



# Achieving net negative emissions in a productive agricultural sector

A review of options for the Australian  
agricultural sector to contribute to the  
net-zero economy

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# Executive summary

Some Australian agricultural groups have goals to reduce greenhouse gas (GHG) emissions to net-zero by 2030. These goals are key for Australia to be consistent with the Paris Agreement and the Agreement's target of net-zero emissions by the second half of the century. Negative emissions provided by the land sectors are common components of pathways that restrict warming to 1.5 - 2 °C. To achieve these goals, emissions need to be reduced to levels that can be offset by land sinks. This will be challenging to achieve while increasing production to meet growing demand and as climate change impacts become more evident.

Direct agricultural emissions comprised 13.7% (73 Mt CO<sub>2</sub>eq) of Australia's emissions in 2017. In addition, deforestation contributed 26 Mt CO<sub>2</sub>eq, with about 75% due to land clearing for agriculture. Combined, this is a total of 93 Mt CO<sub>2</sub>eq attributable to agricultural activities. Including upstream emissions from electricity adds 8 Mt CO<sub>2</sub>eq, for a total of 101 Mt CO<sub>2</sub>eq agriculture-related emissions or 19% of national emissions.

The contribution of different agricultural sectors to total GHG emissions varies with the amount of production and the amount of emissions per unit product, or emissions intensity (EI). The red meat sector, with high production and high EI, emits the most with an estimated 68.6 Mt CO<sub>2</sub>eq emitted in 2015, including land use change. The emissions associated with non-ruminant meats is lower due to lower production and lower EI from fewer enteric methane emissions produced during digestion, quicker growth, and differences in production. In 2009, cropping accounted for 31.5% (22 Mt CO<sub>2</sub>eq) of all agricultural emissions. This is due to high production of plant products, particularly wheat, with small EIs compared to most meats. Dryland agricultural emissions are dominated by fertiliser manufacture and use, while irrigated crops also have high energy use.

The largest source of agricultural emissions is enteric methane, produced by ruminant animals such as cows. In 2017, 51.5 Mt CO<sub>2</sub>eq of enteric methane was emitted, with animals in extensive systems the major source. Currently, there are no options capable of substantially reducing these emissions. Methods available that can provide significant reductions (20-30%) in enteric methane, such as feeding strategies and 3-NOP, are currently limited to intensive livestock systems.

Demand for red meat is projected to increase, making it unlikely that reductions in the livestock herd will reduce enteric methane emissions. As meat alternatives become less expensive, they may begin to replace consumption of low-grade meats. The mitigation benefit of a switch to synthetic meat is uncertain. Importantly, in the case of plant-based alternatives, the arable land required to support high-protein crops is limited in Australia. The net emissions from meeting protein requirements on lands that are currently high-carbon pastures has not been determined.

The next largest source of emissions from agricultural activity is land clearing. The average annual emissions from deforestation from 2013 to 2017 was 31 Mt CO<sub>2</sub>eq. From 2010 to 2014, about 75% of deforestation was attributable to agriculture. To achieve the most out of afforestation/reforestation efforts and offset emissions from sources that are more technically challenging to mitigate, further reductions in deforestation are necessary. Since 1990 there has been substantial progress on this front with reductions in deforestation emissions of about 66% since the early 1990s. Although impressive, projections suggest emissions will stabilise near current amounts (~30 Mt CO<sub>2</sub>eq).

Agricultural soils emitted 14 Mt CO<sub>2</sub>eq in 2017. Mitigation options include precision agriculture, planting legumes in rotations, and using inhibitors. The effectiveness varies between and within options. In most cases, a 20% reduction is a reasonable expectation. However, increases in emissions in other parts of the system can reduce or offset reductions. Options that reduce the use of fertiliser also decrease emissions from fertiliser manufacture. If widely adopted, technological advances such as renewably produced fertiliser could substantially reduce upstream emissions.

Remaining agricultural emissions were 7 Mt CO<sub>2</sub>eq in 2017. Options to address these emissions include reductions in field burning and using lime and urea efficiently. Manure management is another source of emissions where options are limited to intensive systems. However, intensive systems contribute a substantial amount to these emissions and the use of anaerobic digestion can reduce whole-farm emissions by over 60%, while reducing costs. Renewables on-farm, such as solar powered irrigation systems, can substantially reduce upstream emissions and input costs.

The potential carbon sequestration that can be achieved while maintaining productive systems is uncertain due to climate change impacts and variation across systems. Sequestration rates are strongly influenced by rainfall and management, with trees sequestering carbon faster than soil. Achieving carbon neutrality through sequestration is

more likely in farms that have less emissions to offset. Quantifying the co-benefits of trees on-farm, such as providing shade and windbreaks and restoring land quality, would improve the economic case for establishing trees on farm.

National estimates of the potential land sink are variable and dependant on the carbon price. The total sink from afforestation/reforestation in 2017 was 29.3 Mt CO<sub>2</sub>eq. It has been estimated that reforestation of marginal land and strategic reforestation of non-marginal land with environmental plantings could provide a sink of 45 Mt CO<sub>2</sub>eq at a carbon price of \$26-\$27/tonne. More recently, environmental plantings on 12-24% of intensive agricultural land were estimated to provide an average annual sink of 17.5 Mt CO<sub>2</sub>eq to 2060 with a carbon price of \$153/tonne in 2050. Although the carbon price required to encourage destocking was not estimated, ceasing overgrazing has been estimated to provide an annual sink of 16.5 Mt CO<sub>2</sub>eq until soil carbon approaches a new equilibrium.

Bioenergy is a large component of negative emissions scenarios but comprised just 1.4% of Australian energy in 2016-2017. Expanding the industry would provide job opportunities and new income streams in regional areas. While heavily dependent on the feedstock used and the processes involved, bioenergy can provide substantial reductions in emissions. Several factors must be incorporated into decisions regarding its use. Land use is arguably the most important, as it impacts biodiversity, competition for arable land and the net climate benefit. Use of degraded lands for bioenergy feedstock production can alleviate some of these issues. Sourcing feedstock from wastes is more likely to avoid negative impacts and result in emission reductions, and various waste-to-energy initiatives have shown great potential. The emissions reduction potential of sustainably produced bioenergy in Australia is unknown.

Mitigation options need to be assessed against several criteria. The net effect of mitigation options on whole-farm emissions, including land-use change, is required. Mitigation actions can also have consequences on farm system resilience and efficiency, as well as broad-level implications on food security and biodiversity. Simple scenarios were developed based on emissions trajectories and implementation of various mitigation options at differing levels of ambition. This exercise suggests net-zero agricultural emissions by 2030 is possible but the multiple criteria assessments, research to address other knowledge gaps, and mechanisms to incentivise required changes need to be implemented soon.

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# 1. Introduction

The Australian agricultural industry is a major part of the Australian economy, contributing \$59 billion to GDP in 2017-2018 (Australian Bureau of Statistics 2019). A substantial amount comes from sales of wheat, beef, dairy, sugar, and wine. Current projections suggest agriculture will generate \$84 billion by 2030, although the industry has a goal of reaching \$100 billion in that timeframe. This is accompanied by projected increases in output volumes of 50% by 2050 (National Farmers' Federation 2018). The total emissions associated with Australian agricultural production are also projected to increase, with estimates of total emissions ranging from 78 to 112 Mt CO<sub>2</sub>eq by 2030 (The Centre for International Economics 2013; Commonwealth of Australia 2017a). This is up from 70 Mt CO<sub>2</sub>eq in 2017. In addition to increased demand, future agricultural emissions could increase due to climate change reducing productivity and increasing the frequency of conditions that result in emissions from soils (Bell *et al.* 2012b).

Agriculture dominates land use in Australia (Table 1). A total of 54.1% of the land area is used for grazing. Dryland cropping comprises 3.6% of the land area with all other agricultural land uses combined comprising 3.0% (ABARES 2016). Given the large

*Table 1: Australian land use (ABARES 2016)*

Land Use	Area (km <sup>2</sup> )	Percent (%)
Grazing natural vegetation	3,448,896	44.87%
Grazing modified pastures	710,265	9.24%
Dryland cropping	275,928	3.59%
Dryland horticulture	743	0.01%
Irrigated pastures	6,048	0.08%
Irrigated cropping	9,765	0.13%
Irrigated horticulture	4,552	0.06%
Intensive animal and plant production	1,414	0.02%
Non-agricultural land	3,228,296	42.00%

expanses of grazed land, management of these lands has a major influence on Australian emissions and the emissions associated with red meat products.

Current policies relevant to agricultural and land use emissions at the national level are the Emissions Reduction Fund (ERF), the small-scale renewable energy scheme, and the large-scale renewable energy target (Commonwealth of Australia 2017a). The ERF is a reverse auction in which projects following recognised methodologies to reduce emissions can receive payments. Several of the methodologies are related to the agricultural and land sectors and were originally part of the Carbon Farming Initiative. The small-scale renewable energy scheme is relevant for producers that are considering using renewable energy in their operations. The large-scale renewable energy target is geared toward big projects, for instance some bagasse biomass plants. This scheme would be likely be relevant to other bioenergy plants. States have their own policies in place. For instance, Queensland has the Biofutures 10-year Roadmap and Action Plan (Queensland Department of State Development Manufacturing Infrastructure and Planning 2016) as well as a waste strategy including waste-to-energy (Department of Environment and Heritage Protection 2014).

Profitability is one of the drivers of on-farm decision making. The current price of carbon through the ERF is low, averaging \$14.17 per tonne in July 2019 (Clean Energy Regulator 2019a) and there are complexities and costs associated with carbon trading (Smith 2004; Sanderman *et al.* 2010). Mitigation options that have co-benefits such as reduced costs, increased productivity and/or are straight-forward to implement are those that get the most uptake. Farmers increasingly need to consider several other drivers when making on-farm decisions including animal welfare, environmental regulations, changing consumer demands, sustainability reporting required by supply chains and climate change impacts (Rawnsley *et al.* 2018).

This review explores the literature on emissions from the entire Australian agricultural sector, and thus includes emissions not typically included in assessments of agricultural emissions such as electricity, petrol, and the manufacture of goods used on farm (e.g. fertiliser). Incorporating these emissions is required when determining if farms are carbon neutral. Additionally, including these emissions allows mitigation options to be investigated where agriculture and energy overlap, primarily the use of renewables on-farm, the production and use of bioenergy and other options that are more commercially viable for farmers. Calculations included are from other work or based on the literature available through mid-2019.

## 2. National-level emissions

Agricultural emissions have declined slightly since 1990 (80 Mt CO<sub>2</sub>eq; Figure 1). Recently total agriculture emissions in Australia have been fairly stable ranging from 72.6 Mt CO<sub>2</sub>eq in 2013 to 73.0 Mt CO<sub>2</sub>eq in 2017 (Commonwealth of Australia 2018b). Land clearing emissions have declined by 54.7% since 2000, with net emissions from deforestation of 31.3 Mt CO<sub>2</sub>eq in 2015 (Figure 2 (Commonwealth of Australia 2017a)). Most land-use emissions occur for grazing purposes (Australia National Greenhouse Accounts Land Sector Reporting 2009; Evans 2016). Net land-use emissions from “grasslands remaining grasslands” were 8.7 Mt in 2015, and net emissions from “croplands remaining croplands” was a sink of 4.2 Mt (Figure 2 (Commonwealth of Australia 2017a)).

Despite a short-term stabilisation, total agricultural emissions are projected to increase, driven by greater demand for exports (The Centre for International Economics 2013). The Commonwealth of Australia (2017a) projected agricultural emissions of 73 Mt CO<sub>2</sub>eq by 2020 and 78 Mt CO<sub>2</sub>eq by 2030. The Centre for International Economics’ (2013) agricultural emissions projections are 112.15 and 132.51 Mt CO<sub>2</sub>eq for 2030 and 2050, respectively. Land clearing is also projected to increase slightly, resulting in net emissions of 13 Mt by 2030 (Commonwealth of Australia 2017a). Projected increases in land clearing are driven by increases in livestock numbers (The Centre for International Economics 2013).

Enteric methane - emissions from livestock digestion due to rumen microbes - is the single largest contributor to Australian agricultural emissions (Figure 1). In 2016, enteric methane accounted for 71.9% (49.7 Mt CO<sub>2</sub>eq) of agricultural emissions, excluding those associated with land-use change (Commonwealth of Australia 2018b). Enteric methane emissions have been around 50 Mt CO<sub>2</sub>eq per year for the last several years with changes in these values reflecting changes in the number of cattle and sheep (Commonwealth of Australia 2017a). The number of livestock is primarily a function of grazier’s terms of trade and climate indicators (Commonwealth of Australia 2017a). Given these emissions are associated with ruminant animals, they are primarily attributable to the red meat and dairy sectors.

Most of the remaining methane emissions from agriculture are from manure management. This source accounted for 2.5 Mt CO<sub>2</sub>eq in Australia in 2016 and is associated with all livestock industries, including poultry. Burning of residues (0.2 Mt CO<sub>2</sub>eq) and rice cultivation (0.1 Mt CO<sub>2</sub>eq) comprise the remaining methane emissions.

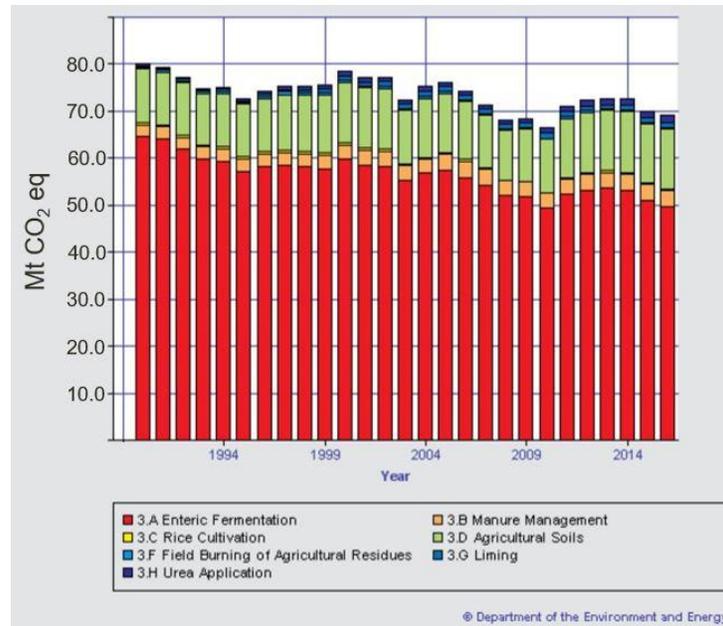


Figure 1: Australian agricultural emissions from 1990 to 2016 (Commonwealth of Australia 2019a)

Nitrous oxide (N<sub>2</sub>O) emissions are primarily from agricultural soils with high nutrient concentrations. Of the 14.0 Mt CO<sub>2</sub>eq of N<sub>2</sub>O emissions from agriculture in Australia in 2016, 12.8 Mt CO<sub>2</sub>eq were from agricultural soils. This is the primary GHG source from cropping sectors and is related to the application of fertilisers. Manure management (1.0 Mt CO<sub>2</sub>eq in 2016) and burning of agricultural residues (0.1 Mt CO<sub>2</sub>eq in 2016) also contribute to N<sub>2</sub>O emissions (Commonwealth of Australia 2016).

