in the Statement of Student Contribution.

9.1 Preface

The paper presented in this Appendix Chapter was also completed during the Candidate's doctoral studies. Though its topic is again climate change and agriculture, specifically the emissions associated with agricultural production, it is presented as an appendix for two reasons. Firstly, the Candidate is not the leading author of this paper. Secondly—unlike the paper presented in Chapter 3 where the Candidate was also a co-author—this paper did not fit as well with the other papers presented in the main body of work, and it did not address the question of climate change's effect (both policy and biophysical impacts) on the study area.

The location of the study was the Wheatbelt township of Wongan Hills (shown in Figure 1.1, on page 3).

Does growing grain legumes or applying lime cost effectively lower greenhouse gas emissions from wheat production in a semi-arid climate?

9.2 Abstract

Agriculture production contributes to global warming directly via the release of carbon dioxide (CO_2), methane and nitrous oxide emissions, and indirectly through the consumption of inputs such as fertilizer, fuel and herbicides. We investigated if including a grain legume (*Lupinus angustifolius*) in a cropping rotation, and/or applying agricultural lime to increase the pH of an acidic soil, decreased greenhouse gas (GHG) emissions from wheat production in a semi-arid environment by conducting a streamlined life cycle assessment analysis that utilized *in situ* GHG emission measurements, rather than international default values. We also assessed the economic viability of each GHG mitigation strategy. Incorporating a grain legume in a two year cropping rotation decreased GHG emissions from wheat production by 56% on a per hectare basis, and 35% on a per tonne of wheat basis, primarily by lowering nitrogen fertilizer inputs. However, a large incentive ($93 per tonne of carbon dioxide equivalents reduced) was required for the inclusion of grain legumes to be financially attractive. Applying lime was profitable but increased GHG emissions by varying amounts depending upon whether the lime was assumed to dissolve over one, five or 10 years. We recommend further investigating the impact of liming on both CO_2 and non-CO_2 emissions to accurately account for its effect on GHG emissions from agricultural production.

Keywords: agriculture; economic analysis; grain production; greenhouse gas emissions; nitrous oxide; streamline life cycle assessment.

9.3 Introduction

Semi-arid and arid regions represent one third of the global land area and are widely used for grain production (Harrison and Pearce, 2000). Developing strategies for minimizing greenhouse gas (GHG) emissions from these regions is therefore important if global emissions from agriculture are to be lowered. Agriculture production contributes to global warming directly via the release of carbon dioxide (CO_2), methane

(CH_4) and nitrous oxide (N_2O) from soil, and indirectly through its demand for inputs such as fuel and fertilizer (Robertson and Grace, 2004; Smith et al., 2008; Smith et al., 2012). Furthermore, GHG emissions from agriculture are predicted to increase as the world's population continues to grow and the demand for meat and grain increases (Smith et al., 2007). Development and deployment of economically viable mitigation practices that decrease GHG emissions from agriculture is therefore essential. The development of strategies for decreasing GHG emissions from agricultural soils in semi-arid regions has received limited attention, with the limited analysis that has occurred, relying on hypothetical rather than regionally-specific field data (Engelbrecht et al., 2013).

Nitrogen (N) fertilizer production and its application to land contributes significantly to agricultural GHG emissions (Robertson et al., 2000; Gasol et al., 2007; Biswas et al., 2008). The Haber–Bosch process for producing synthetic N fertilizer results in 0.375 mole of CO_2 per mole of N produced (Schlesinger, 1999); while its subsequent application to crops and pastures enhances soil N_2O emissions via microbial activity (Firestone and Davidson, 1989) and CO_2 emissions from hydrolysis when N fertilizer is applied as urea (Eggleston et al., 2006). Increased use of synthetic N fertilizer since the industrial revolution has increased atmospheric N_2O concentrations from 271 ppbv to in excess of 320 ppbv (Solomon et al., 2007). Decreasing GHG emissions from the production and use of synthetic N fertilizer therefore has the potential to significantly lower the contribution of agriculture to global warming.

Incorporating grain legumes into cropping rotations can lower synthetic N requirements and may decrease GHG emissions from agriculture. Conservative estimates indicate 50 to 70 Tg N per year is fixed biologically in agricultural systems, despite the progressive replacement of legume rotations with synthetic N fertilizers over the past four decades (Smil, 2001; Crews and Peoples, 2004; Herridge et al., 2008). Whilst it has been suggested that including grain legumes in crop rotation may increase the risk of soil N_2O emissions, this is typically not the case (Jensen et al., 2011). Rather global and regional analyses indicate replacing a portion of cereal crops with legumes is likely to lower GHG emissions from crop production, although these calculations largely utilize international default values for estimating soil GHG emissions derived from temperate climates (e.g., Lemke et al., 2007; Nemecek et al., 2008; Jensen et al., 2011; Eady et al., 2012; Engelbrecht et al., 2013). Indeed, the discussion of the effects of crop rotation on

GHG emissions, and the use of site-specific emission data, is inadequate (Kendall and Chang, 2009). A streamlined life cycle assessment (SLCA) of GHG emissions, which accounts for emissions across production stages and utilizes site specific, field-based measurements for a range of climates and soil types, is needed to fully assess the role of grain legumes in mitigating agricultural GHG emissions.

In addition to decreasing the use of synthetic N fertilizers, mitigating soil N_2O emissions resulting from the use of synthetic N fertilizers is also recommended as an approach to lowering GHG emissions from agricultural soils (Smith et al., 2008).Soil N_2O can be emitted in direct response to the N fertilizer application, via biological processes such as nitrification or denitrification, or indirectly via N leaching and runoff, as well as from ammonia (NH_3) volatilization (Eggleston et al., 2006). Most strategies for decreasing N_2O emissions from cropped soils focus on improving N fertilizer use efficiency by fine-tuning plant growth-limiting factors and improving the synchrony between plant N uptake and N supply from all sources (Cassman et al., 2002; Ladha et al., 2005). These approaches, however, are unlikely to be effective at mitigating N_2O emissions that do not occur in direct response to N fertilizer applications. For example, a significant proportion of N_2O emissions from semi-arid agricultural soils can occur post-harvest, when the soil is fallow, and in response to summer-autumn rainfall (Barton et al., 2008; Galbally et al., 2008). Increasing soil pH, by applying agricultural lime ($CaCO_3$, herein referred to as 'lime'), may be one approach to decreasing N_2O emitted in semi-arid environments in response to summer rainfall events (Page et al., 2009; Barton et al., 2013a; Barton et al., 2013b). However, liming will only decrease total GHG emissions from these agricultural production systems if mitigated N_2O emissions are greater than the CO_2 emissions resulting from the dissolution and transport of the lime. For example, the Intergovernmental Panel on Climate Change (IPCC) assumes that all of the carbonate contained in lime ($CaCO_3$) will be released as CO_2 within the first year of application (Eggleston et al., 2006).

The overall objective of this study was to investigate strategies for decreasing GHG emissions resulting from the use of N fertilizers in rain-fed cropping systems in a semi-arid region. Specifically we investigated if including lupin (a grain legume commonly grown the region) in the cropping rotation, or applying lime to increase soil pH, decreased the life cycle global warming potential of wheat produced in a semi-arid climate. This was achieved by incorporating locally derived field-based measurements of GHG emissions derived from a companion study (Barton et al., 2013b) into a life cycle assessment (LCA) analysis. The economic viability of each rotation was also assessed, and where necessary, the financial incentive required to lower emissions calculated.

