School of Civil and Mechanical Engineering

Carbon Footprint Analysis of Broadacre Livestock Grazing Systems in Southern Australia

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Declaration

To the best of my knowledge and belief this thesis contains no material previously published by any other person except where due acknowledgement has been made.

This thesis contains no material which has been accepted for the award of any other degree or diploma in any university

Signature: Danielle Gale

Date: 1 September 2020

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I dedicate this work to my grandparents – each of whom have played such an important role in shaping me into the person I am today.

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Abstract

The livestock sector is one of the largest sources of global anthropogenic greenhouse gas (GHG) emissions, contributing around 15% of total emissions. Emissions from the Australian broadacre livestock sector mirror these global trends, with projected increases in domestic and international demand for meat products likely to result in concurrent increases in production and emissions. The dependence of southern Australian livestock producers on dryland pastures means that they will also be exposed to projected changes to climate. Recognition is now given to the need to improve livestock system productivity while minimising the environmental impact of those systems. The challenge for the sector now, is to overcome the current barriers to adoption of mitigation strategies by identifying and promoting practices that reduce emissions without hindering overall productivity.

The goal of this study was to investigate the carbon footprint of broadacre livestock production in south-western Australia under different pasture systems and identify productive and regionally appropriate strategies with mitigation potential for on-farm uptake. To achieve this, the study incorporated an integrated approach that involved sheep and beef cattle case study farms, biophysical modelling and life cycle assessment (LCA) methodologies. The development of comprehensive carbon footprint frameworks to account for this integrated approach enabled a cradle to farm gate analysis of the emissions associated with the production of one kilogram of saleable liveweight (kg CO_2-e/kg LW). This in-depth framework allowed for detailed analyses of the whole-farm system, including the role of different pasture systems, stock classes, intra-annual variations and farm management practices relating to the carbon footprint of the enterprise in question. Importantly, it also enabled targeted investigations into the mitigation potential of identified strategies.

The carbon footprints of the examined sheep production enterprises ranged from 8.18 to 10.60 kg CO_2-e/kg LW produced for sale, while the footprints of the beef cattle enterprises ranged from 9.17 to 13.20 kg CO_2-e/kg LW. Across all, the primary hotpot was enteric methane production, contributing between 74.4% to 85.1%, whilst the secondary hotspot was either nitrous oxide from crop residue, excreta nitrous oxide, or emissions from the production of inputs, depending on the considered system. The breakdowns of the whole-farm carbon footprints across feedbases, stock classes and months of the production year revealed that perennial pasture systems were typically the most emissions efficient of the grazed feedbases (6.32 to 15.38 kg CO_2-e/ kg LW), followed by annual pasture (8.10 to 14.21 kg CO_2-e/kg LW) and then crop stubble, if

present (15.05 to 38.57 kg CO_2-e/kg LW). Feedlots, where present, were more emissions efficient, or equivalent, to perennial pasture (2.05 to 6.69 kg CO_2-e/kg LW), reflecting the high feed conversion efficiency of feedlot finished stock. The analyses revealed that the EI of a feedbase was determined by reproductive and grazing management, which were in turn, determined by the attributes of the feedbase itself.

Importantly, this research determined that perennial pasture systems can reduce the carbon footprint of livestock production. This occurs not through a direct reduction of emissions, but rather through improved farm productivity. The provision of out-of-season feed enables farms to employ productivity-driven farm practices such as accelerated joining and the backgrounding and agistment of stock alongside a farm breeding herd, while also reducing supplementary feeding requirements. Alongside the carbon footprint benefits of perennial pastures, this research also demonstrated that there are reproductive and grazing management practices with both on-farm productivity and emissions benefits for southern Australian livestock systems. The mitigation potential of such practices varied across both the practice and livestock enterprise under consideration, ranging from 0.4 to 20.8%, highlighting that the magnitude of mitigation will be farm-specific, dependent on the characteristics of the farm in question.

This study contributes significantly to the existing body of knowledge in the field of carbon footprint analysis and sustainable livestock production, both in Australia and internationally. This research was the first of its kind to consider the carbon footprint of beef cattle production and the second to consider sheep meat production in south-western Australia. Within Australia, it was the first study to conduct an intra- and inter-farm carbon footprint comparison of perennial and annual pasture systems. The study makes a significant contribution to the current knowledge gap in Australia with regards to whole-farm carbon footprint mitigation analysis. The research also highlighted the need for further streamlining of methodological approaches and improved data quality controls in livestock carbon footprints and LCAs. Importantly, this research emphasises the importance of farm-level analyses in recognition of the heterogeneity of livestock production systems and the fact that mitigation options will be farm-specific. The approach conducted in this study demonstrates the depth of intra-farm analysis possible in carbon footprint research and should represent a pathway to initiating the transition to more the carbon-efficient production that the industry has been trying to achieve.

Publications relating to the thesis

Conferences

Gale, D., Biswas, W. & Pritchard, D. (2017) *Reducing the carbon footprint of livestock production in the Mediterranean region of Southern Australia*, Presented at the 3rd International Conference on Global Food Security, Cape Town, South Africa, 3-6 December 2017. Conference programme.

Media Coverage

Landgrafft, T. (2015) *Carbon mitigation PhD student hopes research will ensure food security*, ABC Rural, Western Australia, 27 October 2015. Website: https://www.abc.net.au/news/rural/2015-10-27/curtin-phd-student-hopes-research-will-ensure-food-security/6877964

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List of abbreviations

ABARES	Australian Bureau of Agricultural and Resource Economics
ABS	Australia Bureau of Statistics
AFI	Australian Farm Institute
AI	Active ingredient
BOM	Bureau of Meteorology
CER	Clean Energy Regulator
CFI	Carbon Farming Initiative
CH_4	Methane
CO_2	Carbon dioxide
CO_2-e	Carbon dioxide equivalent
CP	Crude protein content
CPI	Crude protein intake
CSIRO	Commonwealth Scientific and Industrial Research Organisation
CY	Clean yield percentage
DAP	Direct Action Plan
DAP	Diammonium phosphate
DEE	Department of Environmental and Energy
DISER	Department of Industry, Science, Energy and Resources
DPIRD	Department of Primary Industries and Regional Development
dLUC	Direct land use change
DM	Dry matter
DMA	Dry matter availability
DMD	Dry matter digestibility
DSE	Dry sheep equivalent
EDM	Edible dry matter
EF	Emission factor
EI	Emissions intensity
ERF	Emissions Reduction Fund
EQ	Emissions quantity
FAO	Food and Agricultural Organisation of the United Nations
FU	Functional unit
GHG	Greenhouse gas
GW	Greasy wool production
GWP	Global warming potential
HI	Harvest index

IPCC	International Governmental Panel on Climate Change
ISO	International Organisation for Standardisation
LCA	Life cycle assessment
LCI	Life cycle inventory
LEAP	Livestock Environmental Assessment and Performance Partnership
LU	Land use
dLUC	Direct land use change
LW	Liveweight
LWG	Liveweight gain
MA	Milk production
MAP	Monoammonium phosphate
MC	Milk consumption
ME	Metabolisable energy
MLA	Meat and Livestock Australia
MP	Milk production
MSDS	Material Safety Data Sheet
N	Nitrogen
NAR	Northern Agricultural Region
NGGI	National Greenhouse Gas Inventory
NIR	National Inventory Report
N_2O	Nitrous oxide
NPP	Net primary productivity
PI	Potential intake
PMA	Protein mass allocation
RI	Relative intake
SAMM	South African Meat Merino
SRW	Standard reference weight
tkm	tonne-kilometre
UNFCCC	United Nations Framework Convention on Climate Change
WP	Clean wool production

1 INTRODUCTION

1.1 Introduction

This thesis aimed to quantify the carbon footprint of broadacre livestock production in south-western Australia, with focus on the influence of pasture systems, through the application of an integrated approach developed specifically for this research.

1.2 Background

The pathway to achieving food security is one of the greatest challenges facing our global community. The world's population is projected to increase to more than 9.6 billion by 2050 with a corresponding predicted increase in global food demand of more than 70%, as compared to 2010 levels (Gerber et al., 2013). Changing consumption patterns, driven by urbanisation and rising incomes in developing countries, will see shifts towards diets with more meat, further driving up the demand for livestock products (Alexandratos & Bruinsma, 2012; Steinfeld et al., 2006). Exactly how to increase livestock production to meet this demand in an environmentally, economically and socially sustainable manner is a complex and multi-faceted issue.

Globally, livestock production accounts for approximately 80% of agricultural land and is predominantly utilised by extensive, grass-based ruminant production systems (Herrero et al., 2015). Historically, increased agricultural production has largely been accomplished through increased arable land use. However, land availability is limited and competing demands for other commercial and ecological use mean that potential land expansion to 2050 will be limited to 5% of current arable land allocated to agriculture (Alexandratos & Bruinsma, 2012; Tilman et al., 2001). Furthermore, expansion opportunities exist just a few countries, namely in Latin America and sub-Saharan Africa. This land scarcity, along with the increasing scarcity of other resources such as water, is driving the intensification of livestock systems to answer the demand for livestock products.

Intensification plays a key role in global livestock production, driving the almost four-fold increase of meat production observed over the past 40 years (Alexandratos & Bruinsma, 2012; Steinfeld et al., 2006). Early efforts to intensify livestock production focussed on increasing productivity per animal (Herrero et al., 2015). This was largely aided by the "Green Revolution" which saw increased crop yields, including crops produced for livestock feed, and the advent of managed pastures. Improvements were aided in part by the transition to a higher input system, with the use of fertilisers

and pesticides, and an increased reliance on fossil fuels and irrigation (Hochman et al., 2013). More recently technological and animal breeding advances have further intensified livestock production systems (Thornton, 2010). However, the high growth rates in production attained in the late 20th century have slowed and higher yields are likely harder to attain (Hochman et al., 2013). In addition, productivity gains experienced by the sector have not been attained in an ecologically sustainable manner. The role of livestock as a major contributor to a suite of environmental issues is considered a matter of urgency. Future intensification of the sector must occur innovatively, and critically, in an environmentally sustainable manner.

The contribution of the livestock sector to environmental degradation is well-established. The resource-intensive nature of livestock production means that it is in direct competition with other sectors for increasingly scarce resources such as land and water. In addition to this, livestock are major contributors to land degradation, water pollution, biodiversity loss and climate change (Rojas-Downing et al., 2017; Steinfeld et al., 2006). In particular, the sector is one of the largest sources of global anthropogenic greenhouse gas (GHG) emissions, producing 14.5% of total anthropogenic emissions and between 44 to 53% of global methane and nitrous oxide emissions, respectively (Gerber et al., 2013). Beef and sheep meat production are responsible for 44% of total emissions from the global livestock sector, with dairy production contributing a further 21% (Alexandratos & Bruinsma, 2012). As well as being net emitter of GHG emissions, the livestock sector is particularly vulnerable to the current and projected effects of climate change. Changing weather pattens and an increase in severe weather events will affect feed quality and availability, water availability, the persistence of certain livestock species, incidence of pests and diseases, for example (Ghahramani et al., 2019; Henry et al., 2018a; Rojas-Downing et al., 2017). However, the actual impact of climate change on livestock systems will vary, both regionally as a result of changes to climatic variables, but also between enterprises due to differences in production systems, adopted practices and inherent risk exposure (Porter et al., 2014).