Conversion of forest into grazing or croplands is the second largest source of agriculture-related emissions, which are reported as Land Use, Land-Use Change and Forestry (LULUCF) emissions (Figure 2). In 2010, land clearing for agriculture resulted in 56 Mt CO<sub>2</sub>eq of emissions, approximately 10% of national emissions (Longmire *et al.* 2014). Recently this has declined, with all emissions from land-use change, including land-use change to mining and settlements, resulting in 44.8 Mt CO<sub>2</sub>eq of emissions in 2016 (Commonwealth of Australia 2018b). Emissions from land-use change have been relatively stable since 2015 (Commonwealth of Australia 2018d).

Emissions of CO<sub>2</sub> from soils in Australia vary from year to year, sometimes providing a sink and other times being a source. These emissions are calculated using FullCAM modelling and are LULUCF emissions. The 2011 to 2015 average emission of “croplands remaining

croplands” was 0.1 Mt CO<sub>2</sub>eq (Figure 2). Emissions from grasslands and croplands have declined since the 1990s (Commonwealth of Australia 2017a).

Sources of fossil CO<sub>2</sub> emissions are relatively small for Australian agriculture. In recent years, agriculture’s Scope 2 emissions, which are indirect CO<sub>2</sub> emissions from purchased electricity, have ranged from 1.8 Mt CO<sub>2</sub>eq in 2014 down to 1.3 Mt CO<sub>2</sub>eq in 2016 (Commonwealth of Australia 2016). Scope 2 emissions are typically not included within the agriculture inventory, nor are Scope 3 emissions, which are emissions that are part of the supply chain that result from activities not controlled by the producer. Manufacture of fertiliser or transport of goods are examples of Scope 3 emissions for farms. Life cycle assessments used to calculate emissions intensities (EI), i.e. the emissions per unit of product, typically include Scope 2 and 3 emissions.

Although the biological emissions of methane and nitrous oxide are much more effective at trapping heat than carbon dioxide, they do not persist in the atmosphere as long (IPCC 2014). Their lifespans are more comparable to carbon stored in land sinks, such as trees and soil (Parliamentary Commissioner for the Environment 2019). Therefore, biological based-land sinks are more appropriately used to offset methane and nitrous oxide than fossil carbon dioxide emitted to the atmosphere, which remains for centuries. There are also

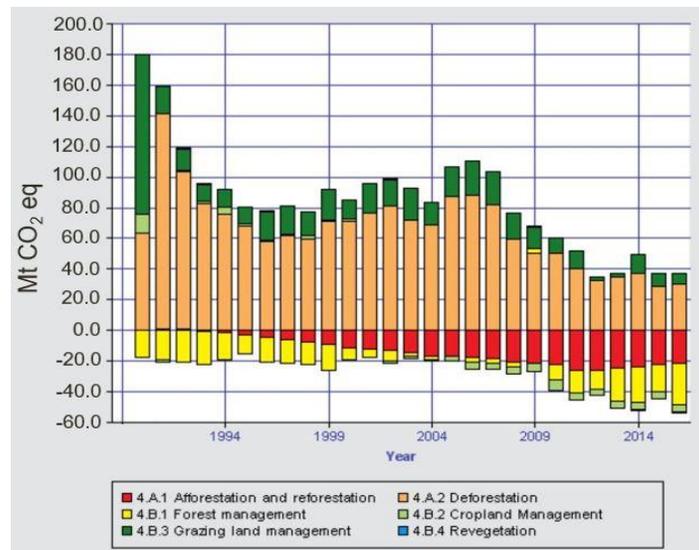


Figure 2: Land use and land use change emissions in Australia from 1990 to 2016 (Commonwealth of Australia 2016)

practical reasons for linking agricultural and LULUCF emissions. Decisions regarding both the mitigation of these emissions and carbon sequestration activities are made by landholders. Land-sinks can also provide offsets while the technical challenges in mitigating methane and nitrous oxide are addressed (Parliamentary Commissioner for the Environment 2019).

### 3. Emissions intensities by sector

In Australia the EI of several agricultural products have been relatively well-researched across regions, although studies are largely cradle to farm-gate, with less known about emissions resulting from activities downstream of the farm. This is the case for livestock sectors as well as wheat and sugarcane. Less is known regarding EI of products in differing systems (Renouf and Fujita-Dimas 2013), such as the difference between conventional and organic systems. This comparison will require careful consideration of all soil emissions, including changes in soil carbon stocks over time.

The total GHG emissions from a given agricultural industry is influenced by both EI and the level of production. Ruminant livestock such as cattle and sheep have the highest EI. Typically, monogastric livestock have the next highest EI, followed by irrigated cropping and then dryland cropping. The combination of EI and the size of that industry determines the overall emissions contribution. For instance, in Australia, cattle and sheep are large agricultural producers with high EI, usually over 5 t CO<sub>2</sub>eq/t, and thus are responsible for most of Australia's agricultural GHG emissions. The red meat industry estimates the production of beef and sheep meat was responsible for 68.6 Mt CO<sub>2</sub>eq of emissions in 2015, including land-use change, production of feed, on-farm and processing emissions (Mayberry *et al.* 2018). The wheat industry contributes about 8.6 Mt CO<sub>2</sub>eq per year, (assuming a median EI of 327 kg per tonne and an average (2011-2017) production of 26.4 million tonnes (ABARES 2018)). All cropping accounted for 31.5% of Australia's agricultural emissions in 2009 due to the high level of crop production in Australia (Tan *et al.* 2013).

Reductions in EI allow for increases in production to result in smaller increases in emissions that would otherwise occur, since a unit of product can be produced with fewer emissions. For instance, total enteric methane emissions increased only 1.6% with a doubling of milk production due to a 40% reduction in the methane EI (Moate *et al.* 2016). Improved efficiencies often decrease input costs and/or increase farm profits. This provides a strong incentive for implementation by producers. However, these options in isolation often do not lead to total emissions reductions.

Many EI estimates do not include carbon emissions or uptake by soils (Ridoutt *et al.* 2017). This could result in a decrease in net emissions in the case where climate, agricultural management factors, and other influences allow for soil carbon sequestration and would

increase emissions when this is not the case. For instance, emissions of CO<sub>2</sub> from the soil were much higher from the burnt cane system than a green-cane harvesting sugarcane system (Denmead *et al.* 2008). This could be a large effect, particularly in systems that incorporate perennial plants such as orchards and pastures.

The unit of analysis and other study design choices also influence EI estimates. Units that incorporate more processing, such as a kilogram of retail ready meat compared to a kilogram of liveweight, are more likely to lead to higher estimates (Peters *et al.* 2010; Wiedemann *et al.* 2010). Allocation, which is how emissions are attributed to multiple products, can also have large impacts on EI results (Eady *et al.* 2012). Although there is compelling evidence for production processes, region, and annual variability to influence EI, methodological differences can be a substantial source of variation.

Given the demands on existing arable land and the emissions consequences of converting lands to cropland, the type of land used by livestock systems is an important factor that is not necessarily captured in life cycle assessments. There can be a trade-off with intensification of livestock systems. Increasing grain feeding improves growth and reduces methane emissions, resulting in lower EI, but increases reliance on arable land and increases water use (Ridoutt *et al.* 2014; Wiedemann *et al.* 2015a). The need for arable land to produce animal feed results in increased importance of feed source on EI of pork and chicken production compared to beef and lamb (Nijdam *et al.* 2012). One way to address this issue is by determining the edible protein efficiency conversion, which is the protein content of the product divided by the human edible protein in the feed consumed by livestock. Protein conversion efficiency of Australian red meat exported to the USA was 7.9 for grass-fed beef, 2.9 for bone-in lamb, and 0.3 to 0.5 for grain-fed beef, which had lower EI (Wiedemann *et al.* 2015a). Another way is to include an area-based functional unit in life cycle assessment as well as the traditional mass-based functional unit (Salou *et al.* 2017), although there are complications regarding land quality or suitability for various uses. Incorporating land quality indicators, such as net primary productivity, into land use metrics and multi-indicator approaches are areas in which life cycle assessments are developing (Ridoutt *et al.* 2011; Ridoutt *et al.* 2014).

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### 3.1 Emission intensities of animal products

Emissions intensity of some Australian meat products have been improving in recent years. It has been estimated that the EI of beef, excluding land-use change emissions, declined 14%

from 1981 to 2010. However, this has been accompanied by a sevenfold increase in the use of land for feed production. Emissions from land use and land-use change had a high degree of uncertainty but were estimated to have declined 42% over the same period largely due to restrictions on deforestation (Wiedemann *et al.* 2015b).

## Beef systems

Estimates of EI of beef in Australia are displayed in Table 2 and Figure 3. Weaning rate and average daily gain explained much of the variation in estimates that was observed in a life cycle assessment of beef from eastern Australia (Wiedemann *et al.* 2016a). Emissions intensity of a system in Gympie with weaners as the primary product were greater than a system in the Arcadia Valley with finished steers as the primary product (Eady *et al.* 2011b). Variation over time in the EI of an organic farm in Victoria was due to differences in the animals used (breeders vs. weaners) and the amount of weight gain in different years (Peters *et al.* 2010). Enteric methane was the primary contributor to EI, often comprising over 70% of the total (Peters *et al.* 2010; Browne *et al.* 2011; Eady *et al.* 2011b; Wiedemann *et al.* 2015a; Wiedemann *et al.* 2016a).

## Sheep systems

Sheep meat generally has lower EI than beef (Table 2 and Figure 3), this is partly due to sheep reaching market weight faster than cattle (Peters *et al.* 2010). Land use and land-use change emissions associated with sheep range from a sink of 2.4 kg CO<sub>2</sub>eq/kg to an additional emission of 0.4 kg CO<sub>2</sub>eq/kg depending on the carbon sequestration scenario

Table 2: Emissions intensity of meat products in kg CO<sub>2</sub>eq per kg of product.

Animal product	Liveweight	Carcass Weight	Retail ready/ bone free meat
Beef	10.1 to 22.9 <sup>1,2,3</sup>	--	14.4 to 34.5 <sup>4,5,6</sup>
Sheep meat	5.1 to 7.9 <sup>5,7,8</sup>	10.2 to 12.6 <sup>9,10,11</sup>	16.1 to 19.4 kg <sup>4,5,6</sup>
Pork	2.1 to 4.5 <sup>5,12</sup>	3.1 to 5.5 <sup>13</sup>	6.3 to 7.4 <sup>5,14</sup>
Chicken	1.1 to 2.6 <sup>5,15, 16</sup>	--	2.5 to 3.1 <sup>15,17</sup>

<sup>1</sup>Eady *et al.* (2011b), <sup>2</sup>Ridoutt *et al.* (2011), <sup>3</sup>Wiedemann *et al.* (2016a), <sup>4</sup>Clune *et al.* (2017), <sup>5</sup>Eady *et al.*

(2011a), <sup>6</sup>Wiedemann *et al.* (2015a), <sup>7</sup>Biswas *et al.* (2010) <sup>8</sup>Wiedemann *et al.* (2016b) <sup>9</sup>(Peters *et al.* 2010)

<sup>10</sup>Browne *et al.* (2011) <sup>11</sup>Eady *et al.* (2012) <sup>12</sup>Wiedemann *et al.* (2016c), <sup>13</sup>Wiedemann and Watson (2018)

<sup>14</sup>Wiedemann *et al.* (2010) <sup>15</sup>Wiedemann *et al.* (2017) <sup>16</sup>Bengtsson and Seddon (2013) <sup>17</sup>Hall *et al.* (2014)

(Wiedemann *et al.* 2015a; Wiedemann *et al.* 2016b). Enteric methane comprises greater than 80% of the cradle to farm-gate emissions for both wool and sheep meat (Biswas *et al.* 2010; Brock *et al.* 2013; Wiedemann *et al.* 2015a).

Emissions intensity of wool production is highly variable, primarily due to differences in allocation method and production systems (Table 3, Figure 3). Allocation methodology has a substantial impact on EI of sheep products (Wiedemann *et al.* 2015c; Cottle and Cowie 2016). A larger proportion of emissions are associated with co-products in sheep systems than in beef systems (Wiedemann *et al.* 2015a). Estimates using economic allocation shift the emissions to high value co-products (Eady *et al.* 2012) and can be more variable than other allocation methods (Brock *et al.* 2013; Wiedemann *et al.* 2015c).

*Table 3: Emissions intensity of Australian wool categorized by allocation method and system type,  $\mu\text{m}$  is the diameter of the wool. All values are for greasy wool unless otherwise indicated.*

System/ wool type	Allocation		
	Mass basis	Economic	Biophysical / Resource Use
100% Merino (superfine/ fine wool)	18.1 -18.7 (18.5 $\mu\text{m}$ ) <sup>1*</sup> 10.5 (17 $\mu\text{m}$ ) <sup>2</sup>		25.1 (17 $\mu\text{m}$ ) <sup>3</sup>
Dual-purpose Merinos (wool-meat)	8.6 (21 $\mu\text{m}$ ) <sup>2</sup>	24.9 (19 $\mu\text{m}$ ) <sup>6</sup>	19.5-21.7 (20-21 $\mu\text{m}$ ) <sup>3</sup>
First cross/ prime lamb (meat focus with wool)		15.3-16.7 <sup>4</sup> 14.8 (19 $\mu\text{m}$ ) <sup>6</sup>	
Mixed system (wheat-wool)		28.7 (19.5 $\mu\text{m}$ ) <sup>5</sup> 6.6 <sup>4</sup>	33.6 (19.5 $\mu\text{m}$ ) <sup>5</sup>
Mixed system (beef-wool-sheep meat)	8.5 (17 $\mu\text{m}$ ) <sup>7</sup>	35.8 (17 $\mu\text{m}$ ) <sup>7</sup>	20.7 (17 $\mu\text{m}$ ) <sup>7</sup>

<sup>1</sup>Browne *et al.* (2011) \*clean-fleece, <sup>2</sup>Wiedemann *et al.* (2015c), <sup>3</sup>Wiedemann *et al.* (2016d) protein mass allocation, <sup>4</sup>Biswas *et al.* (2010), <sup>5</sup>Eady *et al.* (2012) resource use allocation, <sup>6</sup>Brock *et al.* (2013) <sup>7</sup>Cottle and Cowie (2016) protein mass allocation

## Dairy systems

Emissions intensity calculations for Australian milk that incorporate the protein and fat content are just over 1 kg CO<sub>2</sub>eq per kg milk (Figure 3). A kilogram of milk corrected to a standard of 4.0% fat and 3.3% protein EI averaged 1.11 kg CO<sub>2</sub>eq based on 139 farms and using an allocation based on feed requirements ( $\bar{x}$  = 89.8% to milk) (Gollnow *et al.* 2014)

and 1.04 kg CO<sub>2</sub>eq based on 41 farms with all emissions allocated to milk (Christie *et al.* 2012). A review estimated the mean EI of Australian and New Zealand milk as 1.34 (SD 0.4) kg CO<sub>2</sub>eq per kg or litre (Clune *et al.* 2017). Milk production explains the most variability in EI between farms. Production could serve as an indicator of emissions at the national level, but variability between farms precludes its use at a farm level. Differences in EI between farms were partially related to the amount of grain, with supplementary forage improving EI of milk (Christie *et al.* 2012). Methane is the largest contributor to emissions but comprises comparatively less to EI of dairy than red meat, with averages of 55.5% (Christie *et al.* 2012) and 57% (Gollnow *et al.* 2014) attributable to enteric fermentation. Milk production is associated with a greater contribution of emissions from manure and cradle to farm-gate electricity and diesel than other livestock sectors (Ridoutt *et al.* 2011; Christie *et al.* 2012; Gollnow *et al.* 2014; Wiedemann *et al.* 2016b).