9.4 Materials and methods

9.4.1 Study site and experimental design

The effect of incorporating a grain legume in a cropping rotation, and applying lime, on GHG emissions from the wheat production was investigated in south-western Australia. The field site was located at Wongan Hills (30° 89' S, 116° 72' E) on a free-draining sand (Typic Quartzipsamment; USDA, 1992), which has an average annual rainfall of 374 mm that mainly falls in winter (Commonwealth Bureau of Meteorology, www.bom.gov.au/climate/averages). The field study consisted of a randomized-block design: two cropping rotations (lupin-wheat, wheat-wheat) by two liming treatments (0, 3.5 t ha^{-1}) by three field plot replicates (Barton et al., 2013b). Lime sand was surface applied to the soil approximately 2.5 months (18 March 2009) before planting in Year 1 with the aim of achieving a soil pH > 6.0 so as to influence the biological processes responsible for N_2O emissions. In Year 1 (June 2009), plots were either seeded to lupin (for the lupin-wheat rotation) or to wheat (*Triticum aestivum* cv Carnamah; for the wheat-wheat rotation), with N fertilizer only applied to the wheat (75 kg N ha^{-1} as urea). The following year (Year 2; June 2010) all plots were planted to wheat with the amount of urea applied to the lupin-wheat rotation taking into account the residual N from the 2009 lupin crop (Barton et al., 2013b). Consequently in 2010, the lupin-wheat plots received 20 kg N ha^{-1} as urea, while the wheat-wheat plots received 50 kg N ha^{-1}. Additional chemical inputs were recorded, and were typical of local farming practices. Each year the crops were harvested in November and the yield recorded for each plot. Soil GHG emissions (N_2O and CH_4) were measured continuously (subdaily) from each plot throughout the two year study using an automated chamber system connected to a gas chromatograph located at the field site, providing very high resolution (temporal) data. For further details of the study site, including the measurement of *in situ* N_2O and CH_4 emissions see Barton et al. (2013b).

9.4.2 Streamlined LCA assessment of GHG emissions from each cropping rotation

9.4.2.1 Goal and scope

The goal of the LCA was to compare GHG emissions from a lupin-wheat rotation with that emitted from a wheat-wheat rotation; both with or without lime. This was achieved after establishing the functional unit, selecting system boundaries, determining data requirements for the life cycle inventory (LCI), and finally calculating the GHG emissions for each cropping rotation. The functional unit was: 1) one hectare of cropped land; or 2) the production and transportation of one tonne of wheat to the port. We adopted a streamlined LCA (SLCA) approach that considered cradle-to-port GHG emissions, but ignored activities after the port (Todd and Curran, 1999; Engelbrecht et al., 2013). Consequently, our research considered GHG emissions in terms of an LCA, but with a focus on one impact category only, i.e., climate change (Finkbeiner et al., 2011).

9.4.2.2 Life cycle inventory

A LCI was completed prior to conducting the SLCA and consisted of the inputs (e.g., fertilizers, herbicides) and outputs (e.g., CO_2, CH_4, and N_2O) from three life cycle stages: pre-farm, on-farm and post-farm. *Pre-farm* activities included farm machinery manufacture and the production, plus transport of chemicals and fertilizers to the study site at Wongan Hills, and were calculated on a per hectare basis for each year (see Supporting Information Table 1). Most of the pre-farm emissions were calculated using emission factors available from the Australian LCA database (RMIT, 2007), and emission factors not available in the Australian database were developed by gathering basic information from the local industries (e.g., CSBP, a local fertilizer company, provided energy consumption information for determining the GHG emission factor for super phosphate production). The GHG emissions from the manufacture of farm machinery were estimated using the USA input/output database (Suh, 2004), based on the value of the machinery, with allowances for exchange rates and inflation. The USA input/output database contains environmental emission data for the production of US$ 1 equivalent farm machinery. The current price of farm machinery was deflated to the 1998 price (in AUD) at 2.98% per year. Following this, the 1998 price of machinery in AUD/hectare was converted to1998 US$ by multiplying by 0.6. Once the machinery cost for one tonne of wheat production was determined in terms of 1998 US$, this value was then multiplied by the GHG emission factor of machinery production (kg CO_2e-

/US$). Greenhouse gas emissions from the transport of inputs to the study site were calculated using the Australian LCA database (RMIT, 2007). Various modes of transportation were used including shipping, rail and articulated trucks (30 tonne), with the tonnage of input transported from manufacturer to the farm recorded (tkm). Where sea transportation was used to transport inputs, a single sea journey on a tanker to the port closest to the manufacturer was assumed. The GHG emissions from the production of chemicals was calculated using the Australian LCA database (RMIT, 2007). Herbicides not included in this Australian LCA database were converted to glyphosate equivalents before calculating GHG emissions, while GHG emissions associated with fertilizers not included in the Australian LCA database (e.g., super phosphate, Macro Pro, Big Phos Mn) were calculated using information collected from local fertilizer manufacturers (CSBP). The emission factor for urea production includes CO_2 associated with energy used to produce urea, plus the fossil fuel derived CO_2 used to manufacture the urea (i.e., $2\ NH_3 + CO_2 \rightarrow H_2N\text{-}COONH_4$). The amount of CO_2 that is used to manufacture the urea is subsequently released when the fertilizer is applied to land it is therefore included in the on-farm GHG contribution (see below). Only the CO_2 associated with the energy used to produce urea is considered in the pre-farm data.

On-farm data included information associated with the planting, maintaining and harvesting the crop, plus soil GHG emissions (see Supporting Information Table 1). The GHG emissions from fuel consumed during farm machinery operation were calculated using the Australian LCA database (RMIT, 2007). Machinery usage was expressed in terms of the amount of litres of fuel per hectare of land utilizing machinery typical for the region (L hr^{-1} ha^{-1}; See Supporting Information Table 1). Fuel consumption was dependent on land area, machinery width and the number of times the machinery passed across the land. Only direct N_2O emissions and CH_4 emissions from soil were quantified at the experimental site (Barton et al., 2013b), with indirect N_2O emissions, and CO_2 emission from urea hydrolysis, estimated using the Intergovernmental Panel on Climate Change (IPCC) default values (Eggleston et al., 2006). Indirect emissions include the N_2O emissions from N leaching and runoff, as well as those from NH_3 volatilization. The N_2O emissions from N leaching were assumed to be zero as the ratio of mean annual evapotranspiration (Et) to annual precipitation (P) was >1 for the experimental site, and the IPCC methodology predicts leaching only occurs when Et/P is between 0.8 and 1. For NH_3 volatilization, the IPCC methodology assumes that 10% of N fertilizer applied will be emitted as NH_3 via volatilization thereafter a portion of

NH_3 will be converted to N_2O following its deposition to land (Eggleston et al., 2006). A conversion factor of 0.08% was used to calculate the proportion of deposited NH_3 released as N_2O in this study, as this value is consistent with the value used by Australia to estimate direct N_2O emissions from the application of N fertilizer to non-irrigated land. Carbon dioxide emissions from lime dissolution were calculated using three scenarios based on different dissolution periods:

Scenario I: Lime dissolved within one year of application. This scenario is consistent with the IPCC's recommended approach to calculating CO_2 emissions from lime dissolution (Eggleston et al., 2006).