Climate change is a critical policy issue, however the requirement of the livestock sector develop GHG mitigation strategies is complicated by the fact that the sector must also develop strategies to combat the effect of climate change on production, without compromising the path to food security. The challenge to meet these three requisites has seen the advent of “climate-smart agriculture”; the development of agricultural practices that sustainably increase the agricultural productivity, enhance the resilience and adaptability of farmers to climate change, while also reducing GHG

emissions (Campbell et al., 2014; Lipper et al., 2014).

The Australian livestock sector faces similar challenges to those currently experienced on a global scale. Emissions from the Australian agricultural sector follow global trends, currently contributing 14% of total national anthropogenic emissions. Enteric methane comprises over 68% of this value (DISER, 2020b), reflecting the dominance of the production of beef cattle and sheep on broadacre grazing systems in Australia (A.D. Moore, Bell, & Revell, 2009). Ruminant livestock have been an important contributor to the Australia's economic growth (Stokes et al., 2010). With 65% of annual production exported (MLA, 2019c), Australia is the largest sheep meat exporter in the world and the third largest beef exporter (MLA, 2019a, 2019b). Proximity and access to expanding overseas markets is likely to continue to drive strong demand in this dominant sector of Australia's livestock industry (DAFWA, 2009). However, meeting this demand will be further complicated by the susceptibility of the industry to the adverse effects of projected climate change. The dependence of southern Australian broadacre livestock production on the supply of forage from dryland pastures means that it will be significantly exposed to predicted changes in climate over coming decades (Bell et al., 2012b; Ghahramani & Moore, 2016; Moore & Ghahramani, 2013), with some regions predicted to experience substantial losses in productivity and profitability (Howden et al., 2008). Without imminent action climate change will place at risk the capacity of southern Australian livestock production systems to continue the long-term productivity growth and resilience that has underpinned its historical success.

1.3 Research problem

Internationally and domestically, the livestock sector is a significant GHG emitter and shares responsibility for the mitigation of these emissions. If Australia is to achieve its target of a 26-28% reduction in national emissions by 2030 in accordance with the Paris Agreement (DISER, 2015), the livestock sector will need reduce its current emissions output and this requires immediate action. Considerable research has been undertaken to identify strategies to reduce emissions in livestock production systems (Beauchemin et al., 2020; Herrero et al., 2016). More recently, efforts have also focussed on strategies which enhance the resilience of the sector to the effects of projected climate change (Ghahramani et al., 2019; Henry et al., 2018b; Rivera-Ferre et al., 2016).

Despite this, widespread implementation of such strategies remains low (IPCC, 2014a). Barriers to uptake include the lack of policy drivers, impaired dissemination

of this knowledge to industry and low on-farm adoption due to cost constraints, lack of access to the technology itself and lack of demonstrated success by other adopters (Eckard & Clark, 2020a; Kragt et al., 2017). Clearly the decision as to whether mitigation is adopted lies primarily with the farmer, whose main motivators are productivity and profitability. Targeting mitigation strategies which also satisfy these motivators, either through emissions reduction resulting from productivity gains or from financial incentives attached to carbon offset schemes, is critical to short- to medium-term emission reductions in the sector.

Broadacre livestock producers in southern Australia have a long history of adaptability and innovation, spurred by challenging climatic and environmental conditions. Broadacre systems in southern Australia are reliant predominantly on temperate annual pastures, which in turn is dependent on the highly variable regional climate characteristic. Another challenge to the productivity of these systems are soils of low fertility, degraded from ongoing cultivation and the clearing of native perennial vegetation to make way for shallow-rooted crops and pastures (Dear & Ewing, 2008). Despite these challenges, producers have experienced productivity increases by implementing regionally appropriate and sustainable practices. These include the introduction and breeding of more productive livestock and pasture species, the use of fertilisers and legumes to enhance soil fertility, reduced tillage and tree planting, the application of lime and gypsum for soil remediation, the adoption of rotational grazing management, controlled stocking rates, and the introduction of perennial species (Chapman et al., 2009; Wolfe, 2009). Some of these strategies, such as the introduction of legumes, more productive livestock practices and tree plantings, have also been identified as potential mitigation strategies.

The mitigation potential of other strategies has not been explored further despite offering opportunities for dual productivity-mitigation outcomes. One example is the introduction of perennial pasture species. Perennials have the ability to increase the profitability and productivity of livestock systems (Descheemaeker et al., 2014), as well as alleviating environmental issues such as secondary soil salinity (Farquharson et al., 2013). Various perennial species have the ability to reduce enteric methane (Durmic et al., 2010; Revell et al., 2011), nitrous oxide (Dalal et al., 2003; DPI, 2007) and carbon dioxide emissions (Lawes & Robertson, 2012; Sanderman et al., 2013). Despite considerable focus on individual GHG emissions, there is an absence of research into the whole-farm GHG emissions of producing livestock on perennial pasture systems and of comparisons between perennial and annual pasture systems in southern Australia. Given the suite of benefits that perennials can offer and their

current use by some producers, there is clearly scope for further examination.

The whole-farm system analysis of GHG emissions, whereby all emissions arising from the various components and processes of an enterprise are considered, is a valuable approach to examine the carbon footprint of farming systems. For each enterprise, it enables the identification of emission hotspots and subsequent application of targeted mitigation strategies. Most international and Australian recommended mitigation strategies for application in livestock production systems have been developed at a component level. A whole-farm analysis of a strategy enables the impact across all components of a farming system to be considered (Rawnsley et al., 2016).

Whilst multiple whole-farm beef and sheep carbon footprint studies have been conducted in southern Australia, most are concentrated in the south-east regions and limited to the calculation of the carbon footprint and identification of hotspots, with a qualitative nod to potential mitigation strategies. The few studies which have examined the effect of proposed mitigation strategies, are limited to modelled farms in concentrated regions. No whole-farm carbon footprint study has investigated the impact of potential mitigation strategies in south-western Australia.

Given the heterogeneity of livestock production systems across southern Australia, it follows that the carbon footprints of these systems will also differ. It also means that there is a large array of possible mitigation strategies with varying degrees of suitability and impact, dependent on the characteristics of the enterprise in question (Howden et al., 2007; Rivera-Ferre et al., 2016). Whilst regional and state-level recommendations can be useful for policy makers and benchmarking analyses, it is widely recognised that the identification and application of strategies must occur at the farm-scale (Del Prado et al., 2013; Rawnsley et al., 2016). Such an approach could improve the level of adoption by producers and make progress toward national reduction of emissions through the application of a more targeted approach.

Consequently, there is an opportunity to investigate the carbon footprint of livestock production systems in south-western Australia to inform the development of mitigation strategies and identify areas for prioritisation. Furthermore, there is an opportunity to examine the effects of strategies with both productivity and mitigation potential in these respective livestock systems and target greater adoption through the regions considered.

1.4 Research goal and objectives

1.4.1 Research goal

The goal of this study was to investigate the carbon footprint of broadacre livestock production in south-western Australia under different pasture systems and identify regionally appropriate mitigation strategies for adoption by producers.

1.4.2 Specific objectives

To achieve this goal, the following objectives were developed:

Objective one - Develop a comprehensive tool that allows the calculation of the carbon footprint of sheep and beef cattle enterprises and examination of ensuing mitigation strategies.

Objective two - Quantify the carbon footprint of livestock production systems in south-western Australia, with focus on perennial versus annual pasture systems.

Objective three - Examine the mitigation potential of identified strategies on the carbon footprint of livestock production systems and provide regionally appropriate recommendations for application.

1.5 Significance of the study

Overall, this research will improve the understanding of the carbon footprint of, and effect of potential mitigation strategies on, livestock production systems. This will benefit the long-term sustainability of these agricultural production systems, being of relevance at both the international and Australian scale. The study has a number of unique characteristics, as follows:

- This research is novel in that it is the first to conduct whole-farm carbon footprint analyses across a representative range of broadacre livestock production systems in south-western Australia. There is well-evidenced difficulty in comparing the results of carbon footprint studies because of differences in methodological assumptions and data quality, for example. This study compares four livestock systems using consistent methodology to ensure research integrity.
- This research is the first to calculate the whole-farm carbon footprint of beef production systems in south-western Australia. Whilst the carbon footprint of wool production in the region has been examined, this research is only the second to consider meat production from sheep enterprises in south-western Australia.

- This is the first whole-farm carbon footprint study to examine the influence of annual versus perennial pasture systems, using farm-specific data which captures intra-annual variations and enables the performance of different pasture systems to be compared on inter- and intra-farm scales.
- This research is also the first to examine the impact of potential mitigation strategies on the carbon footprint of livestock production systems in south-western Australia. The findings of this research contribute to the identification of regionally suitable strategies with both productivity and mitigation potential. In doing so, the study intends to highlight the role of strategies which have already overcome barriers to adoption and can be readily adopted by willing participants.

It is anticipated that the findings of this research will enhance current knowledge regarding the carbon footprint of broadacre livestock production in southern Australia. It is intended that the identification of strategies with mitigation potential and demonstrated implementation by producers who were driven by other motivators, such as productivity and profitability, will assist in overcoming the current obstacles to climate resilience and mitigation.

1.6 Research design and methodology

This study adopted an integrated methodological approach involving life cycle assessment (LCA) methodologies and biophysical modelling. Carbon footprint analyses were conducted on sheep and beef production enterprises located in south-western Australia. Following this, the influence of strategies, recognised for both their mitigation and productivity potential, on the carbon footprint of livestock[1] production systems was examined.

Livestock enterprises are complex biological systems with many interacting components, including climate, soil, plants and livestock. Whole-farm system models can capture these interactions and examine the influence of climatic or management practices, both at the component- and farm-scale. Whole-farm system models have been developed to calculate a range of indicators, from productivity to profitability to environmental impacts such as GHG emissions. LCA is a holistic environmental management tool which can be applied at the farm-level to quantify the environmental impacts of a product or process. Initially developed for the industrial sector with fixed input to output systems, more recently, LCA has been increasingly applied in the

[1] For the remainder of the thesis, the term “livestock” refers to sheep and beef cattle only.

assessment of the environmental impacts of agricultural products or processes. Traditionally, LCAs consider the entire life cycle of a product, however due to the complexity of the livestock system including the multiple supply chain pathways and end products, most agricultural LCAs consider only one or two of the stages, for example, on-farm, processing or retail. Full LCAs consider multiple environmental impact indicators. A carbon footprint, by comparison, follows the principles of LCA but assesses one environmental impact indicator, GHG emissions. The principles of LCA (ISO, 2006a, 2006b) and carbon footprint analysis (ISO, 2018) guided this research through the "cradle to farm gate" carbon footprint analyses conducted. LCA consists of four phases; goal and scope definition, life cycle inventory (LCI) analysis, life cycle impact assessment and interpretation of results.

The foundational stage of the methodological approach of this study was the development of two carbon footprint Frameworks, one beef cattle- and one sheep-centric, which could be applied to broadacre livestock enterprises in southern Australia. These Frameworks were developed in accordance with the research goal and scope, comprising of an inventory component which entailed the compilation of detailed farm-specific data and also an impact assessment component, which calculated the carbon footprint of the enterprise in question. In line with good practice recommendations, all GHG emissions were calculated following the Australian National GHG Inventory (NGGI) methodologies (DISER, 2020b), which in turn, follows the Intergovernmental Panel on Climate Change (IPCC) methodology (IPCC, 2006). Whilst these Frameworks enabled whole-farm system analysis, they also permitted in depth intra-farm analyses, including the role of different pasture systems, intra-annual variations, stock classes and farming practices.