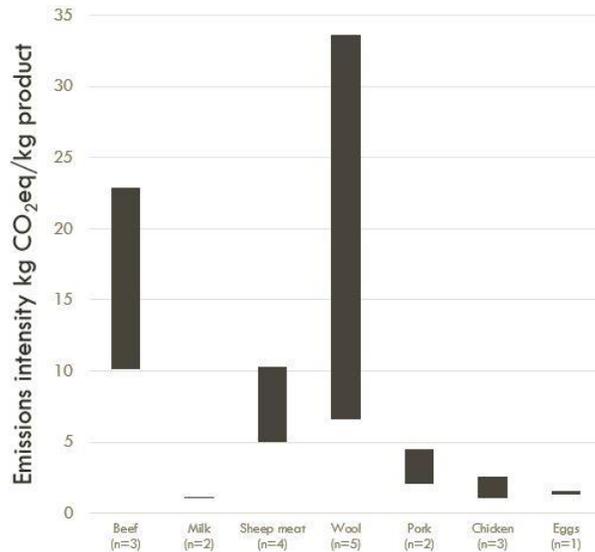


Figure 3: Emissions intensity of Australian animal products based on a review of the literature, citations in text. Meat product ranges are limited to estimates for kg liveweight.

## Piggeries

Emissions intensity estimates for pork are shown in Table 2 and Figure 3. The type of production system is a primary source of variation. Emissions intensities were 3.1 and 5.5 kg CO<sub>2</sub>eq per kg carcass weight for a deep-litter system and a slatted and flushed system, respectively. Within conventional piggeries, feed-conversion ratio explained 88% of the

variation in EI (Wiedemann and Watson 2018). Land use and land-use change emissions associated with pork production ranged from 0.08 to 0.7 kg CO<sub>2</sub>eq/kg pork with variation due to the amount of soybean meal included in the diet (Wiedemann *et al.* 2010).

Methane from manure was the major source of emissions from pork production, comprising 66% of emissions from the slatted and flushed system (Wiedemann *et al.* 2010) and 50% of wholesale pork emissions. A substantial portion of emissions (26.9%) are attributable to feed production (Wiedemann *et al.* 2016c). Remaining emissions were comprised of meat processing (8.0%), farm energy and services (6.5%), and indirect N<sub>2</sub>O emissions from the manure management system (3.6%). In contrast to ruminant animals, enteric methane is a small proportion (4.2%) of emissions (Wiedemann *et al.* 2016c).

There is a target to reduce EI of Australian pork to 1.0 kg CO<sub>2</sub>eq/kg liveweight. Scenarios suggest that this could occur as soon as the 2020s depending on adoption rates of new technology and market conditions (Wiedemann and Watson 2018).

## Poultry systems

Chicken meat has low EI compared to red meat and pork (Table 2, Figure 3), due to the lack of enteric fermentation, the relatively quick rearing time, lower feed requirements, and housing that allows for controlled feeding and manure collection (Biswas and Naude 2016). When including downstream processing, EI ranges from 3.45 to 3.71 tons CO<sub>2</sub>eq per ton product (Bengtsson and Seddon 2013; Biswas and Naude 2016). Including land-use change emissions increased EI by 0.5 to 0.9 kg CO<sub>2</sub>eq/kg liveweight (Wiedemann *et al.* 2017).

Feed production was the largest source of EI for chicken, stressing the importance of feed conversion ratio, which is the amount of feed required for a given amount of weight gain in the animal (Bengtsson and Seddon 2013; Hall *et al.* 2014; Wiedemann *et al.* 2017). Feed production accounted for 64% to 75% of emissions when land use and land-use change emissions are included and 55% to 60% when they were excluded (Wiedemann *et al.* 2017). The grow-out phase comprised of energy use for housing and manure was another major source of emissions (42%) (Bengtsson and Seddon 2013; Wiedemann *et al.* 2017). Although EI was much lower than grass-fed beef or lamb, the occupation of arable land was greater (Wiedemann *et al.* 2017).

Based on a system expansion allocation, EI of egg production in Australia was estimated as  $1.3 \pm 0.2$  kg CO<sub>2</sub>eq/ kg and  $1.6 \pm 0.3$  kg CO<sub>2</sub>eq / kg for caged and free-range eggs, respectively. Feed conversion ratio was a main source of the difference between caged and

free-ranged eggs. Feed grain production was the main source of emissions, followed by on-farm energy use and manure management (Wiedemann and McGahan 2011).

## Emissions intensity of protein production

When based on protein content, animal sources have much higher EI than plant sources, such as peas and soya. A review of life cycle assessment studies worldwide found vegetal sources of protein to have EI of about 4 to 20 kg CO<sub>2</sub>eq/ kg of protein. Other meat substitutes, poultry, and beef from dairy cows were also within this range. Eggs, milk, and pork were similar but with higher maximum EI of up to about 75 kg CO<sub>2</sub>eq per kg of protein for pork. Intensive and extensive beef were higher still and exhibited the greatest range in EI from about 75 to 145 and 160 to 640 kg CO<sub>2</sub>eq/kg protein, respectively (Nijdam *et al.* 2012). Other metrics of comparing the EI of foods incorporating nutritional quality are available, including EIs of Australian products based on nutrient density. The same general trends occur, with animal products having greater EI (Doran-Browne *et al.* 2015).

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## 3.2 Emission intensities of plant products

### Wheat production

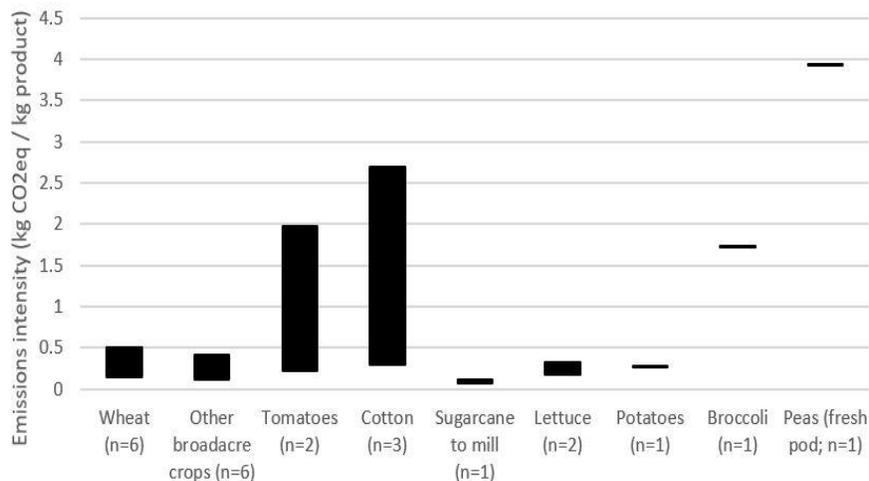
Emissions intensities of wheat in Australia are generally lower than in other parts of the world, ranging from 153 to 500 kg CO<sub>2</sub>eq per tonne of wheat (Biswas *et al.* 2008; Brock *et al.* 2012; Eady *et al.* 2012; Muir *et al.* 2013; Brock *et al.* 2016; Simmons and Murray 2017). The manufacture, transport and application of urea fertiliser was the major contributor (about 30%-40%) in most cases (Brock *et al.* 2012; Muir *et al.* 2013; Simmons and Murray 2017). In another case, N<sub>2</sub>O emissions from the soil were the major source of emissions (Biswas *et al.* 2010). The amount of rainfall, the co-occurrence of livestock on-farm (Biswas *et al.* 2010), and the use of regionally specific N<sub>2</sub>O emissions information (Biswas *et al.* 2008; Brock *et al.* 2012) influence the extent to which N<sub>2</sub>O contributes to EI. N<sub>2</sub>O emissions can also be influenced by the application of lime, with increasing N<sub>2</sub>O likely, and lime application itself associated with substantial emissions. For instance, increasing lime application from 31.5 to 200 kg/ha/year increased cradle-to-gate emissions of wheat from 200 to 300 kg CO<sub>2</sub>eq with lime contributing 39.5% of total emissions (Brock *et al.* 2012).

## Non-wheat broadacre crops

Many broadacre crops have similar emissions as wheat (Figure 4). Reported EI include 110 to 260 kg CO<sub>2</sub>eq per tonne of barley (Eady *et al.* 2011a; Simmons and Murray 2017), 222 to 285 kg CO<sub>2</sub>eq per tonne of canola (Eady *et al.* 2012), 270 kg CO<sub>2</sub>eq per tonne of sorghum (Eady *et al.* 2011a), 325 kg CO<sub>2</sub>eq per tonne of corn (Tan *et al.* 2013), and 180 kg CO<sub>2</sub>eq per tonne of rice (Maraseni *et al.* 2009). Emissions for the production and harvesting of canola were calculated at 420 kg CO<sub>2</sub>eq/t grain (Brock *et al.* 2016). The major contributor to emissions is commonly the manufacture and use of nitrogen fertilisers. For example, fertilisers comprised 65% of emissions for corn (Tan *et al.* 2013) and 88.7% of emissions for rice production in New South Wales (Maraseni *et al.* 2009).

## Cotton production

The EI estimates of cotton are variable ranging from an average of 345 kg CO<sub>2</sub>eq per tonne of cotton lint and seed for irrigated and dryland cotton in New South Wales (Tan *et al.* 2013)



*Figure 4: Emissions intensity of Australian plant products based on literature review, citations in text. Note the y axis range differs substantially from Figure 3.*

to 2674 kgCO<sub>2</sub>eq/t in an irrigated system in Queensland (Maraseni *et al.* 2010a) (Figure 4). In New South Wales, increases in emissions associated with irrigation were offset by increases in production (Tan *et al.* 2013). Cotton management options, including the type of irrigation and the use of no-till have a large impact on EI. Based on data from two farms, the optimal management strategy in terms of EI was a zero tillage system using GM cotton

and lateral move irrigation which emitted 760 kg CO<sub>2</sub>eq per tonne of cotton (Khabbaz 2010). Soil emissions can also vary greatly between systems (Maraseni *et al.* 2010a). Soil N<sub>2</sub>O emissions are often a major source of emissions (Maraseni *et al.* 2010a; Tan *et al.* 2013) with energy use for irrigation contributing substantially in some cases (Maraseni *et al.* 2010a).

## Sugarcane production

Estimates of EI of sugarcane can include an offset due to the co-product, bagasse, replacing fossil-fuel use. All emissions of GHG associated with sugarcane production in Queensland aggregated to just under 1 kg CO<sub>2</sub>eq/kg of monosaccharide. However, a conservative estimate of energy produced by bagasse was nearly as much, leaving a net EI between 0.1 and 0.2 kg CO<sub>2</sub>eq per kg of monosaccharide. In the low input scenario, the EI of sugar was negative. N<sub>2</sub>O emissions vary temporally and spatially and the variation could result in large differences across farms and years (Renouf *et al.* 2008). Another study estimated an emission intensity of 0.07 to 0.11 tonnes CO<sub>2</sub>eq per ton of sugarcane delivered to the mill. If it is assumed that: 1) 0.08 tonnes of CO<sub>2</sub>eq are produced for every tonne of sugarcane (Renouf *et al.* 2010); and 2) 35 million tonnes of cane are purchased by the mills, then emissions of 2.8 Mt CO<sub>2</sub>eq is attributable to sugar. The industry reports energy from bagasse reduces Australian emissions by 1.5 Mt annually (Australian Sugar Milling Council 2019).

## Horticulture

The EI of horticultural products varies. The largest EIs were between 1.17 and 3.94 t CO<sub>2</sub>eq/t for green peas, asparagus, broccoli, sweet corn, French and runner beans, and zucchini/ button squash (Maraseni *et al.* 2010b). All other products investigated had EIs less than 1 t CO<sub>2</sub>eq/ t. Several had EIs in the range of 0.2 to 0.3 t CO<sub>2</sub>eq /t including cabbages, fresh carrots, rockmelon, tomatoes, and onion. Emissions intensity of potatoes was 0.27 t CO<sub>2</sub>eq /t. The smallest EIs were from mushrooms 0.06 t CO<sub>2</sub>eq /t, cucumbers 0.13 t CO<sub>2</sub>eq /t, and celery 0.18 t CO<sub>2</sub>eq /t. Emissions intensities also vary within product in some cases. For instance, the EI of tomatoes for the Sydney market varied from 0.39 to 1.97 kg CO<sub>2</sub>eq due to differences across seasons and production system (Page *et al.* 2012). Due to higher production, the potato industry was the highest emitter with annual emissions of 0.32 Mt. The next highest emitters annually were lettuces (0.09 Mt), sweet corn (0.09 Mt), broccoli (0.08 Mt), tomatoes (0.07 Mt), and pumpkins (0.06 Mt) (Maraseni *et al.* 2010b).

The major source of emissions varies across different vegetables. Emissions associated with irrigation can be high (70-80%) due to the energy required to move water. This was the case for fresh pod green peas, asparagus, French and runner beans and zucchini/button squash, which are all among the vegetables with the highest EI. Energy used in heating greenhouses was a major source of emissions for tomatoes, comprising 83-85% of emissions from medium- and high-tech greenhouse systems (Page *et al.* 2012). Transportation was another major contributor for tomatoes (Page *et al.* 2012) and lettuces (Gunady *et al.* 2012). Agricultural machinery used for harvesting can be a substantial source of emissions for horticulture systems including strawberries (Gunady *et al.* 2012) and lettuce (Gunady *et al.* 2012; Maraseni *et al.* 2012). N<sub>2</sub>O emissions comprised a substantial (21-25%) proportion of emissions of shelled green peas, broccoli, and sweet corn (Maraseni *et al.* 2010b). In orchards, N<sub>2</sub>O emissions ranged from a low of 298 grams/ha/year in a Tasmanian apple orchard to 7.6 kg/ha/year in a Queensland lychee orchard (Rowlings *et al.* 2013; Swarts *et al.* 2016). The low end of this range was in systems in cool temperate regions with low rates of nitrogen fertiliser application and efficient drip irrigation systems (Swarts *et al.* 2016).

The EI values for horticulture are sensitive to energy use. For instance, upgrading irrigation systems from a hand shift to a drip system resulted in a reduction in EI of lettuce from 0.22 to 0.17 t CO<sub>2</sub>eq/t (Maraseni *et al.* 2012). The cradle-to-farmgate EI of tomatoes ranged from 0.3 kg CO<sub>2</sub>eq/kg in field-grown tomatoes to 1.97 kg CO<sub>2</sub>eq/kg for those grown in a high-tech greenhouse (Page *et al.* 2012). Use of renewables in a high-tech greenhouse in South Australia has been reported to save 14,000 tons (0.014 Mt) of CO<sub>2</sub>eq per year (Allen 2015), or 21% of annual emissions associated with Australian tomato production.