Scenario II: Lime assumed to dissolve in five years. Consequently this scenario equates to two-fifths of the CO_2 emissions from Scenario I, as it only includes the first two years (current LCA timeframe) of the five year dissolution period in the LCA; and

Scenario III: Lime assumed to dissolve in 10 years, equating to one-fifth of the CO_2 emissions from Scenario I, as it only includes the first two years (current LCA timeframe) of the 10 year dissolution period in the LCA. This scenario was chosen as it is represents the regularity that growers would apply 3.5 t ha^{-1} of lime in the study region.

Post-farm emissions included grain storage (5.6 kg CO_2 per tonne of wheat) and also 19.2 kg CO_2 per tonne of wheat transported to port (Kwinana, Western Australia) with a 30 tonne truck (Biswas et al 2008; see Supporting Information, Table 9.6).

9.4.2.3 Calculating GHG emissions from each cropping rotation

Individual greenhouse gas (CO_2, N_2O, CH_4) emissions from each production stage were converted to CO_2-eq using established conversion factors (Eggleston et al., 2006). Greenhouse gas emissions (as CO_2-eq) were then calculated on either a *per hectare* basis or a *per tonne of wheat* basis for each cropping rotation (with or without lime). The annual CO_2-eq per hectare (kg CO_2-eq ha^{-1} yr^{-1}) was calculated by summing CO_2-eq from each year and then dividing by the number of study years (two).

Total GHG emissions per tonne of wheat (CO_2-eq per tonne wheat) were calculated differently for each cropping rotation. For the wheat-wheat rotation, CO_2-eq per tonne wheat was calculated by summing the CO_2-eq ha^{-1} for each year and then dividing by the total wheat yield (t ha^{-1}) for the two years. Calculating the CO_2-eq per tonne wheat for the lupin-wheat rotation was more complicated, requiring the allocation of emissions from lupin production to the wheat production. The approach adopted for this allocation (described in the subsequent paragraph) is broadly consistent with the approaches proposed for allocating the environment impact of applying N derived from animal and green manure to crop rotations (van Zeijts et al., 1999; Knudsen et al., 2014).

The lupin was included in the cropping rotation to decrease the synthetic N fertilizer applied to the subsequent wheat crop. However, as only a proportion of the N from the lupin is used by the subsequent wheat crop, only a proportion of the emissions from the lupin crop were allocated to the following wheat crop. This proportion or 'allocation factor' was calculated by dividing the total amount of fertilizer avoided (i.e., saved) by the amount of N contained in the lupin crop (above- and below-ground):

$$\text{Allocation factor} = \frac{Nfertsaved}{LupinN_{AG} + LupinN_{BG}} \qquad \textbf{Eq 9.1}$$

where *Nfertsaved* is the amount of N fertilizer saved by growing the lupin (30 kg N ha^{-1}), $LupinN_{AG}$ is the amount of N contained in the above-ground biomass of the lupin crop (kg N ha^{-1}), and $LupinN_{BG}$ amount of N contained in the below-ground biomass (kg N ha^{-1}). The total of $LupinN_{AG}$ plus $LupinN_{BG}$ varied from 199 to 241 kg N ha^{-1} depending on liming treatment (Unkovich et al., 2009; Barton et al., 2013b), meaning the allocation factor ranged from 12 to 15%. Therefore the CO_2-eq per tonne wheat for the lupin-wheat rotation was calculated by summing 12–15% of the GHG emitted from the lupin crop production (2009–2010) with the GHG emissions from wheat production in the second year of crop rotation (2010–2011), and then dividing this summed value with the wheat yield from the second year of the rotation (i.e., 2010–2011).

9.4.3 Economic analysis of each cropping rotation

A budgeting analysis was conducted to determine the economic viability of each rotation on a per hectare basis (\$ ha^{-1} yr^{-1}), and if necessary, the incentive required to make a lower emitting rotation financially attractive for grain producers. To assess the economic viability of each rotation, the costs of inputs from the LCI was calculated, and the financial return from the grain yield determined. With the exception of grain prices (which were based on the average real farm-gate prices between 2007 and 2011), all prices were sourced from local suppliers. Machinery costs included an allowance for depreciation, labor, repairs and maintenance. Indirect, fixed production costs like land taxes were omitted as these would be identical for all rotations. Grain growers typically apply lime intermittently, consequently the net present value of the costs and benefits of lime and its application were annualized assuming a realistic commercial discount rate of 7%, and reapplication every 10 years; this timeframe is considered to be conservative as research in the study area has found applying lime at 2.5 t ha^{-1} continued to increase wheat yield by 25% up to 15 years later (Tang et al., 2003). The cropping rotations were treated as discrete options for two specific years with the (undiscounted) net returns of the rotations averaged across the two years. All monetary values are presented in Australian dollars ($AUD). Where a rotation caused fewer emissions, but had lower profitability, the minimum amount of money farmers would have to receive for it to be financially attractive to change to the lower emitting rotation (expressed in terms of $ per tonne of reduction in CO_2-eq emissions) was determined. These incentive payments were only calculated using per hectare emissions because the financial attractiveness of a rotation depends on the net profit from the entire cropping sequence.

9.4.4 Statistical analysis

A statistical analysis was conducted to assess if CO_2-eq emitted for each stage of wheat production was significantly affected by either cropping rotation or the application of lime. All data were statistically analyzed using a general linear model (completely randomized design) (Genstat, 2009). Post-hoc pair-wise comparisons of means were made using least significant difference (LSD; 5% level). It was not possible to conduct the statistical analysis of CO_2-eq on a per hectare basis (except for on-farm N_2O and CH_4 emissions) as inputs did not vary between field replicates.

9.5 Results

Including a grain legume in the cropping rotation generally decreased GHG emissions on both a per hectare and per tonne of wheat basis, irrespective of the application of lime ($P<0.05$; Figure 9.1and Figure 9.2). However on a per tonne of wheat basis, GHG emissions did not differ between the two cropping rotations when lime was assumed to dissolve in five years ($P<0.05$; Figure 9.2b). Including a grain legume in the cropping rotation did not compromise wheat yield in the second year of the cropping rotation (see Supporting Information, Table 9.7).