The Frameworks were applied to calculate the emission intensity (EI) of livestock produced for sale (CO_2-e/kg LW produced) across four livestock production enterprises. These farms were located across major farming regions in south-western Australia and were selected for a range of factors including; adoption of innovative practices, enterprise type, climatic conditions and pasture systems. It was not the intention for these farms to, for example, represent all sheep and beef enterprises in the region, but rather to build on the existing repertoire of carbon footprint information in south-western Australia, to examine the influence of pasture systems and to inform the identification of farming strategies with mitigation potential.

For each enterprise, in-depth information regarding farm inputs (i.e. chemicals, machinery, supplementary livestock feed) and farm practices (i.e. animal, crop and pasture management) was sourced primarily from the farmer. The biophysical model,

GrassGro (Donnelly et al., 1997; Freer et al., 1997), was used to generate site-specific monthly pasture and animal outputs for input into the framework, whilst SimaPro software (PRé Consultants, 2014) was used to source emissions information for the impact assessment stage.

The final stage of this research involved the identification of farm-level mitigation strategies with real potential for uptake by livestock production systems in south-western Australia, and potentially, southern Australia. Strategies, implemented by farmers for improved productivity, but with mitigation potential were selected. The influence of these strategies was examined through integration in the Frameworks, and promising options identified. The findings of the research could also have relevance at the international level where similar production systems exist.

1.7 Thesis outline

This research thesis consists of eight chapters as presented in Figure 1.1. The current Chapter has summarised the rationale, goal and objectives for this research, and outlined the methodological approach adopted to address these objectives.

Chapter Two presents a comprehensive review of existing Australian and international studies that have investigated whole-farm GHG emissions from sheep and beef production enterprises. This includes studies that have provided benchmarking carbon footprints of such systems and those which have examined the impact of proposed mitigation strategies on ruminant production. A critical analysis of the methodological approaches is also presented.

Chapter Three discusses the development of the methodological approach adopted for this study. Namely, the development of beef cattle and sheep carbon footprint Frameworks which address the methodological concerns identified in Chapter Two and enable achievement of the objectives of this. The integration of the Frameworks and biophysical modelling is outlined, which informs the targeted analysis of the mitigation potential of farm practices.

Chapter Four presents detailed inventories for the four case study farms examined in this study, including the characteristics of the livestock, pasture, crop and soil components of each system, along with the management practices applied by the respective farmers. An initial analysis of animal and soil emission sources is conducted, with preliminary comparisons drawn across, and within, each farm, between different pasture systems. Through these initial analyses strategies adopted for productivity purposes by the respective farmers which influenced emissions were

identified.

Chapter Five presents and interprets the results of the whole-farm carbon footprint analyses of each examined enterprise. The methodological approach ensured detailed analyses were conducted between pasture systems and stock classes, across each month of the production year. This enables not only the performance of the enterprises to be compared, but also the performance of different components within an enterprise.

Chapter Six explores, through scenario analyses, the impact of perennial pasture systems on the carbon footprint of livestock production systems, using the four cases study farms as examples. Furthermore, the potential influence of carbon sequestration as an emissions sinks is examined.

Chapter Seven examines in detail, the impact of selected farm practices strategies on the carbon footprints of the livestock production systems. These practices comprised a combination of livestock reproductive and grazing management strategies adopted by one or more of the case study farms (as identified in Chapters Four and Five) and strategies identified in existing research as promising options for the mitigation of farm emissions.

Lastly, Chapter Eight concludes this thesis, presenting the main results of the study and the implications of these findings. The Chapter also discusses areas of research requiring further investigation identified through the process of this research.

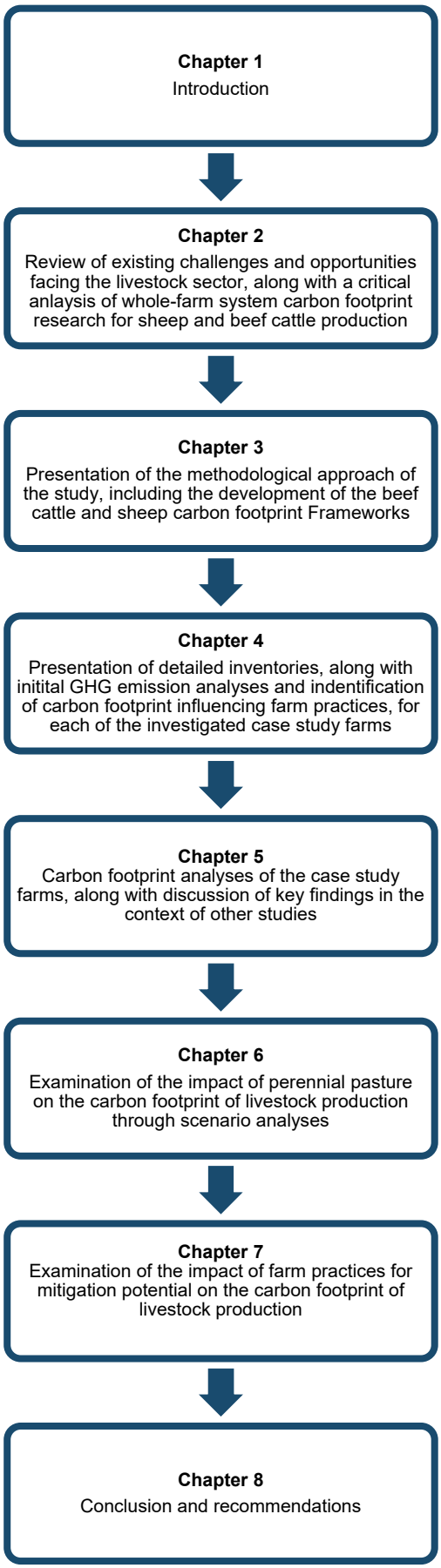


Figure 1.1 - Thesis outline

2 LITERATURE REVIEW

2.1 Introduction

This Chapter will explore the role of ruminant livestock with regards to climate change, alongside the key challenges and opportunities arising for both the international and southern Australian livestock sectors. In recognising the commitment of the livestock sector to reduce emissions despite lack of widespread mitigation to-date, the potential of whole-farm system analyses to enable farm-scale carbon footprint analyses and targeted assessment of potential mitigation strategies is investigated. A comprehensive review of existing international and Australian livestock whole-farm analyses is undertaken, identifying current knowledge gaps and opportunities, along with methodological considerations to enable further improvements in whole-farm analyses. Finally, promising mitigation strategies for on-farm adoption by producers into the short-term are highlighted for further investigation.

2.2 Ruminant livestock: both a source of GHG emissions and vulnerable to climate change

The production of livestock is associated with a number of negative environmental impacts; along with competing for increasingly scarce resources, the sector contributes to land degradation, water pollution, biodiversity and climate change (Steinfeld et al., 2006). In particular, the contribution of livestock to climate change draws substantial attention as the sector contributes 14.5% of global anthropogenic emissions (Gerber et al., 2013). However, livestock production, along with agricultural production, is also more vulnerable to climate change than any other industry. The livestock industry, both as a net emitter of GHG emissions whilst also susceptible to the impacts of climate change, means it that has become the focus of a wide range of stakeholders and countless proposed interventions. Based on the presented evidence, to date all have fallen short of what is required to achieve sustained and viable change.

2.2.1 GHG emissions from livestock production systems

The principal emissions produced by livestock production systems include methane (CH_4), nitrous oxide (N_2O) and carbon dioxide (CO_2). The contribution of these non-carbon dioxide emissions to total anthropogenic emissions is significant, with methane emissions from the livestock sector contributing 44% of anthropogenic methane emissions, and nitrous oxide comprising 53% of anthropogenic nitrous oxide

emissions (Gerber et al., 2013). Methane and nitrous oxide are much more effective at trapping heat in the atmosphere to that of carbon dioxide and over a 100-year time period, the global warming potentials (GWPs) are 28 and 265 times greater than carbon dioxide, respectively (IPCC, 2013). Applying these GWPs converts these gases to carbon dioxide equivalent (CO2-e), enabling the three GHGs to be compared for their respective contributions to global warming. This means that whilst the livestock sector produces lower quantities of actual methane and nitrous oxide than carbon dioxide, the higher GWPs of the two means that, in terms of CO_2-e, their impact is far greater. In fact, in terms of CO2-e, methane and nitrous oxide emissions make up over 73% of global livestock sector emissions (Gerber et al., 2013).

Ruminant supply chains are the primary source of global emissions produced by the livestock sector, contributing 80% of total emissions (Opio et al., 2013). There are multiple processes in these chains which contribute to these emissions. Methane is produced predominantly through enteric fermentation by ruminant livestock, but also through the anaerobic decomposition of livestock manure. Nitrous oxide is released both directly and indirectly from soil. Direct pathways include nitrous oxide produced through; the application of organic and inorganic nitrogen (N) fertilisers, manure management and the decomposition of organic material including animal waste and plant material. Indirectly, nitrous oxide is produced through leaching and runoff from soil and atmospheric deposition of N. Carbon dioxide is generated through the production, transportation and on-farm consumption of farm inputs such as chemicals, livestock feed and fuel, while also being released from soil via urea hydrolysis and following applications of lime (LEAP, 2015b, 2016).

Emissions also arise from land use (LU, soil carbon losses) and from direct land use change (dLUC, i.e. forest to cropland, forest to pasture). Land use can also act as a sink, through soil C sequestration. It is a requirement of the Australian National GHG Inventory (DISER, 2020b) and core standards for carbon footprint analysis (ISO, 2018) to report LU and dLUC separate to other emissions. This is because there still remains much uncertainty regarding data availability, data resolution and the most appropriate methodological approach to calculate these emissions (LEAP, 2015a). It is for these reasons, that these emissions have been excluded from most analyses to date.

Of the emissions sources described, enteric methane production is the predominant source in ruminant production systems. So much so, that it comprises almost 6% of global anthropogenic emissions, or 40% of the global livestock sector's emissions (Gerber et al., 2013). Enteric methane is a by-product of ruminal fermentation, the

process by which ruminants can digest high-fibre plant material unsuitable for monogastric livestock. However, with an energy content of 55.22 MJ/kg (Brouwer, 1965), it also represents an energy loss, in the range of 8-14% of the digestible energy intake of ruminants (Cottle et al., 2011; Johnson & Johnson, 1995). Hence, along with a being significant source of anthropogenic emissions, enteric methane represents a loss of potential productivity and profitability to producers.

Similar to enteric methane, many of the other emissions produced in ruminant production systems represent both the release of emissions into the atmosphere and a loss of energy or nutrients. Nitrous oxide from animal waste, for example, is the second largest source of global livestock emissions (15-16%) and represents undigested protein consumed by livestock. The third largest source, synthetic fertiliser application (12%) in part represents inefficient input use (Gerber et al., 2013; Smith et al., 2014). In fact, emissions from fertiliser are set to overtake those from animal waste if current growth trajectories continue, reflecting the increased reliance on higher inputs to achieve productivity gains (Smith et al., 2014). So while the release of emissions to the atmosphere exacerbates the effects of climate change, they also represent inefficiencies in the system, often with economic implications for supply chains. Reducing these losses will have widespread benefits.