## Wine production

Emissions from the Australian wine industry have been estimated for the UK market. A 0.75 litre bottle of Australian wine was estimated to produce 1.25 kg CO<sub>2</sub>eq. This leads to an annual emission of 210,000 t CO<sub>2</sub>eq for Australian wines consumed in the UK, with most of this (70%) attributable to viticulture and distribution. Of the options investigated, the largest reduction, 27,000 t CO<sub>2</sub>eq (13%) per year, was associated with bulk shipping (Amienyo *et al.* 2014).

## 4. Mitigation options: potential and application

Agricultural production involves the integration of soil nutrients, soil water, climate, plant growth and animal production. Any GHG mitigation option can affect multiple parts of the system leading to additional reductions or increases in emissions. For instance, growing legume and shrub species that reduce methane in animals could also increase soil carbon sequestration (Mayberry *et al.* 2018). Changes in emissions can spread across the supply chain, such as when reducing the use of urea fertiliser reduces both on-farm N<sub>2</sub>O emissions from soil, as well as pre-farm gate emissions associated with the manufacture and transport of urea (Brock *et al.* 2012). In contrast, options can decrease one GHG while increasing another. For instance, splitting the application of fertiliser can decrease N<sub>2</sub>O emissions but slightly increase fossil-fuel energy use to apply fertilisers to the crop (Muir *et al.* 2013). Similarly, sequestering carbon in soil can increase emissions of N<sub>2</sub>O (Palmer *et al.* 2017). Despite the grouping of most options by GHG in this section, there are likely consequences on emissions elsewhere in the system.

Whole-farm system analysis needs to be performed to determine the net consequences of implementing a given mitigation option (de Boer *et al.* 2011; Montes *et al.* 2013; Rawnsley *et al.* 2018). Additionally, impacts on productivity, farm system resilience (Christie *et al.* 2012; Rawnsley *et al.* 2018), and food security more broadly (de Boer *et al.* 2011) should be considered.

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### 4.1 Increasing efficiency of production

Increasing productivity is used across sectors to reduce EI by increasing the efficiency of operations (Waghorn and Hegarty 2011). Unlike mitigation options that rely on the carbon price as an incentive to be implemented, increasing the efficiency of production is usually associated with improved profitability. For instance, a study has

- **Potential:** Low to moderate
- **Applicability:** Widespread, all industries
- **Stage:** Established and ongoing

shown that implementing multiple compatible options to increase beef productivity and reduce EI resulted in increased gross margins (Harrison *et al.* 2016).

The environmental benefits of improved efficiency are substantial with a meta-analysis of life cycle assessments showing that increasing input use efficiency would have greater environmental benefit than switching to organic products or grass-fed beef (Clark and Tilman 2017). Life cycle assessments of organic farming often show higher EI than conventional farming due to lower production. However, the details of N<sub>2</sub>O emissions and CO<sub>2</sub> emissions or uptake in these systems are often incomplete. More research on the differences in emissions between these systems in Australia is required (Meier *et al.* 2015).

## Increasing the efficiency of cropping systems

In cropping sectors yield gaps can be improved by addressing nutrient limitations, weeds, and/or disease (Lawes *et al.* 2018), but yield gaps can also be caused by socio-economic factors. Closing the yield gap would mean a doubling of production in many farms (Hochman *et al.* 2016), which would have financial benefits as well as reducing EI. Closing the yield gap for wheat was estimated to reduce EI for a ton of wheat by 80% and 93% for the Western Australian Central zone and the Queensland Central zone, respectively. In another scenario, 20% increases in production associated with increased application of lime reduced EI by 26% in the Western Australian Central zone (Simmons and Murray 2017).

## Increasing the efficiency of sheep, beef and dairy systems

Management options in the livestock sector that increase production per animal reduce EI (Waghorn and Hegarty 2011). For example, replacing several low producing dairy cows with fewer high producing dairy cows, reducing the mortality of replacement animals (Eckard *et al.* 2010; Hristov *et al.* 2013c; Patra 2014), increasing turnoff weights (Wiedemann and Watson 2018) and increasing livestock fecundity (Hristov *et al.* 2013c; Alcock *et al.* 2015) reduce the EI of livestock products. This is because, on average, animals are producing the same amount of product in a shorter amount of time (less time emitting per animal), or more product in a similar time (more product for a given amount of emissions). Selection of higher producing animals and those with greater feed conversion ratio or residual feed intake increases efficiency, reduces EI, and will likely increase profitability (Waghorn and Hegarty 2011). In sheep systems reductions in methane production per lamb sold were 4.6% for increasing ewe cull ages from 5 to 6, 7.8% for increasing the percentage of ewes scanned for pregnancy from 160% to 180%, and 11.7% for implementing hogget

lambing (Cruickshank *et al.*). Several efficiency options implemented on a modelled pasture-based dairy in New Zealand reduced emissions by 27% (Beukes *et al.* 2010). For extensive beef systems that predominate in northern Australia management options that increase production per animal are some of the few that are feasible (Bentley *et al.* 2008).

Careful selection of feed for ruminants such as sheep and cows can reduce EI through increased productivity from high quality diets as well as reduced methane emissions (Beauchemin *et al.* 2009; Hristov *et al.* 2013a). For instance, including *Leucaena* in grazing systems in tropical areas reduced EI by 23% through both increased productivity and reduced methane emissions (Harrison *et al.* 2015). Supplemental feeding reduced methane EI of Japanese Ox by 24.7% (Charmley *et al.* 2008).

## Increasing the efficiency of piggeries and poultry systems

The net effect of supplemental feed on EI varies between ruminant and monogastric (e.g. piggeries and poultry) systems (Bell *et al.* 2012a). For instance, increased demand for soymeal is associated with increased EI of pork and was a potential source of increasing emissions in scenarios of future production (Wiedemann and Watson 2018). Similarly, reducing the use of commercial feeds in small-scale chicken production would improve efficiency (Hall *et al.* 2014). There is potential for reducing the EI of pig feed through synthetic amino acid additions (McAuliffe *et al.* 2016). There are also novel forage supply chains based on algae (Duong *et al.* 2015) and food waste (Salomone *et al.* 2017; Wiedemann and Watson 2018) that require more research to determine their potential benefits.

## Increasing efficiency by reducing waste

Reducing food waste is major aspect of increasing efficiency of food production that has significant co-benefits in addition to GHG mitigation including other environmental factors, food security and farm profitability. Nationally it has been estimated that food waste in both municipal and industrial waste streams was 7.5 million tonnes in 2008-09. This amount of food waste equates to an annual emission of 6.8 Mt CO<sub>2</sub>eq (Mason *et al.* 2011). Although less is known about the amount of waste on farms, it can be substantial. For example, between 10% and 30% of the banana crop from northern Queensland is wasted due to cosmetic standards set by retailers. This represents an annual loss of 137 billion kilojoules, 11.2 gigalitres of water, \$26.9 million as well as 0.016 Mt of CO<sub>2</sub>eq (White *et al.* 2011). Reducing feed wastage by 5% was estimated by the pork industry to reduce emissions by

10% (Australian Pork Limited 2019). Options for avoiding land-filling edible and non-edible food waste, such as providing food to charities and converting waste to energy, need further development and expansion.

## Emissions intensity trends and limitations

The likelihood of continued improvements in EI is unclear. There have been substantial gains in EI across several products over the last few decades. For instance, enteric methane EI of dairy has dropped by 40% since 1990 (Moate *et al.* 2016) and total annual emissions attributable to beef production declined by 55.3% from 2005 to 2015 (Mayberry *et al.* 2018). Globally there is a trend for increasing decoupling of agricultural emissions and production, with crop and livestock production EI reductions since 1970 of 39% and 44%, respectively. In a business-as-usual scenario, agricultural emissions further decouple by 20% to 55% by 2050 (Bennetzen *et al.* 2016).

However, how these improvements can continue is unclear. The reductions in Australian beef emissions are due almost entirely to reductions in deforestation (Mayberry *et al.* 2018). Further reductions in deforestation will become increasingly challenging and if deforestation were to reach negligible levels, the ability to reduce emissions will have to come from other strategies. In addition, climate change is already impacting yields of agricultural products such as wheat (Hochman *et al.* 2017), which could impact EI gains. Due to these factors, as well as a reduction in research and development efforts, increases in annual average production of 2.1% during the late 70s to mid-90s have fallen to an average of 1.7% from 2006-2007 to 2014-2015, after recovering from a low of 0.3% during the Millennium Drought (Figure 5). These factors suggest that maintaining recent trends in EI may become increasingly difficult.

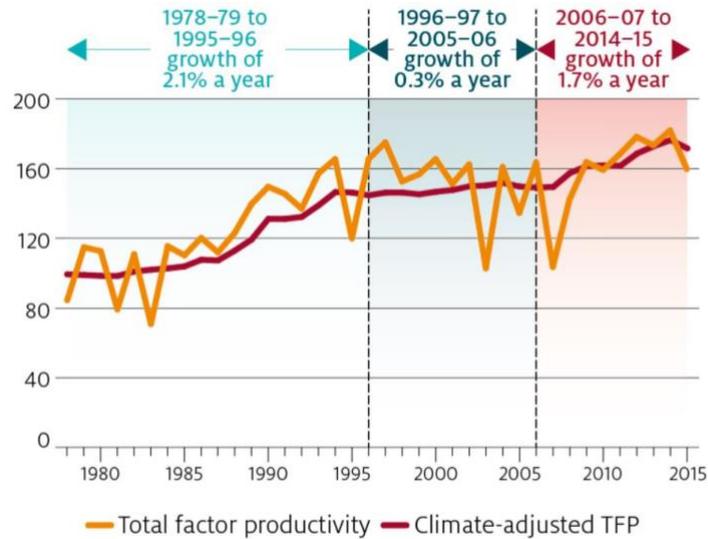


Figure 5 Climate-adjusted productivity growth, cropping farms, 1978-79 to 2014-2015. Source: ABARES (Jackson *et al.* 2018)

Productivity strategies are limited in their mitigation potential due to associated system changes (e.g. higher stocking rates, increased emissions elsewhere in the system due to increased inputs) ultimately increasing total emissions (Beauchemin *et al.* 2008). In addition, there can be risks associated with the intensification measures that can lead to reductions in EI in livestock systems. For instance, in dairy systems of Australia increased reliance on supplemental feeds can leave a dairy subject to financial risk (Christie *et al.* 2012), particularly in dry years when prices of feed can become prohibitive. This reduction in the resilience of farms with intensification is an example of a mitigation option leading to maladaptation.

## 4.2 Methane

### Enteric methane

Given that enteric methane emissions comprise most of Australian agricultural emissions, reductions are critical to achieving net-zero or negative emissions in agriculture. Unfortunately, there are currently limited options, commercially viable or otherwise,

- **Potential:** Low to high
- **Applicability:** Widespread comprises >70% of emissions
- **Stage:** Research to pre-commercial development

capable of sustaining large reductions in enteric methane emissions. Feed additives, farm management to improve efficiency, both through diet and other means, breeding and the development of a vaccine are currently being researched. Most of these options, which require frequent interaction with the animals, are applicable to intensive systems in Australia, such as dairies, that contribute much less to total enteric methane emissions than extensive grazing systems (Figure 6). This limitation results in a focus on management strategies that can realistically be implemented in extensive systems, such as increased productivity. Other detailed reviews on options to mitigate enteric methane emissions are available (Boadi *et al.* 2004; Beauchemin *et al.* 2008; Martin *et al.* 2010; Cottle *et al.* 2011; Hristov *et al.* 2013a).

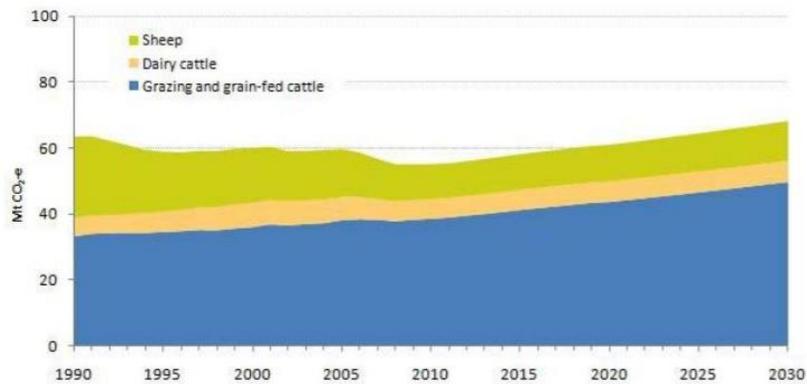


Figure 6: Sources of methane emissions from Australian agriculture. Enteric methane emissions are from grazing cattle (66%), dairy cattle (11%), feedlot cattle (3%), and sheep (20%). NSW Department of Primary Industries (Lines-Kelly 2014)

Several feed additives have been researched as mitigation options for enteric methane including fats, nitrates, 3-nitrooxypropanol (3-NOP), algae (Machado *et al.* 2014), and plant secondary compounds such as tannins (Meale *et al.* 2012). There is a methodology for earning Australian Carbon Credit Units (ACCUs) by reducing enteric methane from dairies through feed additives that increase dietary fat, originally developed under the Carbon Farming Initiative (Department of the Environment 2015c). The addition of supplementary feeds that are high in fats can reduce methane emissions by 20% (Beauchemin and McGinn 2006). However, to date no projects have been submitted using this methodology (Baxter 2019), likely due to the cost of the additives (e.g. canola meal, brewers grain, etc). Nitrates can lead to substantial reductions in enteric methane emissions (Lund *et al.* 2014; Velazco *et al.* 2014), but need to be administered carefully to avoid nitrate toxicity (Meale *et al.* 2012;

Hristov *et al.* 2013a; Patra 2014). Nitrates were less effective in reducing emissions than 3-NOP in 2 Australian dairies and an Australian beef system (Alvarez-Hess *et al.* 2019). Although feed additives can reduce methane emissions it should be noted that these options can have both positive (increase growth rate) and negative (increased transport or N<sub>2</sub>O emissions) impacts on emission intensity and total emissions.

3-NOP is a methane inhibitor that has been shown in multiple studies to effectively reduce methane emissions by impeding the final step of methanogenesis (Hristov *et al.* 2015; Vyas *et al.* 2018). Based on whole-farm system modelling, administering 3-NOP reduced enteric methane emissions by up to 31.9% leading to whole-farm reductions of 17.4%. The breakeven cost of 3-NOP is around \$30/kg or \$50/kg at carbon prices of \$11.82/t CO<sub>2</sub>eq and \$20/ t CO<sub>2</sub>eq, respectively. Increased growth rate is a potential, but not yet proven, co-benefit of using 3-NOP, which would improve the business case. However, 3-NOP is not currently commercially available in Australia and still requires regulatory approval (Alvarez-Hess *et al.* 2019).

Vaccines are another method of manipulating the rumen to potentially reduce methane emissions (Wedlock *et al.* 2013). However, more research is needed to determine the feasibility of developing an effective vaccine.