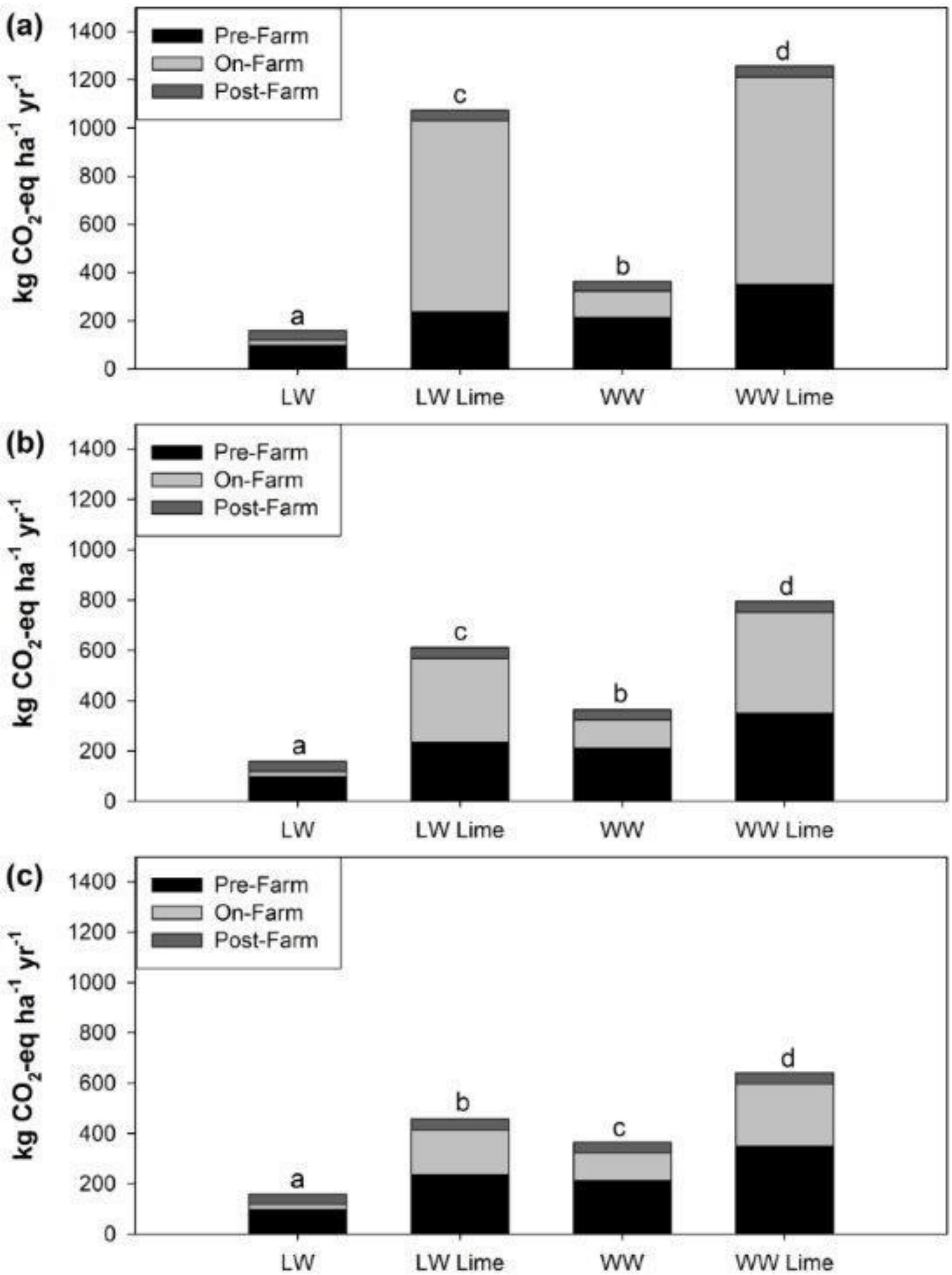

Figure 9.1. Life cycle assessment of greenhouse gas emissions produced per hectare of cropped land per year without lime, and when lime dissolves in (**a**) one year, (**b**) five years, and 10 years (**c**). Input data based on a lupin-wheat (LW) and wheat-wheat (WW) rotation at Wongan Hills, Australia (2009–2011). Columns in the same pane containing the same letter above them are not significantly differently at the 5% level.

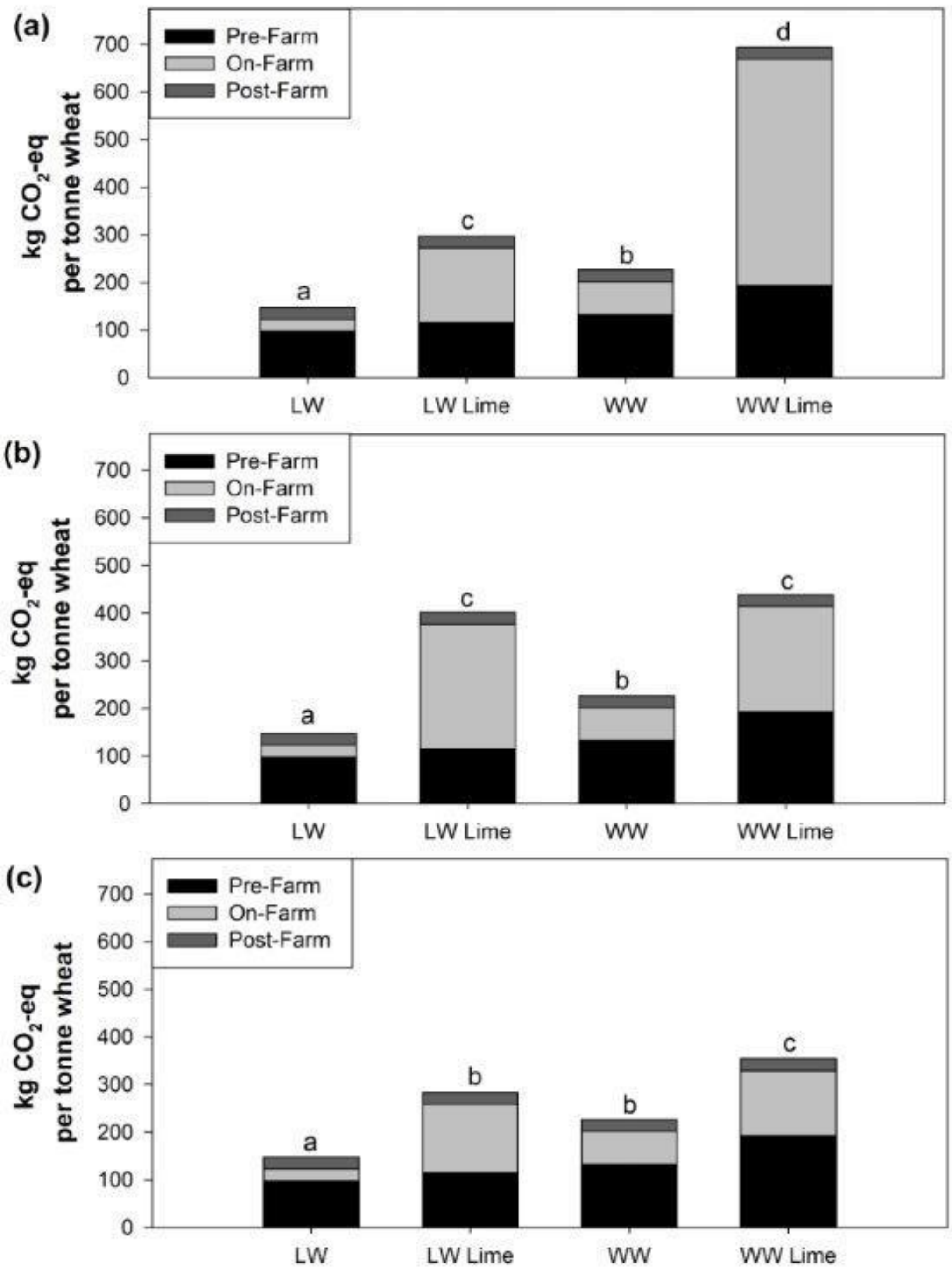

Figure 9.2. Life cycle assessment of greenhouse gas emissions produced per tonne of wheat without lime and when lime dissolves in (**a**) one year, (**b**) five years, and 10 years (**c**). Input data based on a lupin-wheat (LW) and wheat-wheat (WW) rotation at Wongan Hills, Australia (2009–2011). Columns in the same pane containing the same letter above them are not significantly different at the 5% level.