2.2.2 The impact of climate change on livestock production systems

Global climate is changing, driven by human-induced increases in atmospheric GHG concentrations. The past three decades have been successively warmer than any other decade since 1850 (IPCC, 2013). Accompanying the sustained increases in global surface temperature has been observed warming and acidification of oceans and water bodies, rising sea levels and changing precipitation patterns, amongst other effects. None of these variations can be explained by natural variability (Rosenzweig et al., 2008). These changes have already impacted agricultural output and it is estimated that over the past 30 years global agricultural production has been 1 to 5% lower per decade than without these observed changes to climate (Porter et al., 2014).

Projected climate change predicts a continuation and amplification of observed changes to climatic variables, along with increases in extreme weather events such as droughts and floods. Global surface temperatures are predicted to increase by between 0.3 and 4.8 °C by 2081-2100, relative to 1986-2005 baselines (IPCC, 2013). The lowest increase represents the most ambitious scenario, whereby emissions peak around the time of this thesis (2020) and decline rapidly after. It is widely accepted that this scenario is unachievable, and global efforts formalised in the Paris

Agreement in December 2015 are to limit warming to between 1.5 to 2°C (UNFCCC, 2015). Growing evidence suggests that even if the lower target of a 1.5°C increase was successfully met, global agricultural productivity would still decline, with regions such as tropical or dryland areas, more vulnerable (Hoegh-Guldberg et al., 2018).

The livestock sector's reliance on natural resources makes it vulnerable to effects of climate change. Overwhelmingly, observed and projected changes to climate have, and will continue to, negatively influence livestock production (Porter et al., 2014), with changes to the quantity, quality and composition of feed, reduced water availability, heat stress on animals, increased persistence of pests and diseases, and decreased persistence of livestock species in certain areas (Henry et al., 2018b; Rojas-Downing et al., 2017). The social and economic ramifications of these will be significant for producers, particularly those in developing regions whereby farms are often not diversified and livestock are the sole source of income and thus livelihoods (Porter et al., 2014; Thornton et al., 2009).

The observed and projected impacts of climate on livestock are temporally and spatially variable, influenced by variations to climatic factors across geographical regions, socioeconomic conditions and livestock production systems. The way different production systems are exposed to changes to climate are diverse. For example, intensive industrial livestock systems, often located in more urbanised regions, will be affected indirectly, through rising input costs (particularly feed and water) and competition for other land uses. By contrast, low-input extensive grazing systems, dependent on native grasslands or farm-produced pasture supply, will be affected directly through changes to forage production and the incidence of pests or disease (Rivera-Ferre et al., 2016). In fact, extensive grazing systems are most vulnerable, due to this dependence of forage production on climatic factors.

The intra- and inter-annual variability of climatic factors influencing pasture have increased and are projected to continue to do so, particularly in regions where livestock have economic or food security importance (i.e. Australasia and sub-Saharan Africa, respectively (Sloat et al., 2018; Stagge et al., 2017)). Overall, the projected changes to global pasture production and quality under climate change will be negative. However, the impacts will be regionally specific; for example, in temperate regions such as Europe, increased temperatures are likely to result in extended growing seasons, but along with a higher incidence of frost during winter, pasture quality will decline (Ghahramani et al., 2019). By contrast in tropical and subtropical regions, combinations of reduced rainfall and higher temperatures are likely to reduce pasture production and quality (Henry et al., 2018b). Changes to

climate will also influence the persistence of pasture species. Increased atmospheric carbon dioxide levels will favour temperate (C3) species while warmer temperatures will favour tropical (C4) species. The net effect on the outcome of pasture composition is uncertain and will be location-specific (Moore, 2012; Porter et al., 2014). For example, the modelled positive effects of carbon dioxide fertilisation in Australian pasture systems are insufficient to offset the negative impact of reduced rainfall and increased warming (Ghahramani & Moore, 2016). The diversity in the projected impacts of climate change on livestock systems globally, demonstrates that the importance of targeted mitigation tailored to the geographical, political and socio-economic contexts under consideration.

2.3 Livestock production in southern Australia

2.3.1 The southern Australian livestock industry

In Australia, the livestock industry is dominated by the production of beef cattle and sheep on broadacre grazing systems (Moore et al., 2009a). Of the 394 million hectares dedicated to agricultural production in Australia, approximately 87% is used for grazing purposes (ABS, 2018). Almost 90% of this grazing land is used for livestock grazing rangelands predominantly in northern Australia and is dominated by native vegetation. The remaining 10% of allocated grazing land is predominantly improved rainfed pastures, grazed by livestock at higher stocking rates (Figure 2.1).

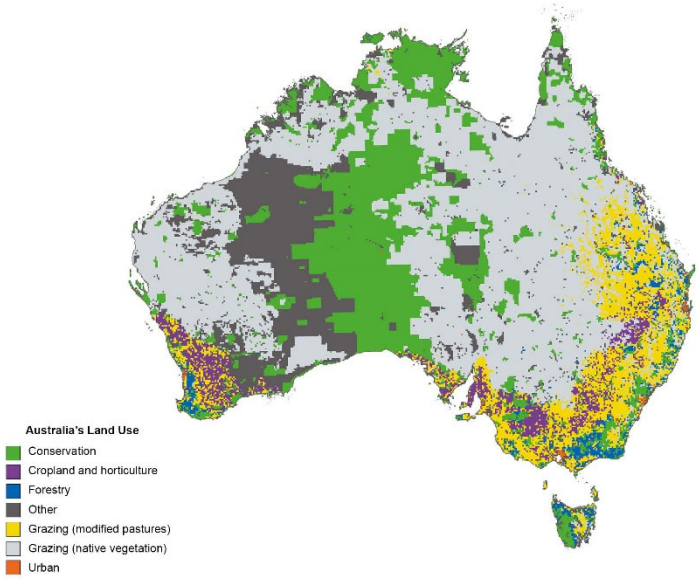


Figure 2.1 - The geographical distribution of various land uses across Australia (Sourced from ClimateWorks, 2020)

Figure 2.2 shows that these improved pastures are concentrated in southern Australia, where Mediterranean type climate and temperate agroecological zones dominate.

Despite the range of agroecological zones across Australia, following the patterns of typical grazing systems, livestock production can be broadly differentiated between northern and southern Australia. Northern Australian livestock systems, located in Queensland, northern Western Australia and the Northern Territory, exhibit summer-dominant rainfall with distinct wet and dry seasons. Unsuitable for sheep production, this region is characterised by the hardier *Bos indicus* cattle breeds, typically grazed at low stocking rates over vast properties owned by large corporate entities. Cattle are destined for either live export, predominantly to Asian markets, or sent to southern Australia for finishing prior to slaughter and subsequent export for large scale consumption in countries such as the USA (PwC, 2011).

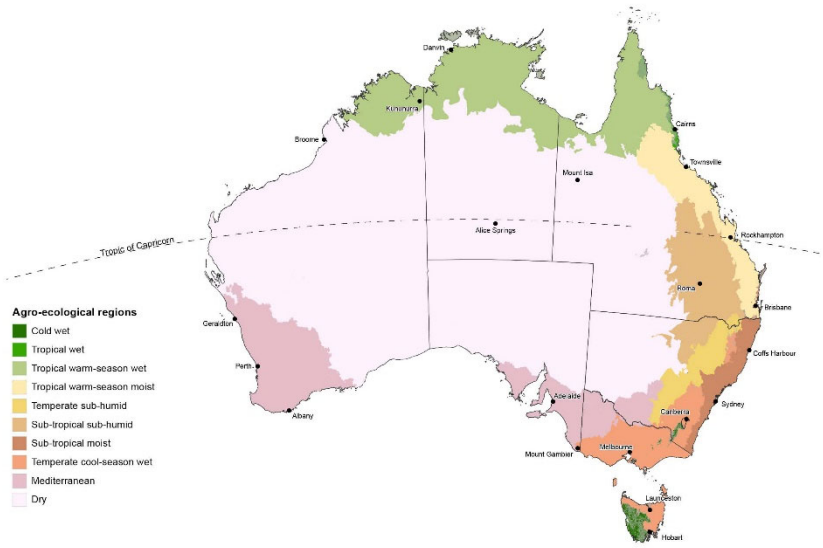


Figure 2.2 - The agroecological regions of Australia
(Adapted from ABARES (2018))

Though southern Australia comprises of a number of different climatic zones across NSW, Victoria, Tasmania, Southern Australia and southern Western Australia, the region overall is characterised by winter-dominant rainfall patterns. Suitable for both sheep and cattle production, livestock are typically produced in more intensive systems, mixed (i.e. crop and livestock) or specialised, with more productive temperate, or more recently subtropical, pastures enabling higher stocking rates. Sheep are predominantly Merino, produced for both wool and meat, crossbred Merino and specialist meat breeds. The majority of lamb (66%) and mutton (96%) produced

is exported, predominantly to Chinese, Middle Eastern and USA markets (MLA, 2019b). Cattle in southern Australia are characterised by higher weaning and growth rates along with higher stocking rates, enabled by climatic conditions and ability to run *Bos taurus* breeds. The resultant premium beef is sold to high value export markets such as Japan and Korea. Feedlot finishing has gained importance in recent decades, increasing finishing efficiencies of cattle destined for both those premium export markets and also domestic consumption (Wiedemann et al., 2015d).

Together, these systems are important contributors to the economic growth of Australia (Stokes et al., 2010), with the red meat and livestock sector contributing $18.5 billion, to Gross Domestic Product (GDP), or 44% of the agricultural industry's total contribution to GDP (MLA, 2019e). Though the southern beef industry accounts for only 25% of grazing land dedicated to beef (PwC, 2011), it supports 40% of Australia's cattle herd and is responsible for 46% of the gross value of national beef production The southern sheep meat industry contains more than 95% of Australia's sheep flock and is responsible for more than 99% of the gross value of sheep meat production (ABS, 2020; MLA, 2019e). The red meat sector is heavily dependent on the export market, with over 65% of annual beef and sheep meat production exported, either through carcass or live export. Despite burgeoning international demand for Australian livestock products, the national cattle herd and sheep flock has been falling (MLA, 2019e), a result of the widespread drought conditions experienced by large parts of Eastern Australia since 2017, major flood events in Queensland in 2019 and bushfires through southern Australia in 2019/2020. This perhaps, shows insight into the increased weather events Australia is expected to encounter under projected climate change. The industry will need to continue to innovate and adapt to develop the resilience required to weather these changes into the future.

2.3.2 Challenges facing broadacre livestock producers in southern Australia

Livestock producers in southern Australia's agricultural regions have a long history of adaptability and innovation, spurred by challenging conditions. These regions are characterised by highly variable climate, particularly rainfall, along with soils typically low in fertility, soil organic carbon and plant available water capacity (Lawes & Robertson, 2012; Turner & Asseng, 2005). Livestock production systems throughout southern Australia are reliant on pasture systems and so producers typically run low-input systems with a high degree of flexibility to remain resilient in the face of inter-annual variability. Winter-dominant rainfall patterns mean that producers must

contend with an annual feed gap. The length and timing of this gap is determined primarily by pasture supply which in turn, is primarily determined by climatic factors such as rainfall and temperature. Managing this gap is of primary concern to producers as it determines the carrying capacity of the enterprise and annual supplementary feed requirements, a large cost to the system (Moore et al., 2009a). In this sense, if not well-managed, it also represents a limitation to the productivity and profitability of an enterprise.