Genetic improvements in cattle can reduce methane emissions by selecting either for increased feed efficiency or reduced methane output. Existing research has shown a reduction of 4.0% in whole-farm emissions attributable to selective breeding in a modelled dairy farm (Bell *et al.* 2011). Animals with a high net feed efficiency have lower methane EI (Waghorn and Hegarty 2011; Hristov *et al.* 2013c), and can be selected for using the “feed saved” breeding value (Pryce *et al.* 2015). Additional research is required to identify any negative associations between breeding selection based on lower methane production and other required productive and functional traits (Buddle *et al.* 2011). Although these options are further from implementation, if developed they could be applied to the extensive systems that are the predominant sources of methane emissions (Figure 6).

Given that methane emissions are directly associated with the number of cattle and sheep in the national herd, options that reduce livestock numbers would reduce methane emissions (Commonwealth of Australia 2017a). Increasing demand for meat production (FAO 2009) indicates that near-term reductions are unlikely (The Centre for International Economics 2013; Commonwealth of Australia 2017a). However, reductions in the demand for red meat, both through vegetal substitutes and non-ruminant meat, have been shown to reduce

emissions. For instance, replacing a portion of ruminant production with kangaroo was estimated to lower GHG emissions by 16 Mt (Wilson and Edwards 2008). A diet with one serving of red meat per week was associated with a 19% reduction in premature mortality and a global average reduction in emissions of 54% (Springmann *et al.* 2018). Based on a meta-analysis of life cycle assessment studies, adopting low meat or no meat diets has a larger impact than switching to grass-fed beef or organic foods (Clark and Tilman 2017). Establishing a tax on food products of 60€ per t CO<sub>2</sub>eq was estimated to result in a 7% reduction in emissions due to reduced animal numbers. This increases several fold if the land spared is used in bioenergy production that replaces fossil fuels (Herrero *et al.* 2016).

Synthetic meat could offer another low-emission alternative, but the extent of the potential mitigation benefit has not yet been determined. While it is expected that substitution options will soon compete with low-grade meat and processed meat (Bonny *et al.* 2017), it is unclear the extent to which healthy and sustainable diets will be adopted (Ridoutt *et al.* 2017). It is also important to acknowledge that high-protein crops require land of higher quality than current grazing lands in Australia.

## Methane from manure and other organic wastes

The potential for reducing methane from manure on intensive livestock farms is substantial. Most of the data regarding reducing manure methane emissions in Australia comes from

- **Potential:** High
- **Applicability:** Narrow to intermediate, piggeries and potentially dairies and feedlots
- **Stage:** Research phase (feedlots) through to ongoing (pork)

piggeries. In modelled covered anaerobic pond-combined heat and power (CAP-CHP) systems where all manure was treated, whole-farm emissions of piggeries were reduced by 60% to 64% leading to EI of 1.6 and 1.4 kg CO<sub>2</sub>eq per kg liveweight depending on the scenario (Wiedemann *et al.*

*et al.* 2016c). Deep litter systems also provide substantial emissions reductions (40% to 80%) over conventional piggeries (Phillips *et al.* 2016; Australian Pork Limited 2019). The pork industry estimates whole-farm emissions reductions of 75% to 84% for farms that capture biogas, an average reduction in emissions across all farms of 51%, and associated reductions in EI from 3.9 kg CO<sub>2</sub>eq /kg to less than 1 kg CO<sub>2</sub>eq/kg of pork (Australian Pork Limited 2019). If 50% of the industry was using biogas, cradle-to-gate emissions from the sector could be reduced 30% (Wiedemann *et al.* 2016c).

Using biogas from manure has been adopted and is expected to increase in piggeries (Wiedemann *et al.* 2010). As of 2018, 13.5% of Australian pork came from farms with biogas capture (Tait 2017). The technology has been adopted in at least one Australian dairy (Cooke 2017) and has potential in other intensive systems in Australia, such as feedlots (Watts and McCabe 2015). It has also been adopted at abattoirs (Baxter 2019), reducing post-farmgate emissions. Limitations to adoption in piggeries include difficulty selling electricity to the grid, issues of scale, and distances between the area where most manure is produced and the location of breeders, which is where the most heat is required (Wiedemann *et al.* 2016c). The economic feasibility of this technology decreases with decreasing size of the operation (Wiedemann and Watson 2018). Short hydraulic retention time storage systems can suit smaller piggeries (McGahan *et al.* 2016). In the case of Australian dairies, the implementation can require system-level changes that have broad implications on farm management which would influence adoption rates. Using gasification to convert dry feedlot manure to syngas and biochar using a biomass integrated gasification combined cycle (BIGCC) system, resulted in net negative emissions of -643 kg CO<sub>2</sub>eq per tonne (Wu *et al.* 2013). This technology is yet to be implemented at scale in Australia or elsewhere.

Projects that capture methane from manure in piggeries (Department of the Environment 2013) and dairies (Department of the Environment 2015b) can earn ACCUs, which improve the cost effectiveness of biogas installations (Wiedemann and Watson 2018). Abattoirs using this technology can also earn ACCUs under the Industrial Energy Efficiency methodology (Department of the Environment 2015a). Piggeries have taken advantage of this opportunity and currently comprise 88.4% of issued ACCUs from agricultural methodologies, leading to mitigation of about 0.48 Mt CO<sub>2</sub>eq (Baxter 2019). Carbon credits have had a large impact on the economic feasibility on these projects, which can require large capital investments (Wiedemann and Watson 2018). Biogas also provides new income streams from electricity generation (Australian Pork Limited 2019).

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## 4.3 Nitrous oxide

### Manure and urine patches

Reported emissions of N<sub>2</sub>O from manure are highly variable due to differing methodologies of data collection and differences due to variation in climate, manure characteristics,

management systems and other factors (Broucek 2017, 2018). Given that conditions for N<sub>2</sub>O formation are not typically met in anaerobic manure systems, they produce less N<sub>2</sub>O than urine deposits (Broucek 2017). A review of mitigation options globally categorised nitrification inhibitors as having high potential to reduce N<sub>2</sub>O emissions (>30% reduction) from manure and urine patches (Gerber 2013). The impacts of land application of various fertilisers, including manure, are discussed in the soil emissions section, below.

Application of inhibitors have been shown to reduce nitrogen losses in urine patches. In New Zealand several urease inhibitor products reduced N<sub>2</sub>O emissions by 42% to 56% (Singh *et al.* 2013), while two nitrification inhibitors reduced N<sub>2</sub>O emissions by 62.3% to 65.8% (Di and Cameron 2012). In another New Zealand study, the urease inhibitor in isolation had modest effects on N<sub>2</sub>O emissions. However, used in conjunction with a nitrification inhibitor, N<sub>2</sub>O emissions were reduced by 39%, 67% and 28% in autumn, spring and summer, respectively.

- **Potential:** Variable within and across strategies
- **Applicability:** Across sectors, major emission from cropping. Total of 15 Mt CO<sub>2</sub>eq
- **Stage:** Some strategies on-going others being researched.

Ammonia losses were also reduced, and pasture production and nitrogen uptake by plants increased (Zaman and Blennerhassett 2010). Reducing N<sub>2</sub>O loss from urine is also accomplished through managing the energy to protein ratio of the diet (Reisinger *et al.* 2017; Eckard and Clark 2018).

## Soil

N<sub>2</sub>O emissions from the soil are the primary emissions in cropping systems. Options for reducing N<sub>2</sub>O emissions from agricultural soils include using inhibitors, substituting conventional fertiliser with other methods of adding nitrogen, and altering management, such as the rate and timing of fertiliser application, use of cover crops or precision agricultural systems.

There is a high degree of variation in the effectiveness of inhibitors applied to agricultural soils. A review of the effectiveness of the nitrification inhibitor, dimethyl pyrazole phosphate (DMPP) reported reductions in N<sub>2</sub>O emissions ranging from 8% to 57% across studies (Lam *et al.* 2017). In an experiment in subtropical cereal cropping systems, DMPP and polymer-coated urea reduced annual N<sub>2</sub>O emissions by 83% and 70%, respectively (Scheer *et al.* 2016). Conversely, annual reductions in N<sub>2</sub>O emissions were inconsistent using

DMPP in subtropical rice (Rose *et al.* 2017). A urease inhibitor and DMPP had no impact on N<sub>2</sub>O emissions or pasture yield in a hot-dry cropping area under irrigation (Dougherty *et al.* 2016). In addition to variable effectiveness, reductions in N<sub>2</sub>O emissions can lead to increased ammonia volatilisation, which is associated with indirect N<sub>2</sub>O emissions. The cumulative effect of this ranges from small reductions in total N<sub>2</sub>O emissions to slight increases in total N<sub>2</sub>O emissions (Lam *et al.* 2017). Even in cases where total N<sub>2</sub>O is substantially reduced, more research is required to determine in what circumstances the cost of inhibitors can be offset by reduced need for fertiliser or yield increases (Scheer *et al.* 2016).

Other sources of nitrogen such as compost and manure may also reduce soil N<sub>2</sub>O emissions. Globally, the technical N<sub>2</sub>O mitigation potential of application of manure to the field is estimated at 10 to 75 Gt CO<sub>2</sub>eq per year (Herrero *et al.* 2016). However, more research needs to be done in Australian conditions to determine the potential of manure to reduce total emissions in these systems (Biswas *et al.* 2008). Currently available information on the use of manure in Australia is inconclusive. Measurements of N<sub>2</sub>O emissions following application of piggery manure vary widely, with emissions following application of effluent between 0.0123 and 0.0165 kg N<sub>2</sub>O-N/kg N in two studies and as high as 0.13 kg N<sub>2</sub>O-N/kg N in another (REFs). Piggery effluent typically has higher N<sub>2</sub>O emissions than piggery litter, which ranged from 0.001 to 0.0023 kg N<sub>2</sub>O-N/kg N (Phillips *et al.* 2016). Following anaerobic digestion, manure has more available carbon and nitrogen which increases N<sub>2</sub>O emissions from land where it is applied (Montes *et al.* 2013). In one case study, replacing fertiliser with manure in a wheat crop in Western Australia resulted in the same total emissions, with emissions in the fertiliser scenario predominately from chemical production and emissions in the manure scenario predominantly paddock emissions (Engelbrecht *et al.* 2013).

Other additives that have been trialled include brown-coal urea and nitrogen-fixing bacteria. Use of a slow release brown-coal urea fertiliser on potted silver beets resulted in a 29% reduction in N<sub>2</sub>O emissions and a 23% to 27% increase in productivity depending on the soil type (Saha *et al.* 2019). A company that has developed nitrogen-fixing bacteria to apply in furrow estimates that 35% adoption in cornfields of the United States would save 20,000 tonnes of direct N<sub>2</sub>O emissions (5.96 Mt CO<sub>2</sub>eq) as well as 500,000 tonnes of nitrate leaching and the associated impacts, which include indirect N<sub>2</sub>O emissions (approx. 1.86 Mt CO<sub>2</sub>eq) (Pivot Bio 2019). This technology would need to be developed for Australian crops and conditions.

A common way to reduce the need for conventional fertiliser is to include a legume in the crop rotation. Growing a crop of chickpeas prior to a wheat crop allows for reduced application of nitrogen fertiliser to the wheat crop leading to a 21% reduction in N<sub>2</sub>O emissions (Muir *et al.* 2013). In a cradle-to-gate life cycle assessment of wheat in Western Australia, a scenario with a lupine crop rotation was associated with a 5.4% reduction in GHG emissions (Engelbrecht *et al.* 2013). Legumes also reduce input costs, increase carbon sequestration in the soil, and generally improve soil health (Muir *et al.* 2013; Stagnari *et al.* 2017). However, it should be acknowledged that if the displaced wheat production occurs in low-efficiency systems, including legumes in the rotation can increase EI (Simmons and Murray 2017).

Several other management decisions can reduce N<sub>2</sub>O emissions from soil. Application times affect the extent to which emissions are reduced, with N<sub>2</sub>O emissions greater when fertiliser is applied before rainfall (Muir *et al.* 2013) and in the hot and moist spring and summer period in subtropical areas (Rowlings *et al.* 2013). Improving drainage (Swarts *et al.* 2016) and, in irrigated systems, monitoring soil moisture to limit the occurrence of conditions in which N<sub>2</sub>O is likely to be produced (>40% water filled pore space) can also reduce emissions (Maraseni *et al.* 2010b). Similarly, precision agriculture techniques, including variable-rate application in which fertiliser is only applied where it is required, can lead to reductions in N<sub>2</sub>O emissions and inputs (Muir *et al.* 2013; Simmons and Murray 2017). In Western Australia and Queensland, fertilising wheat at a variable rate, assuming yields are maintained with a 20% reduction in nitrogen inputs, resulted in a 30% to 34% reduction in N<sub>2</sub>O emissions (Simmons and Murray 2017). Estimates of increased emissions due to increased fossil-fuel use associated with a split application of fertiliser range from about 3 kg CO<sub>2</sub>eq to 14 kg CO<sub>2</sub>eq/ha, which would be offset by small reductions in N<sub>2</sub>O emissions (Brock *et al.* 2012; Muir *et al.* 2013). The addition of cover crops can also reduce N<sub>2</sub>O emissions by reducing residual nitrate (Maraseni *et al.* 2010b). Crops that have been bred for increased nutrient uptake efficiency would reduce input requirements and emissions (Ridoutt *et al.* 2013).

## 4.4 Land-based CO<sub>2</sub>

Emissions from agricultural lands and from land-use change are accounted for in the LULUCF sector in national inventories and are often omitted from life cycle assessments. However, emissions associated with land use and land-use change attributable to

- **Potential:** Moderate to high
- **Applicability:** All land sectors
- **Stage:** Established methods but implementation limited due to several factors including costs

agriculture were nearly a third of national agricultural emissions in 2017. About 75% of deforestation emissions were attributable to agriculture from 2010-2014 (Figure 7). There are several challenges in reducing emissions from land-use change, such as the increasing demand for agricultural products, the impacts of climate change on production, and the potential for competition with an expanding bioenergy sector. Nonetheless, to make the most of reforestation efforts and offset emissions that are more technically challenging to mitigate, it is important that land-use change emissions are minimised. At the production level, this can be achieved through practices such as maintaining yield growth to reduce demand for new cropping lands and thus avoid emissions associated with land-use change (Lobell *et al.* 2013; Herrero *et al.* 2016). This yield increase on currently cultivated land is dependent on increasing efficiencies and may be addressed to some extent by new production systems, such as intensive greenhouses run on renewable energy.

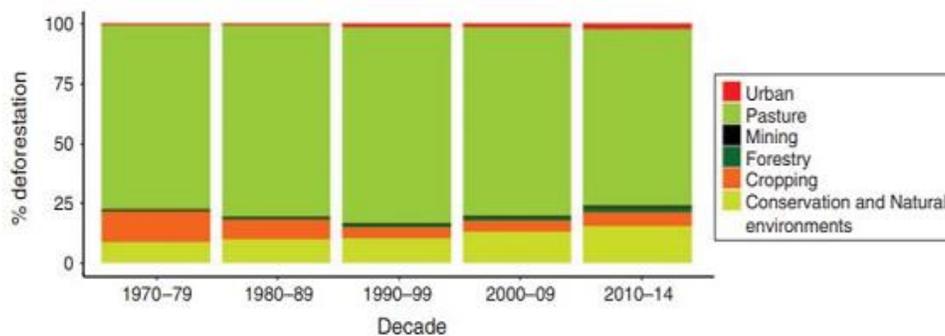


Figure 7: Percentage of total deforestation in each decade by land use. Data are sourced from ABARES (2010), figure from Evans (2016).