9.5.1 Effect of grain legume on cropping rotation GHG emissions in the absence of lime

On a per hectare basis, including a grain legume in the rotation decreased GHG emissions from 364 to 159 kg CO_2-eq ha^{-1} yr^{-1} when lime was not applied. The pre-farm stage contributed approximately 60% to total GHG emissions from both rotations (no lime; Figure 9.1); herbicide and fertilizer production was the greatest source of pre-farm emissions for the lupin-wheat and wheat-wheat rotation, respectively (Table 9.1). The on-farm stage represented 10% of the total GHG emissions from the lupin-wheat rotation, and 30% of the total emissions from the wheat-wheat rotation (no lime; Figure 9.1). Carbon dioxide emissions from urea dissolution was the greatest source of on-farm emissions for the wheat-wheat rotation (no lime), and were 9-times greater than from the lupin-wheat rotation (no lime; Table 9.2).

Table 9.1. Contribution of pre-farm inputs and outputs to greenhouse gas emissions (kg CO_2-eq per year) from one hectare of cropped land. Values are identical for all liming scenarios.

	Lupin-wheat	Lupin-wheat (lime)	Wheat-wheat	Wheat-wheat (lime)
N-fertilizer				
Production[†]	11.3	11.3	100.1	100.1
Transport	1.7	1.7	13.9	13.9
Lime				
Production	0.0	29.6	0.0	29.6
Transport	0.0	108.3	0.0	108.3
Herbicide production	61.0	61.0	73.5	73.5
Farm machinery production	17.6	17.7	17.7	17.9
Other inputs[††]				
Production	3.3	3.3	3.9	2.9
Transport	3.1	3.1	3.6	3.6

[†]Excludes CO_2 emissions from urea hydrolysis [††]Fungicides, oil, non N-fertilizers, pesticides, and rhizobium

Table 9.2. Contribution of on-farm inputs and outputs to greenhouse gas emissions (kg CO_2-eq per year) from one hectare of cropped land for all liming scenarios[†]. Values in the same row containing the same letter are not significantly different at the 5% level.

	Lime Scenario	Lupin-wheat	Lupin-wheat (lime)	Wheat-wheat	Wheat-wheat (lime)	$LSD_{0.05}$[†]
CO_2 from urea		9.4	9.4	86.4	86.4	NA[§]
CO_2 from lime	I	0.0	770.0	0.0	770.0	NA
	II	0.0	308.0	0.0	308.0	
	III	0.0	154.0	0.0	154.0	
Soil N_2O emissions		22.2^{ab}	24.1^{b}	28.2^{b}	16.4^{a}	6.5
Indirect N_2O emissions		0.2	0.2	2.0	2.0	NA
Soil CH_4 emissions		-16.5^{ab}	-15.7^{ab}	-11.8^{ab}	-18.6^{a}	5.6
Farm machinery use		5.7	6.4	6.0	6.7	NA

[†]Scenario I, lime dissolves in one year; Scenario II, lime dissolves in five years; Scenario III, lime dissolves in 10 years
[††]LSD, least significant difference [§]NA, not applicable

On a per tonne of wheat basis, including a grain legume in the cropping rotation, decreased total GHG emissions from 227 to 148 kg CO_2-eq per tonne of wheat when lime was not applied (P<0.05; Figure 9.2). The pre-farm stage represented 58% (wheat-wheat, no lime) to 66% (lupin-wheat, no lime) of total GHG emissions, whereas the on-farm stage contributed 17% (lupin-wheat, no lime) to 30% (wheat-wheat, no lime; Figure 9.2). Herbicide or fertilizer production mostly contributed to pre-farm emissions (Table 9.3), while soil N_2O and CO_2 emissions from the application of urea to land were the main sources of on-farm emissions (Table 9.4).

Table 9.3. Contribution of pre-farm inputs and outputs to greenhouse gas emissions (kg CO_2-eq) from the production and transport of one tonne of wheat to port. Values are identical for all liming scenarios. Values in the same row containing the same letter are not significantly different at the 5% level.

	Lupin-wheat	**Lupin-wheat (lime)**	**Wheat-wheat**	**Wheat-wheat (lime)**	**$LSD_{0.05}$†**
N-fertilizer					
Production††	16.1^{a}	15.3^{a}	62.5^{b}	55.2^{b}	9.3
Transport	2.4^{a}	2.3^{a}	8.7^{b}	7.7^{b}	1.3
Lime					
Production	0.0^{a}	5.0^{b}	0.0^{a}	16.3^{c}	0.7
Transport	0.0^{a}	18.4^{b}	0.0^{a}	59.8^{c}	2.7
Herbicide production	59.3^{c}	55.8^{bc}	45.9^{ab}	40.5^{a}	10.4
Farm machinery production	14.5^{c}	13.5^{bc}	11.1^{ab}	9.9^{a}	2.5
Other inputs§					
Production	2.5^{b}	2.3^{b}	2.4^{b}	1.6^{a}	0.5
Transport	2.8^{c}	2.6^{bc}	2.2^{ab}	2.0^{a}	0.5

†LSD, least significant difference ††Excludes CO2 emissions from urea hydrolysis
§Fungicides, oil, non N-fertilizers, pesticides, and rhizobium

Table 9.4. Contribution of on-farm inputs and outputs to greenhouse gas emissions (kg CO_2-eq) from the production and transport of one tonne of wheat to port for all liming scenarios†. Values in the same row containing the same letter are not significantly different at the 5% level.

	Lime Scenario	**Lupin-wheat**	**Lupin-wheat (lime)**	**Wheat-wheat**	**Wheat-wheat (lime)**	**$LSD_{0.05}$††**
CO_2 from urea		13.4^{a}	12.8^{a}	53.9^{b}	47.7^{b}	8.0
CO_2 from lime	I	0.0^{a}	130.4^{b}	0.0^{a}	424.8^{c}	19.2
	II	0.0^{a}	235.4^{c}	0.0^{a}	169.9^{b}	29.1
	III	0.0^{a}	117.7^{c}	0.0^{a}	85.0^{b}	14.5
Soil N_2O emissions		21.4^{b}	20.9^{b}	17.6^{b}	9.0^{a}	6.7
Indirect N_2O emissions		0.3^{a}	0.3^{a}	1.3^{b}	1.1^{b}	0.2
Soil CH_4 emissions		-14.6^{a}	-12.2^{ab}	-7.3^{c}	-10.3^{bc}	3.4
Farm machinery use		4.9^{b}	4.7^{b}	3.8^{a}	3.7^{a}	0.9

†Scenario I, lime dissolves in one year; Scenario II, lime dissolves in five years; Scenario III, lime dissolves in 10 years
††LSD, least significant difference

9.5.2 Effect of liming on GHG emissions

On a per hectare basis, applying lime at least doubled GHG emissions from both rotations ($P<0.05$; Figure 9.1). Although the dissolution time of the lime did not alter pre-farm GHG emissions in absolute terms, it did alter the proportion of total emissions attributed to the pre-farm stage. For example, pre-farm emissions contributed up to 28% to total GHG emissions when lime was assumed to dissolve in one year, but increased to 55% when lime dissolved in 10 years (Figure 9.1); lime transport, fertilizer production (wheat-wheat only), and herbicide production were all major sources of pre-farm emissions (Table 9.1). The on-farm stage produced 70% of the total GHG emissions from both rotations when lime dissolved in one year, decreasing to approximately 40% when lime dissolved in 10 years (Figure 9.1). Irrespective of the dissolution rate, CO_2 emissions from lime dissolution were the greatest source of on-farm emissions for both rotations (Table 9.2). For example, under the assumption that lime dissolved in one year, CO_2 emissions from liming were almost 9-times greater than CO_2 emissions from urea hydrolysis (Table 9.2). Applying lime also decreased direct soil N_2O emissions from the wheat-wheat rotation ($P<0.05$; Table 9.2).