Whilst climate variability is one of the greatest sources of risk for Australian agriculture (Kingwell et al., 2013), the spread of agriculture has introduced a suite of additional environmental issues that producers now must contend with. The extensive clearing of deep-rooted perennial native vegetation and subsequent replacement with shallow-rooted annual crops and pastures has contributed to widespread secondary salinity and waterlogging exhibited on agricultural land (Dear et al., 2003; Turner & Asseng, 2005). Decades of over overgrazing by livestock and excessive cultivation of already fragile soils has also resulted in soil degradation in the form of soil erosion, soil acidity, increased soil water repellence and reduced nutrient cycling. This severe land degradation, paired with significant inter-annual climate variability has resulted in a focus on sustainable production by research organisations, industry bodies and farmers alike (Chapman et al., 2009; Wolfe, 2009). Consequently, a suite of sustainable and cost-effective farming systems and practices specific to agricultural livestock regions have been implemented. These include the introduction and breeding of more productive livestock and pasture species, the use of fertilisers and legumes to enhance soil fertility, reduced tillage and tree planting, the application of lime to soils, the adoption of rotational grazing and phase cropping systems, controlled stocking rates and the introduction of perennial species (Wolfe, 2009).

In addition to climatic and environmental challenges, livestock producers are faced with a number of socioeconomic challenges, ranging from declining terms of trade, increasing costs of inputs, consumer demand for “ethical, clean and green” products, labour shortages, continued changes to policy and declining research support. The success of a farming enterprise, or the livestock sector, is determined by its adaptive capacity in the face of such challenges.

2.3.2.1 Climate change

Broadacre livestock producers in southern Australia face two main climate risks; inherent climate variability characteristic of the region and the ongoing exposure to climate change. Where climate variability refers to short-term seasonal or inter-annual

fluctuations of climatic variables, climate change refers to long-term trends in climatic factor averages and variability occurring over decades or longer (Loch et al., 2012). Climate change increases the risk exposure of production systems and adds complexity to management decisions that already must consider inter-annual climate variability, along with a suite of other factors.

Climate has already demonstrated sustained changes in Australia. Australia's climate has warmed by 0.9 °C since 1910, accompanied by an increased frequency of extreme heat events and increased bushfire risk (CSIRO & BOM, 2015). Rainfall distribution, while more variable, has also changed, with sustained drying in the south-west and southern-east regions of Australia, particularly over the cooler months which historically provide the rainfall farmers are so dependent on. This drying has been particularly significant in south-western Australia where persistent declines have been observed since the 1970s, compared to the south-east which first observed drying from the 1990s. Declines in cool season rainfall have been as much as 40% in parts of the south-west region (IOCI, 2012). All of these changes are far greater than what would be expected from natural climate variability and are consistent with the effects of increased atmospheric gas concentration (CSIRO & BOM, 2015; IPCC, 2014a).

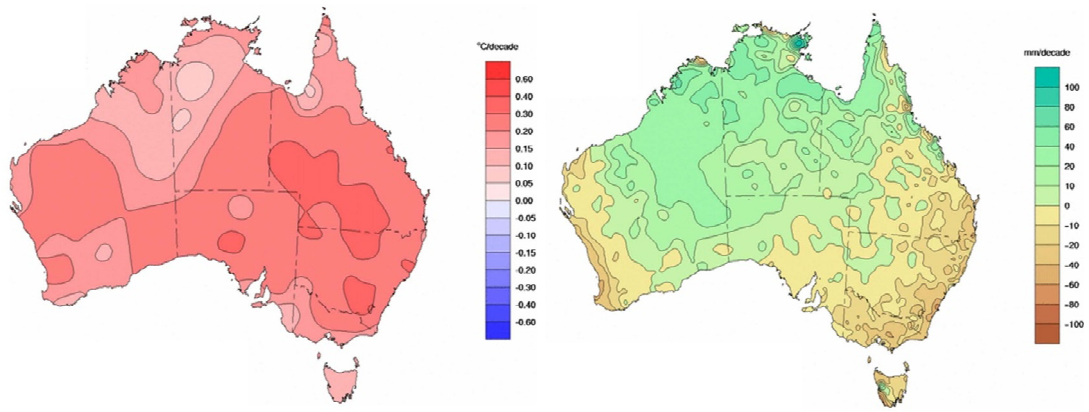


Figure 2.3 - Observed trends in (a) average annual temperature and (b) average annual rainfall, in Australia from 1960 to 2019
(Sourced from (BOM, 2020))

The warming exhibited in southern Australia is set to continue under projected climate change, with average annual temperatures projected to increase by 0.6-1.3 °C as early as 2030. By the end of the century, projected temperatures are expected to range from 1.4-5.1 °C, with the lower value representing a scenario whereby emissions peak around 2040 and the upper value if business was continued as usual (Figure 2.3a)(CSIRO & BOM, 2015). There remains a degree of uncertainty regarding

projected changes to rainfall amounts and patterns (Kingwell et al., 2013), due to competing rainfall drivers; however, across most regions in southern Australia there is high confidence that the observed declines in cool season rainfall will continue. South-western Australia is projected to experience more significant declines (Figure 2.3b), with average annual rainfall falling by up to 20% by the end of the century and cool season rainfall by 32%; far greater than that predicted in south-eastern Australia (IPCC, 2014a) Along with sustained changes in climatic factors, the incidence and severity of weather events, such as droughts and bushfires, is predicted to increase across southern Australia. Whilst the actual presentation of climate change and its impact on livestock production systems will vary regionally (Figure 2.4) and by enterprise, overwhelmingly the outcome is predicted to be negative (IPCC, 2014a).

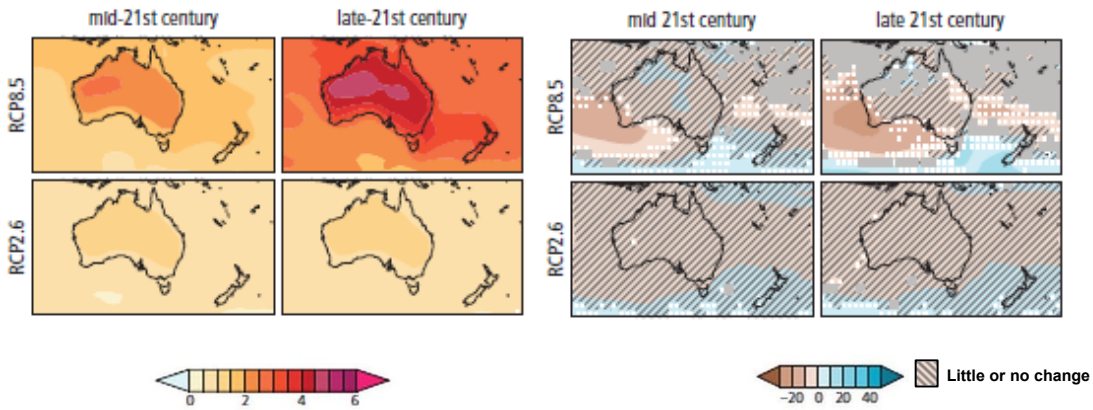


Figure 2.4 - Projected changes to average (a) annual temperature and (b) rainfall in Australia to 2046-2065 and 2081-2100 under two emission pathways (Sourced from IPCC (2014a)). RCP8.5 represents an emission pathway whereby no adaptation is adopted and emissions continue to increase in line with increasing demands. RCP2.6 by contrast, represents a pathway whereby emissions peak in 2020 and then decline, a result of extensive and global mitigation.

The dependence of broadacre livestock systems in southern Australian on pastures as the primary feed source means that they will be particularly vulnerable to the effects of projected climate change. Modelling has shown that without adaptation and only managing stocking rates for erosion risk, projected climate change across 2030 to 2070 could reduce pasture production in livestock systems in southern Australia by between 9 and 14%, respectively. However these declines would disproportionately affect operating profit, with respective declines from 27% in 2030 to 48% in 2070 (Moore & Ghahramani, 2013). Mixed crop and livestock enterprises, typically distributed through lower rainfall regions, are projected to become less viable, particularly in drier sites where increased crop yield variability will affect returns and

potential positive effects on pasture production from increased atmospheric carbon dioxide concentrations will be offset by reductions in rainfall and higher temperatures (Ghahramani & Moore, 2016). Higher rainfall areas are likely to be better able to withstand projected changes to climate, with pasture production projected to increase in some regions (Bell et al., 2012b). The increased frequency and duration of droughts will make it more difficult for producers to recover from what is already one of the greater risk exposures of broadacre farming, with cattle herds predicted to take longer to recover from drought in projected climate change (Godde et al., 2019). Though the actual magnitude of effects are predicted to vary across the livestock sector, the consensus is that, without systemic change to the sector the outcome will be negative in terms of both productivity and contribution to GDP (McRobert et al., 2019).

Though not the focus of this thesis, it is important to acknowledge that climate change in the livestock sector will have far reaching effects beyond direct impacts on the productivity and profitability of the livestock systems. Projected climate change will exacerbate other pre-existing stresses encountered by broadacre farmers, such as invasive weed species, diseases, water scarcity and resource fragmentation (Howden et al., 2008). Reduced agricultural production arising from climate change will impact the sector's export competitiveness (McRobert et al., 2019). Given the dependence of the sector on the export market this would have serious ramifications along supply chains and present severe risks to the viability of the sector. Furthermore, it will impact communities that are so heavily reliant on agriculture through the damaging of social capital. It is likely to further highlight and amplify the social, economic and health inequalities already experienced by regional communities (Hughes et al., 2016).

Without widespread adaptation by the agricultural industry, climate change could result in a GDP loss of 5% by 2050 (Gunasekera et al., 2007). With regional adaptation, Heyhoe et al. (2007) expect this economic impact to be halved. Broadacre livestock producers in southern Australia have already demonstrated that they can perform in a highly variable climate with significant related challenges such as water scarcity and soil degradation through the development of innovative climate risk management practices. Climate change amplifies these challenges and delivers new ones and hence to combat the projected effects, farmers need to continue to adopt existing and new risk management practices, alongside coordinated policy measures which encourage widespread adaptation of these practices (Howden et al., 2008). Ultimately, the overall effect of climate change on the productivity and economic viability of livestock production in southern Australia will depend on how much it is possible to adapt to reduce the impact of such change.

2.3.3 How is the livestock sector addressing climate change?

The livestock sector must address climate change with dual focus; implementing strategies to reduce emissions while adapting to improve its resilience to projected climate change impacts. The Australian agricultural industry contributes 13.5% to national net emissions, behind only the energy and transport sectors (excluding LU and dLUC per reporting requirements (DISER, 2020b)). Most of these emissions arise from ruminant livestock, with enteric fermentation alone comprising almost 70% of agricultural emissions. It is likely that the actual contribution of agriculture, and thus the livestock sector, to net emissions is greater than 13.5% as the Australian National GHG Inventory, from which these figures are obtained, excludes emissions from the manufacture and transportation of farm inputs (i.e. fertilisers, machinery, livestock feed) as well as emissions associated with on-farm fuel and electricity usage. Regardless, as a significant source of GHG emissions, the livestock sector has opportunity to share a responsibility to mitigate its contribution to climate change.