Land-use emissions from croplands and grasslands are also accounted for in the LULUCF sector. Annual variation in these emissions is influenced by climatic factors, particularly

rainfall, as well as changes in farming practices including fire and grazing management. Gradual reductions in these emissions over the last several years are at least partially due to areas converted to these land uses before 1990 slowly reaching new equilibrium carbon amounts (Commonwealth of Australia 2018d).

By increasing the efficiency of the production of feed (Hall *et al.* 2014; Wiedemann and Watson 2018) and the feed conversion efficiency of animals (Hristov *et al.* 2013b; Wiedemann *et al.* 2016d) industries can reduce emissions from intensive livestock systems through reductions in land-use change emissions associated with the croplands required to produce animal feed (Wiedemann and Watson 2018). In piggeries, optimal use of synthetic amino acids to reduce the amount of high protein ingredients such as soybean in feed would reduce environmental impacts. As more piggeries install anaerobic digestors, emissions associated with feed will become increasingly more significant (Wiedemann and Watson 2018). Emissions from egg and chicken systems could be greatly reduced if chicken manure was used as a fertiliser on plants grown on-site for chicken feed (Hall *et al.* 2014). Production systems using insects grown on food waste as a source of protein use less land than traditional feed but require large amounts of electricity. Use of renewable energy in these production systems would improve their potential to provide a GHG benefit (Salomone *et al.* 2017).

## Soil carbon sequestration

The potential for soils to sequester carbon in Australian agricultural systems depends on climate and land management factors (Post and Kwon 2000). In cropping systems, the few viable options for increasing soil carbon sequestration on farm (Young *et al.* 2009; Alcock *et al.* 2015) are using minimum tillage, stubble retention and rotations that include legumes. Incorporating the effects of these in life cycle assessments can have substantial impacts on the results. For instance, maize produced on farms with stubble retention incorporated had 56% less emissions than farms that burned stubble (Grant and Beer 2008). Reviews of studies on the effectiveness of these management options in cropping zones have found increases in soil carbon of about 0.2- 0.3 t C/ha/year. However, carbon accumulation declined with soil depth and over time (Sanderman *et al.* 2010; Lam *et al.* 2013). Importantly, when compared over time, soil carbon stocks in areas with improved management are often declining. Thus, these improvements are reductions in losses, likely still occurring from conversion of native land to cropland, and do not represent additional sequestration (Sanderman *et al.* 2010).

Improvements in carbon sequestration can be offset by increases in emissions elsewhere in the system. For instance, the gains in soil carbon can be negated by increased emissions of N<sub>2</sub>O associated with high carbon soils (Henderson *et al.* 2015; Palmer *et al.* 2017). Similarly, converting cropping lands to pastures increases soil sequestration, but also increases enteric methane emissions of farming systems due to livestock emissions. Soil carbon sequestration is less likely to offset the increased methane emissions in systems with high stocking rates and/or soils that are near the equilibrium soil carbon level (Meyer *et al.* 2016).

In addition to having to offset the emissions associated with increasing soil carbon, limitations include carbon accumulation being finite and reversible. Finite refers to observed increases levelling off as a new soil carbon equilibrium is reached. This can take some time, but eventually a given option will no longer increase soil carbon (Smith 2014). This means that in areas of well-managed, long-term pastures, which would have high soil carbon stocks, increases are unlikely (Eckard and Clark 2018). Soil carbon gains are also reversible, meaning they can be lost if improved management practices are not maintained or if soil carbon stocks are reduced due to climate factors such as drought (Smith 2014).

Furthermore, few of the options for increasing soil carbon can be implemented on Australian farms economically due to expenses around administration, monitoring and the verification of carbon stocks. As of June 2019, only 406 ACCUs for soil carbon had been issued (Clean Energy Regulator 2019b).

In contrast to cropping systems, there is substantial potential for increases in soil carbon on degraded pastureland. In 2002 it was estimated that 11.2% of Australian pasturelands were lightly overgrazed. Average sequestration rates in these areas were 0.09 t C/ha/year resulting in sequestration of 4.4 Mt C/year, or 16.1 Mt CO<sub>2</sub>eq/year, by eliminating overgrazing (Conant and Paustian 2002). Technical mitigation potential of rangeland rehabilitation varies from 0.1 to 0.2 Gt CO<sub>2</sub>eq globally, with Australia well represented (Henderson *et al.* 2015; Herrero *et al.* 2016). Given the extent of grazing land in Australia (54% of total land area), a small improvement across substantial portions of this area would result in large annual sequestration rates (Sanderman *et al.* 2010; Henry *et al.* 2015).

Increasing sequestration by 0.04 tonnes C/ha/year averaged over all grazing lands would lead to sequestration of 60 Mt of CO<sub>2</sub>eq/year (Sanderman *et al.* 2010) or 86% of current agricultural emissions. However, economic methods of increasing soil carbon levels at this scale and over remote areas are lacking. Maintaining soil carbon across all the grazing lands where it is high is important for mitigation as well as adaptation (Meyer *et al.* 2015). This can be accomplished by avoiding overgrazing, reducing bare ground cover, and reducing water and wind erosion (Eyles *et al.* 2015).

In some instances, planting perennial pasture species can lead to increases in soil carbon. In a Merino sheep system in southwestern Australia, planting the perennial kikuyu on 45% of the pasture allowed for stocking rates to be increased from 8.1 to 10.7 DSE while simultaneously reducing net emissions by 0.61 t CO<sub>2</sub>eq/ha/year, an 80% decrease. The mean carbon accumulation on five kikuyu pastures was 0.49 t C/ha/year (Thomas *et al.* 2012). However, perennial pasture species do not necessarily increase soil carbon compared to annual pastures (Chan *et al.* 2011).

Larger increases in soil carbon stocks can be obtained with some other practices, such as adding organic material to soils. In addition to increased soil carbon sequestration, there are benefits of applying compost to agricultural land (Recycled Organics Unit 2006) such as emissions reductions associated with replacing inorganic fertilisers (see CO<sub>2</sub>: pre-farm). However, limitations such as high application rates required, associated transport costs (Quilty and Cattle 2011), and risks such as sodium in manure (Sanderman *et al.* 2010) limit the use of these options in Australian systems.

## Above-ground carbon sequestration

Carbon can also be sequestered in above-ground biomass through reforestation and silvopastoral systems. Potential reductions in emissions are influenced by the site productivity, which is primarily driven by rainfall. Productivity influences both the potential of carbon sequestration and stocking rates. In some cases, it is possible for increased sequestration to compensate for higher stocking. On a high-rainfall site (1200 mm) in Queensland with approximately 0.5 head per ha, sequestration rates of 19.3 to 34.7 t CO<sub>2</sub>eq/ha/year allowed for trees planted on 7 to 13% of the holding to offset on-farm emissions (Eady *et al.* 2011b). In contrast, low sequestration rates (1.5 to 9.8 t CO<sub>2</sub>eq/ha/year) on a site with annual rainfall of 600 mm and a stocking rate of 0.1 head per ha, meant trees would need to be planted on 9% to 60% of the land to offset farm emissions. The higher uncertainty on the low-rainfall site is primarily due to variation in carbon sequestration data from the low-rainfall region (Eady *et al.* 2011b). In a study modelling the emissions from wool, prime lamb and beef systems, stocking rates of up to 22 DSE/ha could almost achieve carbon-neutrality when 20% of the farm was occupied by trees (Doran-Browne *et al.* 2018). Trees sequestered 2.3 and 2.7 t C/ha in environmental plantings and *Corymbia maculate* forestry, respectively. This resulted in about eight times more sequestration than occurred in soil. In a modelled case study based on an operational, high-performing farm, trees offset 48% of emissions over the study period (Doran-Browne *et*

*al.* 2018). In systems with lower stocking rates, fewer trees could offset all on-farm emissions (Doran-Browne *et al.* 2016). Savanna burning management increases sequestration rates in biomass (Murphy *et al.* 2010), as well as reducing emissions compared to late season fires (Maraseni *et al.* 2016).

Reforestation can also result in significant carbon sequestration. Several Australian studies estimate carbon sequestration potentials of reforestation in the range of 2 to 9 Mt CO<sub>2</sub>eq per M ha of plantings (Table 4 (Mitchell *et al.* 2012; Polglase *et al.* 2013)). The national inventory for 2016 reported a sink of 28.3 Mt CO<sub>2</sub>eq (Commonwealth of Australia 2019b) and the reported afforestation/reforestation estate in 2016 was 5.67 M ha (Commonwealth of Australia 2018c). This indicates an average sequestration rate of 5.0 Mt CO<sub>2</sub>eq per M ha. Environmental plantings over large land areas (11-20 Mha, or 12-24% of intensive agricultural land) are estimated to provide an average annual sink of 17.5 Mt

Table 4: Example sequestration rates from reforestation

Average sequestration rate (Mt CO <sub>2</sub> eq per M ha)	Notes and reference(s)
0.9-1.6	projection, (CSIRO 2019)
1.3	(Dean <i>et al.</i> 2012)
1.3 - 4.0	(Kirschbaum 2000)
5.0	(Commonwealth of Australia 2018c, 2019b)
8.1	projection, (ClimateWorks Australia 2010)
8.9	(Eady 2009)
3.7 -15	DCCEE report cited in (Mitchell <i>et al.</i> 2012)
29-61	Increases with increasing rainfall (Shea <i>et al.</i> 1998)

CO<sub>2</sub>eq to 2060 (CSIRO 2019). Maturation of regrowth forest had the fastest sequestration rate (0.36 t C/ha/year) in a Queensland rangeland study. Across 22.7 million hectares, 8.2 Mt C (30 Mt CO<sub>2</sub>eq)/year would be sequestered (Dean *et al.* 2012). Planting narrow belts of mallee on the 28% of Australian agricultural land considered degraded (value <\$2000/ha), resulted in an estimated sequestration of 17 to 26 Mt CO<sub>2</sub>eq (Paul *et al.* 2016). The importance of incorporating biodiversity as a priority in reforestation efforts has been repeatedly demonstrated (George *et al.* 2012; Paul *et al.* 2016; Reside *et al.* 2017). Any broad-scale effort at reforestation must co-occur with effective limits on deforestation (Evans 2018).

The financial viability of these options depends on several factors and requires more research. In grazed savanna woodlands of northern Queensland, both high and moderate stocking systems were estimated to serve as carbon sinks, but the moderately stocked system had more stable annual gross margins, greater accumulated gross margins, and a larger net carbon balance (Bray *et al.* 2014). Reforestation options in western Australia could become economically viable to land owners at carbon prices of AU\$15 (Harper *et al.* 2007), just slightly more than the average carbon price of the first five ERF auctions of \$11.83 (Mayberry *et al.* 2018). To date, 1,326,000 ACCUs have been issued under three reforestation and afforestation ERF methodologies (Baxter 2019). In contrast, a carbon price of \$132/ t C would be required to encourage planting trees in a prime lamb system in southwest Victoria, although this analysis did not include the financial co-benefit of trees (Sinnott *et al.* 2016). There are multiple co-benefits to incorporating trees into agricultural systems, including environmental improvements, production benefits from shade and wind breaks, and the large potential for diversification (Neufeldt *et al.* 2009). A lack of understanding of the on-farm co-benefits of incorporating trees is a known barrier to implementation (Kragt *et al.* 2017; Evans 2018). Other obstacles include policy uncertainty (Harper *et al.* 2017; Evans 2018), the lack of availability of trusted information, and risks associated with establishing trees such as tree death caused by insects or fires (Evans 2018).

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## 4.5 Fossil CO<sub>2</sub>: renewable energy and replacing fossil fuels

Use of renewables can have substantial impacts on emissions and reduce costs on farm. This is particularly relevant in systems with high electricity use such as irrigated systems, greenhouses, and other intensive systems. Large reductions in emissions could be achieved with technologies that reduce the need for traditional fertiliser, however many of these options are still in the research or demonstration phase. The use of bioenergy provides an opportunity for the agricultural sector to develop new markets and income streams. However, several issues arise from the use of agricultural land to provide bioenergy feedstocks.

## Pre-farm

Mechanisms for replacing fossil-fuel intensive fertiliser with more environmentally friendly options are being researched with at least a few ventures nearing commercial production. Several different avenues of research are ongoing including

- **Potential:** High
- **Applicability:** All agricultural sectors, particularly relevant to cropping
- **Stage:** Research through to pre-commercial

‘green’ fertiliser production (Government of South Australia 2019; Yara Pilbara 2019), nitrogen fixing bacteria (Pivot Bio 2019), bio-electrochemical nitrogen fixation that would allow farmers to make their own fertiliser (Liu *et al.* 2017; Milton *et al.* 2017), and genetically modified crops that can fix their own nitrogen (Vicente and Dean 2017). Examples of these technologies include two renewable ammonia production plants being developed in Australia, one in Port Lincoln (Government of South Australia 2019) and one in the Pilbara region of Western Australia. Yara Pilbara (2019) estimates that every kilogram of hydrogen would make 5.6 kg of ammonia and save over 5.5 kg of CO<sub>2</sub>. Pivot Bio, which applies nitrogen-fixing microbes in-furrow during planting, has shown increased productivity in corn and greater returns on investment than fertiliser, as well as decreases in N<sub>2</sub>O emissions. Other technologies are further from implementation and do not have estimates of mitigation potential. Given the contribution of fertiliser manufacture to cropping systems, successful development in this area would have widespread implications, including substantial reductions to emissions.

## On-farm

Use of renewables, such as solar and wind, on farm can have a large impact on the total emissions of intensive production systems (Bundschuh *et al.* 2017). A solar thermal greenhouse that grows 10-15% of

- **Potential:** High
- **Applicability:** High intensity systems with large fossil fuel use
- **Stage:** Established and ongoing

Australia’s truss tomatoes (Neales 2016) was reported to have the potential to save 14,000 tonnes of CO<sub>2</sub>eq per year (Allen 2015), which would be about 21% of the Australian tomato industry’s annual emissions of 66,000 tonnes CO<sub>2</sub>eq (Maraseni *et al.* 2010b). Use of renewable energy to power irrigation in cotton systems in Queensland, which cover over 100,000 ha, was estimated to reduce emissions by 1274 kg CO<sub>2</sub>eq/ha (Maraseni *et al.*

2010a). Irrigation powered by renewables would also have a large impact in vegetable production. For instance, the emissions associated with Tasmanian vegetables, where a large proportion of the energy mix is hydroelectricity, is 4.4 times less than mainland farmers. Renewable-powered irrigation is particularly relevant to vegetables such as fresh pod green peas, asparagus, French and runner beans, pumpkins, zucchini and button squash due to irrigation contributing over 70% of the EI (Maraseni *et al.* 2010b). In addition to emissions savings, renewable energy is associated with cost savings. For instance, a dairy manufacturer estimated a 20% costs savings by entering into a 10-year power purchase agreement where renewable energy was sourced from a wind farm (Scicluna 2018).