On a per tonne of wheat basis, applying lime at least doubled emissions from both rotations ($P<0.05$; Figure 9.2). Again while liming did not alter absolute pre-farm GHG emission, the proportion of total emissions attributed to this stage increased from approximately 40%, when lime was assumed to dissolve in one year, to up to 55% when lime dissolved in 10 years (Figure 9.2), due to lower CO_2 emissions from lime dissolution in the on-farm stage. Lime transport, fertilizer (wheat-wheat only) and herbicide production were the main source of pre-farm emissions (Table 9.3). Up to 70% of the total GHG emissions from both rotations were attributed to the on-farm stage when lime dissolved in one year, which decreased to approximately 50% when lime dissolved in 10 years (Figure 9.2). Lime dissolution was the greatest source of on-farm emissions for both rotations, even when it dissolved in 10 years (Table 9.4). Storage and transport of grain to port (i.e., post-farm emissions) contributed relatively little (<10%) to GHG emissions from the production of one tonne of wheat when lime was applied to both rotations.

9.5.3 Economic viability of cropping rotations

Initial analysis indicated that the lupin-wheat rotation was \$37 ha^{-1} yr^{-1} more profitable than the wheat-wheat rotation with lime, and \$58 ha^{-1} yr^{-1} without lime (see Supporting

Information, Table 9.8). However, wheat yield was unusually low relative to the lupin grain yield in Year 1 (2009 harvest) of the present study. Historical data for the area shows wheat yield to be 166% of lupin yield (by mass), and in 2009 averaged 143% on commercial farms in the present study district (Planfarm, 2010). At the present study site, wheat yield was 111% of the lupin yield in 2009; perhaps because wheat was also grown at the site for two consecutive years prior to the current study, limiting rotational benefits from sowing different crops (Seymour et al., 2012). Consequently, we reassessed the economic viability of each cropping rotation after scaling the 2009 wheat yields reported in this study so that they were 143% of lupin yield. Inputs and environmental conditions were unchanged from the original economic analysis, and it was assumed that GHG emissions from the soil would not differ as a result of the scaling. However, the higher yield increased grain handling and thus emissions per hectare (see Supporting Information, Table 9.8).

After scaling the wheat yield, the wheat-wheat rotations were more profitable than the lupin-wheat rotation. For example, without lime, wheat-wheat was \$20 ha^{-1} yr^{-1} more profitable than the lupin-wheat rotation (see Supporting Information, Table 9.8). At the same time, the wheat-wheat rotation would also emit 371 kg of CO_2-eq ha^{-1} yr^{-1}, which is 2.3 times more than lupin-wheat (see Supporting Information, Table 9.8). Therefore grain producers would require some form of pecuniary incentive to change rotations and realize emissions savings. An incentive equivalent to \$93 per every tonne of CO_2-eq decreased would be required to change from a wheat-wheat rotation to lupin-wheat rotation if lime was not applied (Table 9.5). If lime was applied, then the incentive would need to be \$256 t^{-1} CO_2-eq^{-1} (the time it takes lime to dissolve does not alter this incentive as the changes in emissions when lime dissolves over longer time frames affect both the lupin-wheat and wheat-wheat rotations identically).

Table 9.5. The minimum incentive required to make the lupin-wheat rotation more viable than the wheat-wheat rotation (after scaling 2009 wheat yields) as affected by input and output prices.

Scenario	Without Lime ($ t^{-1} CO_2-eq)	With Lime ($ t^{-1} CO_2-eq)
Standard input & output prices	93	256
Fertilizer & pesticide prices 10% higher	59	219
Fertilizer & pesticide prices 10% lower	127	294
Wheat prices 10% higher	246	456
Lupin prices 10% higher	−8†	129

†Negative value indicates no incentive required

9.6 Discussion

9.6.1 Grain legumes and GHG emissions from wheat production

Including a grain legume in a cropping rotation decreased total GHG emissions produced from rain-fed wheat grown in a semi-arid environment on both a per hectare and per tonne of wheat basis. Utilizing legume-fixed N in a two year cropping rotation decreased emissions from wheat production by 56% per hectare (e.g., 364 to 159 kg CO_2-eq ha^{-1} yr^{-1} when lime not applied), and by 35% per tonne of wheat (e.g., 227 to 148 kg CO_2-eq per tonne of wheat when lime not applied). This occurred as less N fertilizer was applied to the lupin-wheat than the wheat-wheat rotation, which subsequently decreased CO_2 emissions from fertilizer production and urea hydrolysis, and without additional soil N_2O emissions. Decreasing N fertilizer inputs to wheat production also decreased emissions from fertilizer transportation (pre-farm), and indirect soil N_2O emissions (on-farm). In the present study 227 kg CO_2-eq were produced per tonne of wheat when N was sourced from fertilizer and lime was not applied, which is comparable to a previous estimate (304 kg CO_2-eq per tonne of wheat) for the region (Biswas et al., 2008).

Our observations are also consistent with the general expectation that replacing a cereal crop with a legume crop, or substituting fertilizer N with, legume-fixed N will lower GHG emissions from crop production (Lemke et al., 2007; Nemecek et al., 2008; Jensen et al., 2011; Eady et al., 2012; Engelbrecht et al., 2013). However previous research has utilized IPCC default values rather than site or regional specific emission data, and has been largely conducted in more temperate climates that the present study. To our knowledge, this is the first GHG emission analysis that utilizes field-based emission data to quantify the effect of incorporating grain legumes in a cropping rotation on

GHG emissions from cereal grain production in a semi-arid environment. This is important because in semi-arid environments such as the study region, IPCC emission factors have been found to significantly over estimate emissions of N_2O from agricultural soil (Barton et al., 2008; Barton et al., 2010), and agricultural production is widespread in semi-arid regions.

Production, transport and application of N fertilizer, is the greatest source of GHG emission in wheat production in the present semi-arid region. For example in the current study, it contributed 231 kg CO_2-eq per ha, or 144 kg CO_2-eq per tonne of wheat (63% of total GHG emissions when a grain legume was not included in the rotation). This is comparable to a previous study in the same region where N fertilizer supply and use produced almost 190 kg CO_2-eq per tonne of wheat (62% of total GHG emissions; Biswas et al., 2008). Including a grain legume in the present study decreased the contribution from N fertilizer use from 231 to 45 kg CO_2-eq ha^{-1} yr^{-1}, or from 144 to 54 kg CO_2-eq per tonne of wheat. Others have also shown including perennial and annual grain legumes in cropping rotations lowered energy inputs, via decreased N fertilizer inputs, by up to 27% (Zentner et al., 2001; Hoeppner et al., 2006; Rathke et al., 2007). The extent to which incorporating a grain legume into a cropping rotation decreases energy inputs and GHG emissions from crop production will depend on how much N fertilizer is saved, which will in turn be determined by grain legume yield, the type of grain legume grown, and the regularity grain legumes are included in the rotation (Lemke et al., 2007; Peoples et al., 2009).