Under the 2015 Paris Agreement on Climate Change, Australia has committed to reducing GHG emissions by 26-28% below 2005 levels by 2030 (UNFCCC, 2015). Unlike other countries which have assigned specific targets to their agricultural sector, Australia's target is economy-wide (Eckard & Clark, 2020b). Despite this, all sectors are expected to contribute to this target. Excluding the carry over units from Kyoto that the federal government intends to apply in order to meet the current target, Australia is not on track to fulfill its commitment, with net emissions increasing annually since 2014 (DISER, 2020b). At present, the fulfilment in entirety of commitments made by all signatories to the agreement would only limit warming to between 2.0 and 3.0 °C, greater than the ultimate target of 1.5-2.0 °C established at the Agreement inception (UNFCCC, 2015). The initial targets established by parties are set to be revised upwards in 2020 in accordance with the Agreement's ultimate target. This will make Australia's path to achieving its commitment increasingly difficult without immediate and coordinated national action.

The development of climate change policies in Australia has, and continues to, encounter many obstacles, amplified by the fluctuating political landscape of the past decade. In 2012, a Carbon Pricing Scheme was introduced through the *Clean Energy Act 2011* by the then-Labour government. The scheme initially assigned a fixed carbon price directed at liable entities such as large polluters, with the intent that this would support the growth of clean energy technologies, before transitioning to an emissions trading scheme (ETS) in 2015 (CER, 2015). Agriculture was excluded from this scheme and producers were instead encouraged to participate in the Carbon

Farming Initiative (CFI), a voluntary initiative which by which participants could gain carbon offsets by conducting approved projects which sequester carbon or directly reduce GHG emissions. These earned credits could then be sold to entities with liabilities under the carbon pricing scheme (Climate Change Authority, 2014). Following the repeal of the Carbon Pricing Scheme in 2014 by the newly formed Liberal government, the CFI was amended to establish the Emissions Reduction Fund (ERF) which was the cornerstone of the Government's Direct Action Plan (DAP) to reduce GHG emissions. Unlike the CFI, the ERF is to open to all sectors of the economy and the federal government is the sole purchaser of offsets polluters to reduce emissions (Climate Change Authority, 2014).

The Emissions Reduction Fund (ERF), recently renamed again, to the Climate Solutions Fund in 2019 along with an addition $2 billion of funding allocated to future abatement over the next 15 years, has become the Government's default climate change policy (DISER, 2020a). In the context of the broadacre livestock sector, such eligible projects exist either through vegetation management (i.e. tree plantations and regeneration of native forests) or agriculture specific management, including soil carbon sequestration, feeding cattle nitrates to reduce enteric methane, or beef herd management through improved emissions intensity (i.e. emissions per unit of liveweight). Despite the opportunities presented for producers to participate in ERF approved projects, uptake has been limited, and to only a few project types. Of the total 811 projects under the ERF at 2020, over half involved regeneration of native forests on agricultural land and only 78 projects fell under the agriculture category. Of these, only five were being conducted through beef herd management (Macintosh et al., 2019). This demonstrates that mainstream agriculture is still not engaged with the ERF and carbon offset farming, even after almost a decade of operation.

In parallel to federal policies, and in many ways driven by the lack of realised impact from those government policies, industry-driven initiatives are being established to drive the reduction of emissions from the Australia livestock sector. An example of such a program is the "Carbon Neutral by 2030" strategy, launched in 2017, which established a target for the Australian red meat sector to become carbon neutral by 2030. Using annual emission data from the Australian National GHG Inventory (NGGI), the initiative states that the red meat sector is well on its way to neutrality, already having achieved a 57.6% reduction in total emissions from the baseline year in 2005, to the 2016 reporting period (ABSF, 2019). The data from which this figure is calculated considers emissions from LU and dLUC, along with those associated with livestock production. Per the NGGI, it does not consider pre-farm emissions (i.e.

chemical and supplementary feed production and transportation). In fact, an analysis of the results of the study upon which this figure is based (Mayberry et al., 2019), revealed that more than 95% of this reduction in sector emissions is a result of falling emissions from the slowing of deforestation activities on livestock farms. The greatest reduction in land clearing activities, and thus emissions, occurred in the first five years of the reporting period, after which it began to plateau (DISER, 2020b). Total emissions associated with the production of livestock (i.e. enteric methane, manure methane, soil emissions) however, remained relatively constant during the ten-year period. Opportunities for further emission reductions through reduced land clearing are limited; the analysis of the results presented by Mayberry et al. (2019), indicated that even if deforestation on livestock farms was to stop completely by 2030 it would achieve less than half the remaining required emission reduction. To meet this ambitious goal of carbon neutrality within a decade, the red meat sector will instead have to achieve these emission reductions through other pathways, namely through emissions associated with the production of livestock which, at current trajectory, is far from achieving the scale of reductions required. Regardless of the task at hand, the initiative demonstrates the commitment of the industry and stakeholders to reducing the footprint of the industry and to enforce short-term action through significant investment in R & D and widespread policy development.

Despite the lack of continuity and hampered progress in developing a nationally coordinated climate policy, Australian agriculture is moving forward with the development and implementation of practices to both adapt to and mitigate a changing climate. The livestock sector has made some good progress, either through the decreased rates of deforestation as outlined above, or through productivity improvements exhibited through the production of improved livestock production with lower resources and GHG emission output (Wiedemann et al., 2015d). Considerable research has been undertaken to identify strategies to mitigate emissions from livestock enterprises (Beauchemin et al., 2020; Eckard & Clark, 2020b; Guyader et al., 2016; Hristov et al., 2013a; Hristov et al., 2013b) and to adapt these enterprises to climate change (Ghahramani et al., 2019; Henry et al., 2018b; Rivera-Ferre et al., 2016). However, overall uptake of these strategies by farmers to-date has been low, hampered by factors such as cost constraints, uncertainty resulting from impaired knowledge transfer to producers from researchers, lack of access to the technology itself and lack of demonstrated success by other adopters (Eckard & Clark, 2020b; Kragt et al., 2017; Pannell et al., 2006). Despite this, farmers have and continue to, implement other practices on their enterprises to address the incremental climatic and

additional risks they face, along with changing consumer demands. The continued adaptive capacity and resilience demonstrated by farmers in the face of challenging conditions is vital to adapt to Australia's projected climate. However, they need the support of Government, industry, and the dissemination of effective solutions if they are to address the challenges posed (Hochman et al., 2013; Macintosh et al., 2019). Clearly, Australia cannot continue its current trajectory if it is to meet the mitigation targets established by both industry and Government whilst addressing projected climate change and retaining productivity.

2.4 Whole-farm system analysis to quantify GHG emissions

The previous Section highlighted that existing climate policies and research targeting the mitigation of GHG emissions from livestock enterprises have largely focussed on component-level mitigation. There has been poor uptake of such strategies by Australian livestock producers, hampered by barriers such as cost and uncertainty. Whole-farm system analysis presents an opportunity to address these barriers, by improving the information available regarding the current operations of farming systems as well as examining the effect of strategies at the scale upon which farmers operate and make key management decisions.

2.4.1 Principles and benefits of whole-farm system analyses

Livestock production enterprises are complex biological systems comprised of multiple interacting components such as soil, crops, pasture, livestock, farm inputs and climate. A whole-farm system approach enables each component to be considered and the many interactions between components to be captured. This may be in the context of economics, productivity or environmental impact such as GHG emissions. Because of this, whole-farm system modelling is recognised globally as a valuable tool in decision-making processes and the prioritisation of research investment where field experimentation would be too expensive or impractical (Eckard et al., 2014; O'Brien et al., 2016).

In the context of emissions, whole-farm analysis can be applied to quantify overall emissions from a farming system along with the components or processes within the system with the greatest environmental impact. It can also be utilised to examine the effect of a mitigation strategy at farm-scale, revealing interactions or impacts that may not be obvious when considered only in a component-level analysis (Rawnsley et al., 2016; Schils et al., 2007). Most mitigation strategies have been developed to target emissions arising from one component of the farming system. At a whole-farm scale that strategy may have lower or even higher than expected mitigation, and unintended

effects on other measures such as productivity. For example, Harrison et al. (2014a) examined the effect of improving genetic feed-use efficiency of cross-breed sheep in eastern Australia, finding that animal production was higher and enteric methane emissions lower for the same level of intake. At a farm level the reduced pasture intake meant that the farmer increased stocking rates, resulting in a net increase in emissions. This demonstrates the importance of examining interventions at a whole-farm scale.

When examining whole-farm emissions, it is also important to examine results in terms of total or net emissions and productivity. Typically, a decline in total emissions is linked to lower productivity, in the case of livestock production, lower liveweight production. A farmer is unlikely to adopt a practice which reduces productivity and thus profitability, regardless of its impact on farm emissions (Crosson et al., 2011; Foley et al., 2011). Instead, the most common unit of measurement in whole-farm emissions analyses is emissions intensity (EI), or emissions produced per unit of product. An example of the importance of EI is demonstrated through the examination of GHG emissions produced by the Australian beef herd over a 30-year period. During this period, total emissions rose by 19%, however the EI (kg CO_2-e/kg LW) decreased by 14% (Wiedemann et al., 2015d). These productivity gains, achieved through measures such as improved herd productivity and feedlot finishing, would not be evident through examination of emissions alone. Examining the effect of an adaptation or mitigation practice using EI is therefore a valuable approach as it considers the dual goal of productivity gains and emission reduction.

One of the primary benefits of whole-farm system emissions analysis is the flexibility that it affords when addressing various research questions. Whole-farm approaches have been applied to quantify livestock emissions for a number at numerous scales. For example, in the case of beef cattle, analyses have been conducted to quantify emissions of the global livestock herd (Opio et al., 2013), of national herds (Ledgard et al., 2011; Legesse et al., 2016; Wiedemann et al., 2015d), regional herds (Dick et al., 2015a; Pelletier et al., 2010; Toro-Mujica et al., 2017), or individual case study farms (Eady et al., 2011; Peters et al., 2010). At national or regional scales, such analyses are useful for identification of trends or benchmarking analyses, both of interest to policymakers and other stakeholders interested in broader implications of results. For farmers, local or individual farm-scale analyses are of greatest interest, as they enable the examination of results specific to their enterprise type, climate and location (Cottle et al., 2016). There is widespread consensus that analyses of the impact of adaptation or mitigation strategies is recommended to occur at this smaller

scale, as it accounts for the diversity of farming systems and the ensuing differences in effectiveness of strategies between farms (Crosson et al., 2011; Del Prado et al., 2013; Rawnsley et al., 2016).

There are three common types of whole-farm emission analysis tools; biophysical models, whole-farm GHG calculators following national GHG reporting guidelines and Life Cycle Assessment (LCA) frameworks. The suitability of each type depends on the research question posed and the target recipient of the results.

2.4.1.1 Biophysical models

Agricultural biophysical models are mechanistic tools which can simulate detailed whole-farm interactions. In southern Australia such models have been developed for crop systems (APSIM, (Holzworth et al., 2014; Keating et al., 2003), dairy (DairyMod, (Johnson et al., 2008), and pasture (GrassGro, (Freer et al., 1997; Moore et al., 1997).