The mitigation potential of replacing fossil fuels with biofuels on farm is typically small, since this is not a large source of emissions in most systems. It could result in substantial reductions in emissions in systems with higher emissions from machinery, particularly horticultural crops like strawberries and lettuce (Gunady *et al.* 2012; Maraseni *et al.* 2012) in which machinery comprised 58% and 52% of emissions, respectively (Gunady *et al.* 2012). However, the largest potential for biofuels is by contributing to reductions in other sectors, including transport.

### *Agricultural waste to energy*

- **Potential:** High
- **Applicability:** High intensity systems
- **Stage:** Established and ongoing

Waste from many agricultural systems can be used to generate energy including animal manure and plant residues such as bagasse, olive waste, macadamia shells and wine grape waste. Energy can be generated through multiple bioenergy pathways, where feedstocks are

converted to bioenergy using different technologies. For example, biomass can be directly converted to generate energy (Yu and Wu 2010; Brooksbank *et al.* 2014; El Hanandeh 2015) or can undergo anaerobic digestion to produce biofuel (Waste Management Review 2016; Tucker 2018).

Much of Australia's bioenergy is currently produced from waste products. As of 2016-2017, Australia produces 3,501 GWh (1.4%) of its electricity from bioenergy, with 1,425 GWh of that from bagasse (Commonwealth of Australia 2018a). In addition, 440 million litres of ethanol is produced a year, primarily from a New South Wales plant that produces ethanol from waste flour (Rural Industries Research and Development Corporation 2019).

Like other renewable energy technologies, waste-to-energy systems provide economic benefit and emissions reductions (Brooksbank *et al.* 2014; Hall *et al.* 2014; Tucker 2018). Examples of energy, cost and emissions savings associated with these technologies are provided in Table 5. In the case of anaerobic digestors, the carbon benefit is two-fold, replacing grid electricity and preventing methane emissions through decomposition processes (Clean Energy Finance Corporation 2015). Large reductions in emissions can be associated with wastes replacing high-emission inputs. For instance, using bagasse instead of eucalyptus feedstock in pulp production results in greater energy savings than using bagasse in an anaerobic digester or sending it to a landfill with methane capture (Kiatkittipong *et al.* 2009).

Waste flour and biodiesel from waste oil are competitive with oil at \$40 US/barrel (O'Connell *et al.* 2007). That price increases to \$80 US/barrel when using sugar for ethanol or canola for biodiesel. Cereal straw is another potential feedstock for ethanol production. In Western Australia, 8 of 10 agricultural hubs examined could support a plant that requires 250,000 tonnes of feedstock a year while leaving 1 ton/ha on site for soil conservation. Each plant could produce up to 75 million litres of fuel ethanol per year (Brooksbank *et al.* 2014).

Factors that encourage the use of biomass include feedstock that are inexpensive and readily available, short transmission distances, and high costs of alternative power sources (Brooksbank *et al.* 2014). Similarly, factors determining the feasibility of biogas include the cost of gas and electricity and the availability of manure. Availability is primarily influenced by the number of livestock and the extent that manure can be collected. Biogas is more likely to be viable in large operations, with more than about 1000 cows or 500 sows. Climate is another factor that affects the efficiency and therefore viability of anaerobic digestion projects (Brooksbank *et al.* 2014).

Table 5: Benefits of a selection of waste to energy projects relevant to Australian agriculture

Location	Feedstock	GHG savings (CO <sub>2</sub> eq)	Economics	Notes
<b>Biomass energy generation</b>				
Australia <sup>1</sup>	Olive waste to pellets boilers	GHG reduction of 1057 kg/ Mg waste at mill	--	Current best practice for olive waste is composting which saves 12.4 kg
Victoria	Grape marc for steam & electricity	9813 t /yr 100% of fossil oils 69% of LPG	\$1.52 million/yr ROI in 4.5 yrs x10 value as	
Queensland	Macadamia shells for electricity	9500 t /yr	-	Produces 9.5 GWh/y, 1.4 GWh used on site
Victoria	Waste wood to heat greenhouses	Replacing coal briquettes, chosen over LPG	Initial cost x8 the cost of LPG, ROI in 2 years because fuel cost	
<b>Anaerobic Digestion</b>				
Western Australia <sup>2</sup>	Comm./indus. Organic waste	Avg of 7140 t/yr over 20 years		Co-products: Digestate and heat for blueberries
New Zealand	Piggery manure	50% of electricity needs, ROI in 3 yrs	Co-product: Heat to warm piggery	
Victoria	Piggery manure	740 t/yr	\$425,000 annual savings. ROI in 7 yrs	Co-product: potting mix, fertiliser. Co-benefit: water use reduction
New South Wales	Piggery manure	100% of electricity needs, ROI in 1.5 years including ACCUs, 3 yrs without		
Victoria	Dairy waste	100% of electricity needs, ROI in 3 yrs incl. RECs, \$600,000/yr savings <sup>3</sup>	Co-benefit: no untreated effluent going into local bays <sup>3</sup>	
Queensland	Chicken manure & organic waste	~ 6000 t/yr <sup>3</sup>	ROI in 5 to 7 years, \$250,000/ yr savings <sup>3</sup>	

Sources: 1= (El Hanandeh 2015), 2 = (Waste Management Review 2016), 3 = (Australian Government Rural Industries Research and Development Corporation 2019), all others = (Brooksbank *et al.* 2014)

## *Bioenergy from purpose grown crops*

There is a wide range of mitigation potential for various bioenergy options, with the net benefit dependent on the counterfactuals used for comparison, efficiency of feedstock production, and the pathways used to process the feedstock. For instance, intensive processes such as pelletisation reduce the net benefit (Cherubini *et al.* 2009;

Welfle *et al.* 2017). Thus, use of bioenergy can result in small improvements in emissions compared to fossil fuels (VIEWLS 2006; Brooksbank *et al.* 2014) through to net negative emissions where the process results in an overall sequestration of carbon (Yu and Wu 2010; Campbell *et al.* 2011). Producing biodiesel from waste oil (Beer *et al.* 2007) and some algae systems (Campbell *et al.* 2011) results in large reductions in emissions compared to fossil fuels (>70%). Producing electricity from mallee bioslurry resulted in net sequestration of 3.2 kg CO<sub>2</sub>eq per GJ (Yu and Wu 2010). Bioenergy can emit more GHG per unit of energy produced than fossil fuels. For example, biodiesel derived from palm oil grown on cleared rainforest and peat swamp forest was associated with much higher emissions than diesel, with emissions 8 times greater than diesel on cleared rainforest and 21 times more on peat swamp forest (Beer *et al.* 2007). This highlights the importance of land-use change emissions in determining the net GHG emissions of bioenergy.

- **Potential:** Uncertain, and dependent on other priorities and technology development
- **Applicability:** Most land sectors, particularly relevant to cropping and modified pasture systems
- **Stage:** Research through to commercial-level production

The sustainability of biofuel includes several factors, many of which are focused on feedstock production. Inputs, particularly fertiliser, land-use considerations, and biodiversity implications are important factors to consider in evaluating the sustainability of feedstocks used for biofuel, and bioenergy more generally (Tilman *et al.* 2006; Robertson *et al.* 2017). Sustainable feedstocks require no irrigation and little to no fertiliser, are grown on degraded lands that do not require clearing of forest or filling of wetlands. Emissions from biodiesel produced from canola were dominated by fertiliser production (29%) and CO<sub>2</sub> emissions from urea hydrolysis (25%) (Biswas *et al.* 2011). Fertiliser is also associated with other environmental consequences including leaching. Low-input high diversity (LIHD) mixtures of grassland perennials in the United States do not require substantial inputs (Tilman *et al.* 2006). Substantial biodiversity benefits can be realised with small amounts of plant diversity. Identifying native species that are highly productive on marginal lands where abiotic

stressors often limit growth is a key requirement to providing sustainable bioenergy feedstocks (Robertson *et al.* 2017). In Australia, mallee has the potential to meet these requirements. When planted in cropping areas, yield losses are at least partially offset by increases in production due to improvements in dryland salinity (Yu and Wu 2010).

Land-use change is an important consideration for food security and net emissions (Robertson *et al.* 2017). Use of cropping lands for non-food crops can increase food prices or result in land-use change elsewhere (O'Connell *et al.* 2007; Robertson *et al.* 2017). As mentioned previously, conversion from high-carbon lands to biofuel production can result in greater net emissions than fossil fuel use (Beer *et al.* 2007). Establishment on degraded lands addresses these concerns by avoiding competition with food production and allowing for soil carbon sequestration (Tilman *et al.* 2006; Robertson *et al.* 2017). However, if these lands are being grazed, livestock will have to be removed to support biomass production. It was estimated that between 15,000 and 31,000 cows would have to be removed from the Fitzroy basin in Queensland to support biomass production that would produce 5% of aviation fuel demand, although prices received would be much greater with biofuel than cattle (Hayward *et al.* 2015). Careful land management is also required. Adequate residues must be left after harvest to prevent the loss of carbon from soils (Sanderman *et al.* 2010). Soil carbon sequestration and soil fertility can be improved in some systems, for instance when planting perennial feedstocks (Hansen *et al.* 2004), or could be negatively impacted with removal of stubble reducing carbon inputs into the soil (Blanco-Canqui and Lal 2009).

A strong bioenergy industry provides many benefits, including economic benefits particularly favourable for the agricultural industry. Co-benefits include providing baseload power, reducing waste going to landfill, and reducing pollution from particulates, sulphur dioxide and nitrogen oxides (Clean Energy Finance Corporation 2015). Economically, bioenergy provides new revenue streams and expansion into valuable new markets. Bioenergy plants provide regional employment opportunities (Hayward *et al.* 2015; Campey *et al.* 2017). For instance, a case study of a sugar-to-ethanol plant reported 36 permanent jobs and 222 flow-on jobs were created with \$7.7 million added to household income in the region (O'Connell *et al.* 2007). Plants can be owned by farmer' cooperatives as occurs with the ethanol industry in the United States (Brooksbank *et al.* 2014). Co-products of bioenergy production include livestock feeds and high-value chemicals and plastics, increasing income diversity (O'Connell *et al.* 2007; Deloitte Access Economics Pty Ltd and Consulting 2014). Co-products such as distillers grain can be fed to livestock with the potential of reducing enteric methane emissions (O'Connell *et al.* 2007). Biochar is a co-product with potential in

soil remediation, including improving pH and nutrient availability in Ferrosols, managing acid-sulphate soils, and in bauxite rehabilitation (Macdonald *et al.* 2016).

The potential of first-generation biofuels to substitute fossil fuels in the Australian transport sector is limited. If currently exported wheat and coarse grains were used to produce ethanol it could supply 11-22% of Australia's 2007 petrol use. If domestic oil waste and tallow and oilseed exports were used to produce biodiesel it would supply 4 to 8% of Australia's 2007 diesel use. Using wheat to meet a national E10 target would force the import of wheat in drought years (O'Connell *et al.* 2007). Estimates of the potential of advanced biofuels to supply Australian petrol use range from 10% to 140%, due to a lack of knowledge regarding the sustainability and economic feasibility of production using lignocellulose feedstocks (O'Connell *et al.* 2007). The scale-up required to meet 5% of aviation fuel demand with biofuels has been investigated for the Fitzroy region in Queensland (Hayward *et al.* 2015).

Although some feedstocks and processes are already competitive (Clean Energy Finance Corporation 2015), costs can limit implementation (Campey *et al.* 2017). Transport costs limit the locations in which plants can be run economically (O'Connell *et al.* 2007). In some cases, high production levels are required to be competitive (Campbell *et al.* 2011).

Policy guidance regarding expanding the use of biofuels is wide-ranging and includes retaining the discount on excise and applying it to farm vehicles, increased education and promotional activity, encouraging capital investment, as well as federal and state procurement programs and mandates focused on advanced biofuels (O'Connell *et al.* 2007; KPMG 2018). Queensland's vision and associated 10-year roadmap provides an example of policy settings that promote bioenergy development (Glenn 2017). On the logistical side, providing contractual linkages across the supply chain allows for sharing of risk and enables investment in feedstock conversion facilities. Research is needed on the best mechanisms for achieving this, including long-term agreements for land owners to provide feedstock (Hayward *et al.* 2015). Funding at the demonstration level is particularly needed to bridge the gap between small scale pilot projects and commercial level production.

## 5. Scenarios

This section illustrates emissions outcomes in 2030 under four scenarios: 1) retaining the status quo; 2) using a cost-effective option; 3) employing best practice; and 4) applying an optimistic emissions reduction target. The emission targets for 2030 are relative to 2017 values. The calculations include land-use change as well as energy and fuel emissions. A primary assumption is that increases in emissions associated with increased production are the same across emissions sources. Assumptions regarding the mitigation potentials for the scenarios (Table 6) consider the possible effectiveness of available options as well as how widespread they can be applied. For instance, it is likely that there will be enteric methane inhibitors on the market by 2030, however it is less likely these can be applied in extensive systems. Mitigation potential is based on the reviewed literature including the effectiveness of mitigation options in New Zealand (Reisinger *et al.* 2018). The percentage reductions are not directly transferable as intensive systems (e.g. dairy) contribute a larger percentage of New Zealand agricultural emissions than intensive systems in Australia.