Including a grain legume in a two year cropping rotation for the purpose of decreasing GHG emissions would require a large incentive (per tonne of emission saved) to be financially attractive to grain producers in the study region. Despite requiring less expenditure on N fertilizer, the lupin-wheat rotation was still less profitable than wheat-wheat because both the yield of grain per ha, and the price per tonne of this grain, was lower when lupin grain was produced instead of wheat grain. Therefore an opportunity cost would be incurred by growing the lupin-wheat rotation. And so although the difference in emissions between the lupin-wheat and wheat-wheat rotations appears impressive, the absolute size of these emissions saving was small compared to this opportunity cost. For instance changing from wheat-wheat to a lupin-wheat rotation without lime would cause per hectare emissions to fall by 57% (mainly because of reduced emissions from N fertilizer production and use). However in absolute terms, this was a decrease of only 0.21 t CO_2-equ ha^{-1} yr^{-1} for a reduction in profit (i.e., opportunity cost) of \$20 ha^{-1} yr^{-1}, suggesting a financial incentive equivalent to \$93 t^{-1} CO_2-equ would be required to change from a wheat-wheat rotation to lupin-wheat. This is much larger than contemporary global carbon prices. However it should be noted that the financial incentive is sensitive to input costs (e.g., fertilizer and pesticides) and, in particular grain prices (Table 9.5); both of which do vary temporally. Had seasonal conditions in Year 2 (2010) of the study been more favorable, then it is possible that wheat yield would have responded more positively to inclusions of the grain legume in the rotation, lowering the incentive required to make the lupin-wheat rotation financially attractive.

The present study presents a simplified crop rotation so that field-based data (Barton et al., 2013b) could be incorporated in the analyses. Typically grain legumes are included in cropping rotations in the study region, but not every second year. Decreasing the frequency that grain legumes are grown (in comparison to the present study) would decrease the financial incentive required per tonne of emissions reduction to include a grain legume in the cropping rotation, although not necessarily by a large amount, as less frequent legumes would mean less tonnes of emissions reductions. Also, we have considered the financial performance of the rotations in isolation rather than as part of the entire farms operation (Pannell, 1995). The adoption of agricultural practices often depends on a broader range of technical, social, cultural, economic and personal factors, and not just financial attractiveness (Pannell et al., 2006). These limitations aside, the results of the economic analysis still provide a guide to the likely cost-effectiveness (and thus desirability) of pursuing the rotation change in question to decreasing GHG emissions on the study region.

9.6.2 Soil liming and the GHG emissions from grain production

Applying lime increased the profitability of grain production, but at the same time increased total GHG emissions on both a per hectare and per tonne of wheat basis in the present study. Similarly, soil liming increased GHG emissions from grains production from 304 to 466 kg CO_2-eq per tonne of wheat in a previous assessment in the same region as the present study (Biswas et al., 2008). However, the extent to which liming contributes to GHG emissions in the present study varied depending on the rate of lime dissolution (Figure 9.1 and Figure 9.2). Calculating the contribution of soil liming to CO_2 emissions, and specifically the validity of the IPCC default values, has been widely debated (West and McBride, 2005; Hamilton et al., 2007; Biasi et al., 2008). As previously mentioned, the IPCC guidelines for preparing national GHG inventories assumes that, in the absence of country-specific data, all of the carbonate contained in calcic limestone will be released as CO_2 within a year of application (Eggleston et al., 2006). However a review of the contribution of agricultural lime use to CO_2 emissions in the United States estimated only 49% of the applied carbonate was emitted as CO_2 (West and McBride, 2005). Further research clarifying the amount (and timing) of CO_2 emitted by lime dissolution is required. Given our SLCA results were very sensitive to the inclusion of soil liming, such research could have implications for calculating the carbon foot print of agricultural production, and national GHG inventories more generally, where the SLCA is sensitive to CO_2 emissions from liming.

The influence of liming on GHG emissions from agricultural production is often considered low in comparison to other emission sources (e.g., Kendall and Chang, 2009; Brock et al., 2012; Raucci et al., 2015), which is in direct contrast to findings in the present study. For example, Brock et al. (2012) reported a much lower contribution of liming to GHG emissions from wheat produced in south-eastern Australia than found in the current study. We attribute this to differences in lime application rates [i.e., 3500 kg ha^{-1} in present study versus 31.5 kg ha^{-1} in Brock et al. (2012)] and grain yield between the two studies, as both studies used the IPCC methodology (Eggleston et al., 2006) to estimate the CO_2 emissions from lime dissolution. We would argue that the contribution of liming to GHG emissions from agricultural systems will be influenced by amount of lime applied, the assumed dissolution rate, grain yield, and its contribution relative to other GHG emitting inputs (e.g., N fertilizers) and should therefore not be overlooked when conducting agricultural LCAs. In low grain-yielding environments, where N fertilizer inputs and N_2O emissions are minimized, and where large amounts of lime may be required to remediate soil acidity, the influence of liming on GHG emissions from grain production may be greater than temperate environments.

Emissions associated with the use of lime also need to be viewed in the context of total GHG emissions and soil carbon sequestration. For example, soil liming may partly offset other on-farm GHG emissions in rain-fed, agricultural soils in semi-arid region. In the companion study that provided the *in situ* soil N_2O and CH_4 emission data utilized in the present study, increasing soil pH (via liming) decreased cumulative N_2O emissions from the wheat-wheat rotation by 30% by lowering N_2O emissions following summer-autumn rainfall events, and increasing CH_4 uptake (Barton et al., 2013b). This observed phenomenon decreased the GHG emissions of wheat production in the present study by up to 19 kg CO_2-eq ha^{-1} yr^{-1} or 11 kg CO_2-eq per tonne of wheat, but was insufficient to offset the CO_2 emissions resulting from the transport and dissolution of lime (e.g., 292–910 kg CO_2-eq ha^{-1} yr^{-1}, or 141–501 kg CO_2-eq per tonne of wheat; Table 9.2 and Table 9.4). The dissolution of lime can also be a net sink for CO_2 in soil with relatively high pH, but a net source of CO_2 in acidic soils (West and McBride, 2005). However, avoiding liming to decrease GHG emissions risks other adverse environmental impacts like soil acidification.

9.6.3 Contribution of soil N_2O emissions

Several studies have demonstrated that indirect and direct N_2O emissions substantially increase the GHG emissions of agricultural production (Crutzen et al., 2008; Biswas et al., 2010; Popp et al., 2011). In contrast N_2O emissions were negligible in our study, generally contributing less than 10% to total emissions depending on the cropping rotation. This reflects the current understanding that soil N_2O emissions from rain-fed crops in semi-arid regions are very low in comparison to other soils and climates, and significantly less than that predicted using the IPCC emission factors (Barton et al., 2008; Barton et al., 2011). Although soil and agricultural scientists recognize that the proportion of N fertilizer converted to N_2O emissions varies significantly with soil type, climate and land management practices (Stehfest and Bouwman, 2006), this is not as widely recognized by LCA practitioners (Kendall and Chang, 2009). We therefore support recommendations to use regionally specific data when calculating GHG emissions and performing any associated economic analyses for agricultural production systems (Kendall and Chang, 2009; Thamo et al., 2013; Hörtenhuber et al., 2014) rather than IPCC default values (e.g., 1.0%) across all geographic and climatic regions. Furthermore in soils and climates conducive to N_2O emissions (or if IPCC default emission factors had been used in the present study), it should be recognized that the economic incentive required to induce emission-saving practice change may be smaller than in the present study.