GrassGro has been used extensively to model different pasture and livestock systems across regions of southern Australia (Alcock & Hegarty, 2011b; Browne et al., 2015; Clark et al., 2003; Cottle et al., 2016; Ghahramani & Moore, 2015; Harrison et al., 2014a; Harrison et al., 2014b; Moore & Ghahramani, 2013; Thomas et al., 2012). The software acts as a decision support tool which can model the productivity, economic or emission performance of a livestock system, in daily time steps, for a chosen interval. It does this through the simulation of interactions within and between the biophysical components (i.e. soil, climate, pasture, animals) and managerial components (i.e. stocking rate, soil fertility, grazing management, livestock reproductive management) of the system. The model is also able to capture temporal variability, by conducting inter-annual simulations.

Biophysical models can be quite complex and require experienced users to successfully conduct in-depth simulations. For example, despite producing farm-specific daily output data, a key limitation of GrassGro in the context of whole-farm emission analysis is that it only considers enteric methane production. Other farm emission sources including, nitrous oxide emissions from soil, manure methane and carbon dioxide from farm input production and transport are excluded. The model also calculates animal emissions using different methodological approaches and EFs to the NGGI recommendations, following Blaxter and Clapperton (1965) for both sheep and beef enteric methane emissions as opposed to Howden et al. (2004) and Charmley et al. (2016), respectively. Regardless, biophysical models are useful for local- or individual farm-scale analyses and the examination of different management practices. They can be scaled up, to regional scale for example, but this would have

to occur through the aggregation of a large sample size of modelled locations.

2.4.1.2 Farm-scale GHG emission calculators

To comply with the GHG emissions reporting requirements under the Paris Agreement, each party to the agreement must follow the guidelines developed by the Intergovernmental Panel on Climate Change (IPCC) (IPCC, 2006) when reporting national emissions. In considering the diversity of global farming systems, the availability of data and the varying level of expertise across participating countries, there is a three-tier system for quantifying emission sources from agriculture using these guidelines. Each successive tier requires a greater level of detail and improves the robustness and accuracy of results.

The adopted Australian methodology for reporting national agricultural emissions follows a mix of Tier 2 (country-specific methodology) and Tier 1 (IPCC international-level methodology) approaches, along with both country-specific and IPCC default emission factors (DEE, 2019). Two key farm-scale static GHG calculators have been developed which specifically follow the NGGI methodologies. These are the Sheep (S-GAF) and Beef (B-GAF) GHG Accounting Tools (Eckard & Taylor, 2016) and the FarmGAS tool (AFI, 2016). While each tool varies slightly in its functions, both produce emission results in line with IPCC requirements and are quite user-friendly in their operation.

In following the IPCC and NGGI, however, these tools are subjected to the same limitations of these approaches. For example, one of the primary criticisms of the IPCC methodology, and thus the NGGI, for modelling farm-level emissions is that it excludes some emissions sources, such as emissions associated with the production and transport of pre-farm inputs and on-farm non-renewable fuel usage (Crosson et al., 2011). The S-GAF and B-GAF tools, as a result, do not consider emissions associated with the production and transportation of farm inputs or pasture residue. FarmGAS excludes these emission sources, along with carbon dioxide sources such as liming or urea application. The exclusion of these emissions means that these tools underestimate whole-farm emissions, to a degree that can be quite substantial in more intensive livestock systems, such as those which incorporate feedlots (Crosson et al., 2011).

The representativeness of the NGGI has been criticised its use of IPCC default EFs for a number of emission sources (Thamo et al., 2013). These EFs are sourced from studies based on Northern Hemisphere agriculture which can have very different agricultural conditions and operations to Australia (Thamo et al., 2013). Further to

this, the NGGI applies default state or national averages for parameters and requires only seasonal or annual inputs. Whilst the GHG calculators enable users to input study-specific parameters to calculate most emission sources, these default parameters are applied if other information is not available. The level of detail inputted into each tool and in the calculated outputs are dependent on the design of the tools themselves. For example, both FarmGAS and S-/B-GAF only enable seasonal or annual calculations, stock classes are pre-defined and do not account for than one feed type simultaneously (i.e. annual or perennial pasture, crop stubble). S-/B-GAFs do not include a cropping component which means it is not possible to account for emissions which may arise from livestock grazing stubble as part of a farm's grazing management strategy, or the emissions associated with producing supplementary feed from on-farm crops. These are all limitations with flow-on effects from the direct application of the Australian GHG reporting guidelines.

Therefore, whilst these tools are useful for benchmarking at a national or state-level, they are less suitable in the analysis of local-scale or individual farm-scale as they cannot consider a number of interactions between farm components (i.e. climate, soil and pasture) and mask the heterogeneity which exists at this smaller scale (Young et al., 2016). This can have ramifications when examining the effectiveness of mitigation strategies, for example, or the comparison of different farming systems.

2.4.1.3 Life cycle assessment

Life cycle assessment (LCA) is a well-established research method used to evaluate and quantify the resource use and multiple environmental impacts associated with the production and use of a product. Practitioners are guided by a set of core standards (ISO, 2006a, 2006b) which separates LCA into four phases (Figure 2.5).

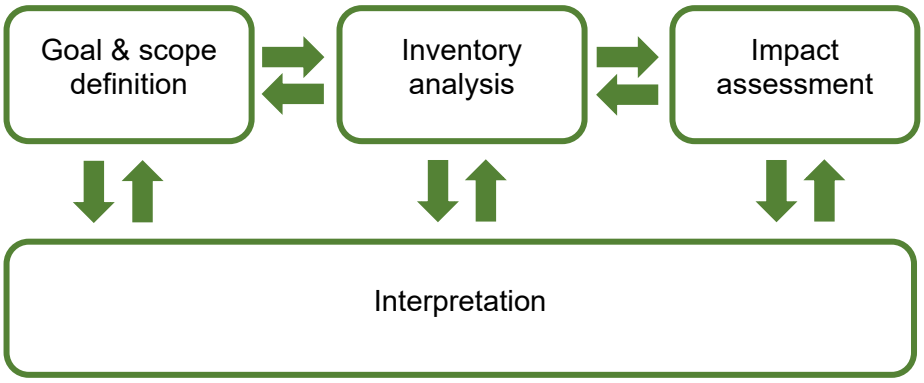


Figure 2.5 - The four phases of life cycle assessment (LCA) (Adapted from (ISO, 2006a))

1. **Goal and scope definition**

This phase involves firstly, the establishment of the study objectives along with the intended use of results. The scope of the study is then clearly defined in a manner that reflects the goal through;

i) The functional unit (FU) reflects the product being assessed and is a quantitative measure of the function of the system (i.e. kg liveweight, kg greasy wool), enabling comparison between different systems or when applying mitigation strategies (LEAP, 2016).

ii) The clear setting out of the system boundaries of the study which determine both the stages of the production system are considered and hence which inputs and outputs are included or excluded.

iii) The selection of impact categories to be assessed; that is, the environmental impacts (i.e. climate change, acidification, water use or resource depletion) associated with the product's life cycle.

iv) Where a production system produces co-products along with the product in question, choice of method to allocate environmental impact between products (see section x for more detail).

2. **Inventory analysis**

All inputs (resources used) and outputs (products, co-products, emissions and waste generated) within the system boundaries are quantified and compiled into a life cycle inventory (LCI). This phase is essential as the results of an LCA are only as good as the data included in the inventory. Inventories range in complexities, with the inventories of agricultural products typically comprising several hundreds of measured and modelled inputs and outputs.

3. **Life cycle impact assessment**

Following completion of the LCI, the flows of resource use and generated emissions are assigned into the chosen impact categories (i.e. climate change) and converted to common units (i.e. CO_2-e) using characterisation factors (i.e. GWPs) to obtain an aggregated value for the impact category examined. This conversion into common units makes it possible to examine the potential environmental impact of individual inventory flows, a particular stage of the product system, or the entire life cycle of the product in question.

4. **Interpretation**

LCA is an iterative process and the interpretation phase is important to:

i) ensure the robustness and accuracy of data and results.

ii) evaluate the inventory analysis and impact assessment against the study goals and scope.

Once these are deemed satisfactory, interpretation then involved the development of study conclusions and recommendations.

Initially, LCA was applied to industrial products and processes (Roy et al., 2009), whereby outputs typically are produced using fixed outputs with little to no spatial or temporal variation. Agricultural products by contrast, are produced in complex biological systems which typically display high intra- and inter-annual variability and will differ substantially across locations. Further to this, unlike most industrial processes, agricultural systems self-produce many of the resources utilised in the production process, such as soil, pasture and livestock, with complex interactions between each (Harris & Narayamaswamy, 2009). Consequently, the inventory stage of an agricultural LCA is more complex than those of manufacturing or other industrial LCAs, and typically involves extensive modelling to define both the inputs and outputs of the system.

Despite the added complexities, if applied appropriately with data of sufficient quality, LCA is a valuable tool to quantify the environmental impact of an agricultural product. It enables the calculation of all emission flows within the system, both those recommended in the IPCC Guidelines and the NGGI, along with others not considered in other approaches, such as emissions associated with the production and transportation of pre-farm inputs. Although subject to a number of the limitations experienced by the GHG calculators described in the previous section, such as the use of some IPCC default EFs and the fact it is also a static model not a biophysical model, LCA allows flexibility in terms of calculations and is adjustable to address the specific research question posed. Livestock LCAs can be conducted using LCA-specific software such as Simapro (PRé Consultants, 2014; Ruviaro et al., 2015; Wiedemann et al., 2016a) or in tailor-made frameworks with varying degrees of complexity (Dougherty et al., 2018; FAO, 2016; O'Brien et al., 2016). Depending on the data applied, LCA can also examine impacts at the animal, herd, paddock or system level, across regions and years, enhancing the scope of potential analyses. It can be applied to quantify the overall impact of producing an agricultural product, to identify “hotspots” or components of the system which contribute the most to the overall impact, or to examine the potential effect of adaptation or mitigation practices across the farming system. Section 2.5 highlights this diversity in approach through a review of conducted livestock LCA studies.

In dealing with such complex systems there arises a number of methodological

considerations such as how to allocate impacts across multiple products (i.e. wool and meat) and how to compare studies given the range of assumptions that can be made in terms of data use, system boundaries and emission methodologies. These considerations are explored further in later Chapters.

In LCA studies the livestock production system is commonly broken into three core stages; feed production (cradle-to-mouth), animal production (cradle-to-farm-gate; where upstream processes aside from feed-related are considered) and primary processing (primary wool, milk and meat processing facilities) (LEAP, 2016). Other downstream stages less considered include secondary processing (production of more complex food products, clothing), retail distribution, human consumption and waste management. Traditionally, full LCAs consider the entire life cycle of a product (production > processing > distribution > consumption > waste). However, due to the complexity of the livestock system with its extensive supply chains and many product pathways, most livestock LCA studies focus on one or two of these stages and are considered “partial LCAs”. The stage which receives the most research attention is the farm, or primary production, stage as it is typically responsible for the greatest proportion of impact along the supply chain (Ledgard et al., 2011; Lieffering et al., 2011; Wiedemann et al., 2015b).

LCAs often consider multiple environmental impacts, such as GHG emissions, land use, water consumption, non-renewable energy use and eutrophication. In agricultural LCAs, for certain impact categories there remains a high degree of uncertainty (McClelland et al., 2018). However, methodological approaches are continually being refined and data availability improved and so more and more, studies are considering multiple impacts. A recent review found that even with these improvement and recommendations for multi-criteria analyses, most studies only consider one to three impact categories (McClelland et al., 2018). Overwhelmingly, studies which consider GHG emissions dominate livestock LCA studies. A study which considers only GHG emissions is termed a “Carbon Footprint”, as opposed to a multi-criterion LCA (Cowie et al., 2012). Despite this, a carbon footprint analysis adopts the same approach and is effectively a subset of a complete LCA (Desjardins et al., 2012; Jones et al., 2014). Similar to LCAs, carbon footprint analyses must also adhere to a set of core standards, ISO 14067 (ISO, 2018).