Table 6: Assumptions associated with the 4 emissions scenarios for 2030. Percentages are reductions from 2017 except for emissions growth

Scenario characteristics	Status quo	Cost-effective	Best practice	Optimistic
Emissions growth	2%/year	0.8%/year	0.5%/year	None
Deforestation	No change	20%	50%	95%
Livestock methane	No change	No change	10%	25%
Manure management	No change	10%	40%	60%
Agricultural soil emissions	No change	No change	20%	40%
Other emissions (urea, liming, field burning)	No change	No change	10%	20%
Grazing land management (LULUC CO <sub>2</sub> )	No change	No change	50%	90%
Energy and fuel emissions	No change	20%	50%	90%
Reduction in overgrazing	No change	No change	50% (8 Mt)	95% (15 Mt)
Trees on farm (% of on-farm emissions offset)	3% (3 Mt)	10% (9 Mt)	20% (14 Mt)	50% (26 Mt)
Reforestation of agricultural land	4 Mt	7 Mt	11 Mt	14 Mt

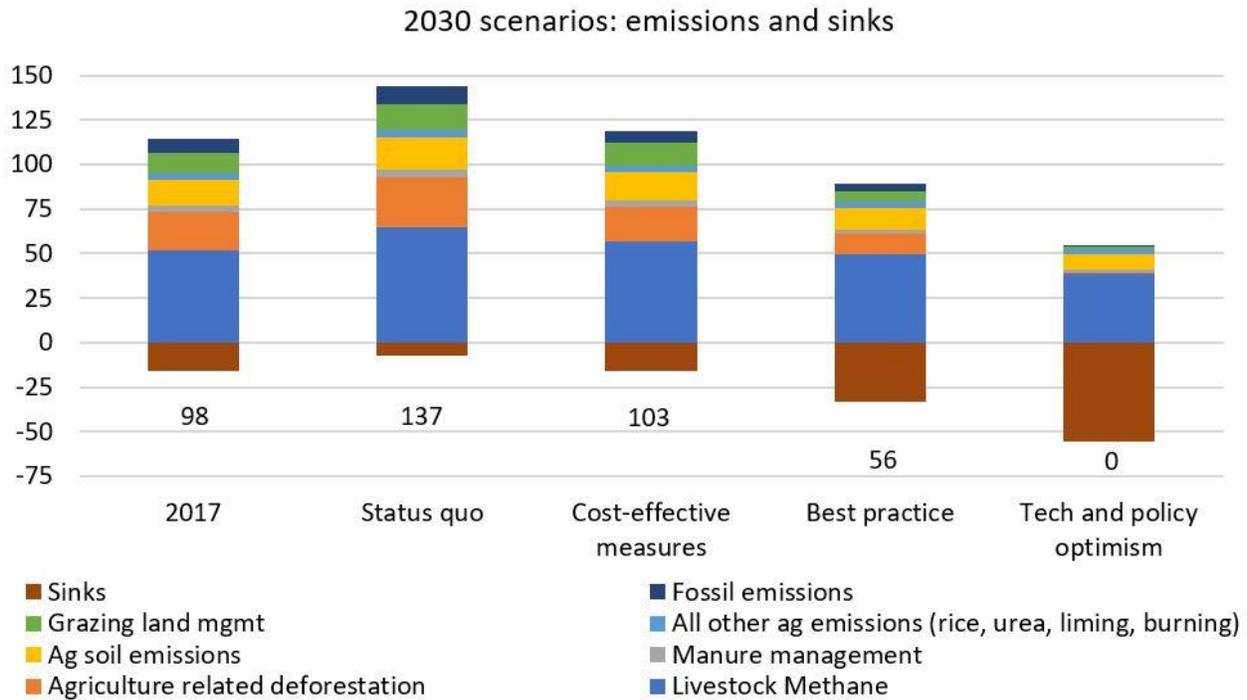


Figure 8: The emissions and sinks associated with the scenarios described in the text. Numbers under the columns are the net emissions in Mt CO<sub>2</sub>eq

There are also overlaps between some categories that are not explicitly addressed. For instance, improvement in carbon emissions from grazing land use (LULUCF sector) and the avoided emissions from ceasing overgrazing would be correlated. This is generally reflected in the scenarios by ensuring reduction of overgrazing is concurrent with reduction in grazing land-use emissions. Double counting the sink from trees on-farm and reforestation is also a risk. This is addressed by including the sink assumed from trees on farm within the total afforestation/reforestation sink reported.

The total sink from reforestation and afforestation on all Australian lands has been stable at about 20 Mt CO<sub>2</sub>eq for several years. There would be a total sink of 16 Mt CO<sub>2</sub> assuming 80% of this afforestation/reforestation sink occurs on agricultural land and trees on farm are part of this total sink. The current afforestation sink is primarily from plantations created since 1990 (Commonwealth of Australia 2013). Establishment of plantations since the 1980s has occurred primarily on farmland (Commonwealth of Australia 2017b), mostly in northern Australia rangelands (Mayberry *et al.* 2018). Thus, 80% of the current land sink is used as a conservative estimate of agricultural land’s contribution. The maximum total annual sink assumed from agricultural land is 40 Mt CO<sub>2</sub>eq in the optimistic scenario. Table

5 displays the assumptions for each of the scenarios and Figure 8 shows the emissions and sinks for each scenario.

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## 5.1 Status quo

If there is little change in the near- to medium-term, it can be assumed that previous estimates of agricultural emissions projections are relevant. Currently available estimates include increases in Australian agricultural emissions by 2030 of 11.4% (Commonwealth of Australia 2017a) and 43.6% (The Centre for International Economics 2013). These estimates reflect increasing demand, agricultural production, and recent trends in EI improvements. Assuming an annual growth in agriculture-associated emissions between these two estimates of 2% and no additional mitigation, emissions in 2030 would be 144 Mt CO<sub>2</sub>eq, including land-use change and fossil-fuel emissions from agricultural production.

The potential sink in this case is limited. Currently there are few farms where emissions are substantially offset by trees. Given the lack of plantation establishment, the reforestation sink is projected to decline to 5 MtCO<sub>2</sub>eq by 2030. However, vegetation regeneration on grazing lands could provide 3 Mt CO<sub>2</sub>eq (Commonwealth of Australia 2013). If 80% of the reforestation is assumed to occur on agricultural land (4 MtCO<sub>2</sub>eq) this is a total sink of 7 Mt CO<sub>2</sub>eq, resulting in a net emission of 137 Mt CO<sub>2</sub>eq.

Assuming no change in bioenergy use (1.4% of Australia's 2016-2017 energy use), and that carbon efficiency of replaced fossil fuels averages 40 kg CO<sub>2</sub> per GJ, agriculture also contributes to avoided fossil emissions in other sectors of 0.5 Mt CO<sub>2</sub>eq. This avoided fossil emission contributes to a reduction in national GHG emissions but is not incorporated in the net balance calculation of the scenarios due to the focus on land sector sinks offsetting land sector emissions (Parliamentary Commissioner for the Environment 2019).

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## 5.2 Cost-effective measures adopted

In this scenario, the application of technologies such as renewable energy for intensive systems and anaerobic digestion for piggeries and intensive dairies become the norm. If cost-effective, established technologies are adopted and adaptation allows for compensation of any further impacts of climate change on yields, EI would improve compared to the recent past. If this is assumed to result in a lower increase in emissions due to increased demand

(0.8%, low end of increases reported in literature (Commonwealth of Australia 2017a)) as well as a reduction in agricultural related deforestation of 20%, mitigation of 10% of total manure emissions from improvements in intensive systems, and 20% of fossil fuel emissions due to uptake of renewables on-farm, then 2030 emissions associated with agricultural activities are 119 Mt CO<sub>2</sub>eq.

Assuming revegetation methodologies in the ERF and offsetting 10% of on-farm emissions through sequestration in soils and trees can compensate for the declining sink of plantations established after 1990, there would be a sink of 16 Mt CO<sub>2</sub>eq and a net emission of 103 Mt CO<sub>2</sub>eq. Achieving a sink of this size from trees by 2030 would require tree establishment immediately or in the near-term. Assuming an increase in bioenergy production to 2% of 2016-2017 energy requirements, agriculture would also contribute to avoided emissions in other sectors of 0.7 Mt CO<sub>2</sub>eq.

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### 5.3 Best-practice widely implemented

In this case, the cost-effective strategies above have been implemented as well as options that are currently close to commercial development, such as 3-NOP, and options that are associated with some cost but have clear co-benefits, such as planting trees on-farm. Further efficiencies in production reduce the growth in emissions to 0.5% year. Increased efficiency and policy intervention result in a reduction in deforestation of 50%. Improved land management results in a reduction of carbon emissions from grasslands of 50% (LULUCF emission). Emissions from fossil fuels used in agriculture are reduced by 50%. Mitigation of total enteric methane emissions of 10% is assumed from the use of 3-NOP or similar product in intensive production systems. Options such as precision agriculture, nitrification inhibitors and other currently available N<sub>2</sub>O mitigation technologies are widely implemented and lead to a reduction in soil emissions of 20%. It is assumed that a substantial reduction in manure methane emissions from intensive systems, which comprises well over 50% of national manure methane emissions (Commonwealth of Australia 2018c), results in a 40% reduction in these emissions. Lastly, it is assumed that reductions in field burning reduce other agricultural emissions by 10%. With these assumptions total emissions associated with agricultural activities are 89 Mt CO<sub>2</sub>eq.

Assuming that overgrazing is ceased on 50% of agricultural land (8 Mt CO<sub>2</sub>eq) (Conant and Paustian 2002), that 20% of on-farm emissions (14 Mt CO<sub>2</sub>eq) can be offset by trees, and that the sink for reforestation is increased by 50% over the cost-effective strategy (11 Mt

CO<sub>2</sub>eq) then there would be a total sink of 33 Mt CO<sub>2</sub>eq. The assumptions regarding revegetation, including trees and soils on farms, lead to a reforestation/afforestation sink similar to the early 2010s peak associated with plantations established after 1990 (25 Mt CO<sub>2</sub>eq). From 1990 to 2000 an average of about 40,000 ha per year were established. This increased to an average of about 80,000 ha per year in the early 2000s and declined following a 2007-2009 average of about 60,000 ha per year (Commonwealth of Australia 2013). Net emissions associated with agriculture in this case are 56 MtCO<sub>2</sub>eq in 2030. Assuming an increase in bioenergy production to 5% of 2016-2017 energy requirements, agriculture would contribute to avoided emissions in other sectors of 1.8 Mt CO<sub>2</sub>eq.

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## 5.4 Technological and policy optimism

In this case, efficiency gains have offset increases in demand, leading to no increases in emissions despite increased production. Anaerobic digestion and use of other low emission options have led to a reduction in manure management emissions of 60%, and a 20% reduction in other agricultural emissions is achieved. Large increases in the use of renewables, and technological developments in fertiliser production have resulted in a 90% reduction in fossil-fuel use. Deforestation is reduced to 95% of that occurring in 2017. Widespread application of best land management practices has reduced carbon emissions on grazing land by 90%. Precision agriculture and other technologies allow for a reduction in soil emissions of 40%. An overall reduction in enteric methane emissions of 25% is achieved. This occurs in intensive industries through the widespread use of 3-NOP or some other technology. In extensive systems enteric methane is reduced by adoption of some technological breakthrough such as a vaccine or slow-release inhibitor (Reisinger *et al.* 2018), by undergoing a reduction in ruminant livestock numbers, or some combination of the two. These assumptions result in agricultural-related emissions of 55 Mt CO<sub>2</sub>eq.

Given the improvements in land management, a 95% reduction in overgrazing is assumed (15 Mt CO<sub>2</sub>eq), as is a 50% offset of on-farm emissions through carbon sequestration (26 Mt CO<sub>2</sub>eq). If this could co-occur with another 30% increase in the sink from afforestation on agricultural land to 14 Mt CO<sub>2</sub>, then the total sink is 55 Mt CO<sub>2</sub>, which results in net-zero agricultural emissions. The feasibility of achieving this amount of carbon sequestration (40 Mt CO<sub>2</sub>eq) through trees on-farm and reforestation of farmland while maintaining high production levels is unclear. Certainly, to make the on-farm and land use change required to achieve this sink within 10 years is a considerable challenge. Reforestation of marginal lands and strategic reforestation of non-marginal land has been estimated to provide a sink of 45

Mt CO<sub>2</sub>eq at a carbon price of \$26-27/tonne (ClimateWorks Australia 2010). Another report suggests limited revegetation can provide a sink of nearly 50 Mt CO<sub>2</sub>eq (Longmire *et al.* 2014). However, the impact of this amount of tree planting on farm productivity is not well understood. In at least some locations, the low quality of the land being planted to trees and the co-benefits could result in a net benefit to production (Yu and Wu 2010). If bioenergy production increases 10-fold by 2030, contributing 14% of 2016-2017 energy requirements, the agricultural sector would also contribute to avoided emissions in other sectors of 5 Mt CO<sub>2</sub>eq.

## 6. Conclusions

Various industries have targets for reducing emissions. The National Farmers Federation aims for the agricultural sector to be trending toward carbon neutrality by 2030. This would include no deforestation, 50% of on-farm energy from renewable sources, a 30% reduction in waste, substantial reductions in enteric methane and a carbon market that provides an income of \$40 billion to the land sector by 2050 (National Farmers' Federation 2018). The dairy industry estimates that increasing the efficiency of all dairies to the level of the dairies that currently have the lowest EI could reduce emissions by 21% by 2030 (Dairy Australia 2018). The Clean Energy Finance Corporation estimates that an investment of \$3.5 to 5 billion into waste-to-energy could result in avoiding 9 Mt CO<sub>2</sub>eq annually from the waste, forestry, and agricultural sectors.

However, there are several obstacles to implementing mitigation options on farms. The economics of several options prevent their widespread adoption. In addition to the difficulties of installing new infrastructure or implementing new techniques on farm, receiving carbon credits for mitigating emissions requires expertise needed to navigate the process and includes costs associated with emission reduction estimation, reporting, and verification (Cowie *et al.* 2012; van Oosterzee *et al.* 2014). This impedes adoption of options such as destocking of overgrazed areas, which has a high potential to increase the land sink but reduces income.

Several knowledge gaps hinder the ability to achieve net-zero or negative emissions. Due to the substantial contribution of enteric methane from extensive systems and the widespread use of fertilisers across agricultural industries, methods that could reduce these emissions would have the greatest impact on total agricultural emissions. Other research that could address N<sub>2</sub>O emissions include determination of the sources of variation in effectiveness of inhibitors and the whole-farm emission consequences of alternative fertilisers in Australian conditions. In addition, investigations are needed on the potential environmental benefit of alternative systems of production such as synthetic meat, forage supply chains based on algae or food waste, and organic farming. In developing a bioenergy industry, the sustainability of various processes and feedstocks, particularly cellulosic feedstocks, need to be determined for the Australian context. Determining effective ways to promote healthy and sustainable diets globally could result in substantial emissions reductions. Integrated

assessments that address the multidimensional impacts and potential trade-offs associated with mitigation options are needed (de Boer *et al.* 2011). This information could inform priorities both in policy development and on-farm decision making.

Research is also required to understand how land-use decisions are made and how those decisions are influenced by social, political and economic drivers (Bustamante *et al.* 2014). Precise valuing of co-benefits would assist the economic case of adding trees to farms or reforesting current agricultural areas. Policies restricting deforestation are required for continued reductions in these emissions. If priorities are to focus on degraded lands, for reforestation or bioenergy production, information on what species could achieve the desired outcomes on those lands will be important. More broadly, determining the sequestration potential in various areas and comparing the benefits and trade-offs between different land-use options (remaining food producing, converting to bioenergy or converting to forest) would assist in farmer decision making.

Efficient use of land resources and effective adaptation will assist in meeting increasing demand (Bustamante *et al.* 2014; Reisinger *et al.* 2017). Insufficient or maladaptation will lead to reduced yields, less improvement in EI, and potentially increased land-use change due to agriculture (Lobell *et al.* 2013). More integrated strategies at a landscape scale can address multiple threats including emissions, catchment disturbance and biodiversity loss (van Oosterzee *et al.* 2014). This would be assisted by consistent and integrated policies across Federal and State agencies. Incentives for reducing food waste both before and after the farm gate, would improve EI and provide several co-benefits such as increasing food access through charitable donations.

The basic scenarios presented here suggest that net-zero or negative emissions from the agricultural sector are possible in the next 10 years. The task is a challenging one, requiring rapid and widespread implementation of currently available mitigation methods, substantial investments in research and development, and incentives to encourage options that are not currently commercially viable. Although requirements for sustainable supply chains will increasingly become a driver in farmer decision-making (Rawnsley *et al.* 2018), consistent policy interventions would speed the transition of the agriculture sector toward net-zero emissions, and ultimately net negative emissions in the longer-term.

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