Including grain legumes in cropping rotations is unlikely to increase GHG emissions of semi-arid agricultural systems as a result of increased soil N_2O emissions (Table 9.2 and Table 9.4). Our field-based research demonstrated that a growing a grain legume did not enhance soil N_2O emissions during either the growth of the grain legume, or during the subsequent wheat crop, when N fertilizer inputs were adjusted to account for residual N from the grain legume crop (Barton et al., 2011). Indeed, total N_2O losses were approximately 0.1 kg N_2O-N ha^{-1} after two years for both the lupin-wheat and wheat-wheat rotations (when averaged across liming treatment). Our observations is consistent with recent reviews of N_2O fluxes from various agro-eco-systems that have also concluded that there is a tendency for legume crops to emit similar, if not less, N_2O than fertilized non-legume crops (MacKenzie et al., 1998; Rochette et al., 2004; Helgason et al., 2005; Rochette and Janzen, 2005; Parkin and Kaspar, 2006; Dick et al., 2008).

9.6.4 **Impact of functional unit**

Expressing GHG emissions on both a per hectare or product (tonne of wheat) basis showed similar trends across treatments. This contrasts with some other agricultural systems (O'Brien et al., 2012). On one hand, expressing GHG emissions on an area basis directly reflects the total emissions likely to enter the atmosphere; on the other, expressing emissions on a product basis reflects the production efficiency (but only for that product, not the agricultural system as a whole). The latter is particularly relevant when considered in the context of increasing global production and associated demand for food. Expressing GHG emissions on a product basis, however can lead to perverse outcomes as a result of choices made when allocating emissions. For example, when we assumed that lime dissolved in five years rather than one, the GHG emissions per tonne of wheat actually increased for the lupin-wheat rotation due to the allocation process used to allocate emissions from the lupin crop to the following wheat crop. Consequently for the five year scenario, the GHG emitted per tonne of grain was the same for the lupin-wheat plus lime rotation as the wheat-wheat plus lime rotation; whereas GHG emitted per hectare were lower from the lupin-wheat plus lime than wheat-wheat plus lime. Furthermore, in low grain-yielding environments expressing GHG emissions per tonne can be misleading by indicating these environments are less efficient than higher yielding environments (Hörtenhuber et al., 2014). We recommend expressing GHG emissions on both per hectare and product (tonne of wheat) basis when using SLCA to assess the global warming potential of agricultural production.

9.7 Conclusions

Including a grain legume in a two-year cropping rotation lowered the GHG emissions of wheat production by lowering the need for synthetic N fertilizer without comprising grain yield, but required a large incentive (per tonne of emission saved) to be financially attractive. By contrast, applying lime to raise soil pH was profitable but increased total GHG emissions from wheat production by varying amounts depending on the time that lime was assumed to dissolve. Analysis of GHG emissions from agricultural production systems is sensitive to the inclusion of soil liming and further research is needed to fully understand the interaction between soil liming and GHG emissions if this common management practice is to be accurately accounted for by SLCA. We recommend expressing GHG emissions on both per hectare and per product (tonne wheat) basis when using SLCA to assess the global warming potential of agricultural production. Our findings demonstrate that while there are land management strategies available to lower GHG emissions from grain production in semi-arid climates, economic incentives may be required to encourage adoption.

9.8 Supporting Information

Table 9.6. Life cycle inventory for grain production in a lupin-wheat and wheat-wheat rotation (with or without lime) at Wongan Hills, Australia.

		Units	Lupin-wheat		Lupin-wheat (lime)		Wheat-wheat		Wheat-wheat (lime)	
			Year 1*	Year 2†	Year 1	Year 2	Year 1	Year 2	Year 1	Year 2
					Pre-farm					
Urea	Production	kg ha^{-1} yr^{-1}	-	26	-	26	146	90	146	90
	Transport	tkm	-	264	-	264	1492	922	1492	922
Other fertilizers	Production	kg ha^{-1} yr^{-1}	80	80	80	80	80	80	80	80
	Transport	tkm	15.4	15.4	15.4	15.4	15.4	15.4	15.4	15.4
Lime	Production	t ha^{-1} yr^{-1}	-	-	3.5	-	-	-	3.5	-
	Transport	tkm	-	-	2121	-	-	-	2121	-
Herbicides & fungicides	Production	L ha^{-1} yr^{-1}	4.2	7.4	4.2	7.4	5.4	7.4	5.4	7.4
	Transport	tkm	27	54.3	27	54.3	46.7	54.3	46.7	54.3
Rhizobium	Production	kg ha^{-1} yr^{-1}	9	-	9	-	-	-	-	-
	Transport	tkm	4.7	-	4.7	-	-	-	-	-
Farm machinery	Production	USD$ ha^{-1}	18.8	19.2	19.3	19.2	19.2	19.2	19.7	19.2
					On-farm					
N_2O emissions, direct		kg ha^{-1} yr^{-1}	0.06	0.09	0.07	0.09	0.08	0.1	0.05	0.06
N_2O emissions, indirect		kg ha^{-1} yr^{-1}	-	0.02	-	0.02	0.01	0.01	0.01	0.01
CH_4 emissions		kg ha^{-1} yr^{-1}	-0.44	-0.55	-0.45	-0.49	-0.25	-0.45	-0.46	-0.69
CO_2 emissions from urea		kg ha^{-1} yr^{-1}	-	19	-	19	107	66	107	66
CO_2 emissions from lime		kg ha^{-1} yr^{-1}	-	-	1,540	-	-	-	1,540	-
Farm machinery fuel		L ha^{-1} yr^{-1}	2.34	2.34	3.62	2.34	2.34	2.34	3.62	2.34
					Post-farm					
Storage		kWh ha^{-1}	0.010	0.008	0.012	0.008	0.011	0.007	0.013	0.008
Transportation to port		tkm ha^{-1}	336	268	382	279	380	231	419	263

*2009–2010 †2010–2011

Table 9.7. Grain yields (t ha^{-1}) for each cropping rotation (with or without lime) at Wongan Hills, Australia.

Year	Lupin-wheat	Lupin-wheat (lime)	Wheat-wheat	Wheat-wheat (lime)
Non-scaled yield				
Year 1 (2009)	1.79	2.03	2.02	2.23
Year 2 (2010)	1.43	1.49	1.23	1.40
Scaled yield				
Year 1 (2009)	1.79	2.03	2.59	2.86
Year 2 (2010)	1.43	1.49	1.23	1.40

Table 9.8. Profit and total GHG emissions before and after scaling Year 1 (2009) wheat yields.

	Lime Scenario	Lupin-wheat	Lupin-wheat (lime)	Wheat-wheat	Wheat-wheat (lime)
Non-scaled yield					
Profit ($ ha^{-1} yr^{-1})		97	116	39	79
GHG emissions (kg CO_2-eq ha^{-1} yr^{-1})	No lime	159	NA*	364	NA
	I		1074		1258
	II		612		796
	III		458		642
Scaled yield					
Profit ($ ha^{-1} yr^{-1})		97	116	117	165
GHG emissions (kg CO_2-eq ha^{-1} yr^{-1})	No lime	159	NA	371	NA
	I		1074		1266
	II		612		804
	III		458		650

*NA, not applicable

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