2.4.1.4 Integration of tools to conduct whole-farm system analyses

Each of the tools described in the previous sections present both advantages and limitations in their ability to conduct whole-farm system GHG emission analyses. For

example, biophysical models such as GrassGro are able to capture and model the interactions between climate, soil, plants and animals in such a way that can calculate detailed daily timestep data for the location under examination, however in terms of GHG emissions, only calculate enteric methane emissions and do so using a different approach to national reporting recommendations. Farm-scale GHG calculators, while in line with international and national emission reporting requirements and quite user-friendly, do not consider all emission sources. Default EFs and parameters are based on state or national averages, along with a restricted scope of analysis and means that whilst these tools may be useful for benchmarking at a national or state-level, they are less suitable in analyses at local-scale or individual farm-scale or examination of management practices. LCA, or carbon footprint analysis in the context of GHG emissions, considers all emissions arising on a livestock production system, regardless of the type of application. In line with good practice, animal and soil emissions are calculated following IPCC and NGGI requirements, which means that LCA is also subject to default agriculture EFs. However, LCA has greater flexibility than other tools as it can be conducted outside of LCA-specific software and can be developed in such a way that allows it to accommodate a more in-depth analysis of whole-farm emissions than other static whole-farm models.

To overcome the limitations associated with individual whole-farm tools, numerous studies have instead adjusted their methodological approaches to accommodate multiple tools. Harrison et al. (2014a), for example, aggregated the daily time step outputs from GrassGro to seasonal emissions calculated using S-GAF, to examine the effect of various management practices on an experimental farm in Victoria, Australia. Similar approaches were adopted by other sheep and beef production studies located in south-eastern Australia (Browne et al., 2015; Browne et al., 2011; Doran-Browne et al., 2015). Cottle and Cowie (2016) and Cottle et al. (2016) integrated daily GrassGro outputs into monthly emissions calculations following NGGI guidelines to quantify emissions across multiple enterprises in southern Australia. In a different approach, Eady et al. (2011) combined B-GAF and FarmGas with Simapro to calculate whole-farm emissions from case study beef cattle enterprises in Queensland, Australia, enabling consideration of all emission sources. Others have combined all three tools, such as Brock et al. (2013) who used daily timestep data from GrassGro to generate seasonal animal and soil emissions in S-GAF which were then entered into Simapro to conduct an LCA of wool production in NSW, Australia. Both southern Australian beef enterprises and livestock production enterprises in south-western Australia are underrepresented in studies which integrate such farm-

scale tools.

The integration of multiple tools has been adopted as the chosen approach in several international studies. For example Beukes et al. (2011) used a country-specific mechanistic biophysical model in conjunction with a whole-farm GHG calculator to examine the effect of various mitigation strategies on the GHG emissions associated with dairy production in New Zealand. Foley et al. (2011) combined the output of a bioeconomic model with a tailor-made GHG emission calculator to examine the effect of different beef production systems in Ireland. Toro-Mujica et al. (2017) applied a country-specific bioeconomic model in line with an LCA approach to develop an approach which enabled the calculation of the carbon footprint of sheep production in the semi-arid zone of Chile.

The integration of multiple farm-scale tools has the potential to improve the accuracy of whole-farm results. Integrating outputs from biophysical models into emission calculators or LCA frameworks reduces the uncertainties associated with modelling individual emission sources, particularly those dependent on animal and plant parameters such as enteric methane and excreta nitrous oxide emissions. Integration of biophysical models and NGGI-following GHG calculators, whilst a valuable approach, does not incorporate all emission sources. In addition to this, it often results in the daily time step data of GrassGro being aggregated to seasonal averages. This makes it difficult to capture the intra-annual variations which may occur at monthly intervals (Cottle & Cowie, 2016; Cottle et al., 2016). For example, growth and feed quality patterns of pasture species which do not align with the traditional seasons or animal practices such as lambing/calving or sale which occur predominantly across specific one or two months. As demonstrated above, there is a lack of Australian research which uses integrated biophysical models and LCA approaches. Given that both tools can be suitable for farm or local-scale analyses, there is great potential for their integrated application in the examination of the carbon footprint of livestock enterprises in southern Australia.

2.5 A review of whole-farm system GHG emission analyses in livestock production systems

This Section presents a review of international and Australian livestock LCAs and carbon footprint studies. Internationally, there is a plethora of sheep and beef cattle LCAs and carbon footprint studies. This Section sets out to highlight those deemed most pertinent to the present study. The merits of different approaches, methodological considerations and current gaps in research are explored.

In considering Australian-centric research, alongside the existing LCA and carbon footprint studies examined (Table 2.1), this review considers select livestock studies which have conducted whole-farm system GHG emission analyses. As these studies exclude one or more emissions sources (i.e. pre-farm emissions, soil emissions) and have not followed the four phases of LCA, they cannot be considered carbon footprint analyses. However, as they conduct detailed analyses of the livestock component, often through the integration of biophysical modelling and examination of target mitigation strategies, they have contributed to the further understanding of whole-farm system analysis. These studies are considered in in the context of potential mitigation strategies for adoption in the later sections of this Chapter.

2.5.1 Common goals of whole-farm carbon footprint studies

In their examination of global livestock LCA and carbon footprint studies, LEAP (2016) stated that the goals of livestock studies largely focus on one of three categories; “hotspot identification”, “commodity analysis” and “benchmarking”. Henry (2012) grouped these three categories into two, highlighting that the goals of these studies usually align with the scale upon which they are conducted.

- Firstly, comparative analyses of commodities and/or benchmarking analyses are typically conducted at regional or country scale. These studies predominantly apply regional, national, or even international data to produce results.
- Secondly, studies which examine specific supply chains for the purpose of hotspot identification or comparative analyses of methods of production, for example. These studies predominantly apply case study farm or modelled case study farm data.
- In addition to the above, the literature review in the present study determined an additional goal in a smaller subset of studies, the analysis of mitigation strategies. Such analyses have been conducted both with the application of regional or countries average data, as well as farm-specific data, with varying outcomes.

A large proportion of studies, particularly earlier publications, conducted benchmarking and/or commodity analyses through the application of regional or national averages. For example, Ledgard et al. (2011) and Lieffering et al. (2011) employed national survey data to ascertain the carbon footprint of exporting lamb and beef, respectively, from New Zealand to the UK. Nguyen et al. (2010) quantified the carbon footprint of “typical” beef production systems in the European Union (EU),

applying EU averages. Similarly, Casey and Holden (2006) and Mogensen et al. (2015) applied national survey data to quantify country-level carbon footprint estimates of “typical” beef production systems found in Ireland, Denmark and Sweden, respectively. The carbon footprint of sheep production in Chile and Australia has also been benchmarked using national and state averages (Toro-Mujica et al., 2017; Wiedemann et al., 2016b). Vergé et al. (2008), Legesse et al. (2016), Capper (2011) and Wiedemann et al. (2015d) used census and national data to generate comparative carbon footprints of national beef production across decades (i.e. 1986-2010) in Canada, the USA and Australia, respectively. Other studies have applied large-scale regional averages to compare livestock production across regions within a country (Pelletier et al., 2010; Wiedemann et al., 2015c; Wiedemann et al., 2016c). The primary benefits of conducting analyses at this scale are that they generate useful production averages to inform policy makers and can also act as scoping analyses to generate findings for further investigation.

However, the use of the census or regional/national averages which enable the generation of estimates of the environmental impact at national or regional levels, also means that downscaling the findings to farm or local levels is accompanied by high risks of misinterpretation. A production average is not representative of the temporal or spatial differences which occur within that average. For example, Casey and Holden (2006) examined the effect of different strategies, such as shorter turnoff and different dietary components, on the carbon footprint of typical Irish beef production generated using the national averages. They then made recommendations for future application of strategies based on the analyses conducted on the “national average” production system. Livestock production systems are extremely diverse and so the suitability and magnitude of impact of such strategies will also vary across those systems. For example, in the eastern Australian beef production benchmarking analysis conducted by Wiedemann et al. (2015c), results were generated for both regional average farms (RAFs) modelled from ABARES data and case study farms (CSFs) in the same region. For both regions considered, RAFs overstated the average carbon footprint obtained from CSFs in that respective region, by 3-7%. The footprint of the CSFs varied by up to 25%. Applying the same approach to sheep production in southern Australia, RAFs overstated the average footprint of the CSFs by 4-23% (Wiedemann et al., 2016c). Individual footprints for the CSFs in this study were unavailable so it was not possible to calculate the variation between these CSFs. The approach adopted by these studies is useful as it considers both regional averages and individual farms. It also demonstrates that along with the diversity of

impacts across different farms, regional averages can be quite distinct from averages taken by a sample of case study farms. These studies did not investigate why these footprints differed and did not provide comprehensive enough results to enable such an investigation. They highlight why caution should be made when downscaling or extrapolating findings using such averages.

A second limitation of applying large-scale average data to conduct carbon footprint analyses is the interpretation afforded. Livestock systems are complex and thorough carbon footprint analyses of such systems involve the integration of large quantities of inter-related parameters. Census and national data are often not available at this level of detail and this can make the ensuing interpretation of the carbon footprint difficult as it may not be possible to identify influencing factors. For example, Dyer et al. (2014) compared the carbon footprint of sheep and beef production in Canada, alongside a comparison across eastern and western Canada, through the application of census data into a GHG calculator. They found that the EI of sheep production (expressed as kg CO_2-e/kg protein) was higher than that of beef production and that there were differences across regions. However, the ability to investigate the factors influencing this difference was limited as the data and model did not distinguish between several potentially influencing factors, such as animal fecundity, breed, slaughter weights and age amongst production livestock. Whilst comparisons were made within the study and with others, without the ability to conduct detailed analyses of the factors influencing these EIs, these were largely inferences from final EI values. Whilst studies such as this are useful in scenarios where no prior carbon footprint has been conducted for the country, region, or industry in question, their primary benefit is clearly as a scoping analysis as the data is often not comprehensive or representative enough to conduct more in-depth analysis.

The review of literature revealed that livestock LCA and carbon footprint studies which examine specific supply chains through the application of case study data or modelled case study data are increasingly common. Though such approaches were evident in earlier studies (Beauchemin et al., 2010; Eady et al., 2011; Edwards-Jones et al., 2009; Peters et al., 2010; Veysset et al., 2010), recent studies apply farm-specific data in across benchmarking/hotspot analyses (Bragaglio et al., 2018; Cerri et al., 2016; Dougherty et al., 2018), examination of farm practices with productivity and mitigation potential (Becoña et al., 2014; Hyland et al., 2016; Jones et al., 2014; Nieto et al., 2018; Toro-Mujica et al., 2017; Veysset et al., 2014) and in-depth analysis of productivity and mitigation strategies (Bogaerts et al., 2017; McAuliffe et al., 2018; Stanley et al., 2018). The primary benefit of farm-scale approaches such as these are

that they capture the diversity of farming systems and can identify farm characteristics and practices which influence the carbon footprint of a system. Such a targeted approach is recognised as the most appropriate method for examining the impact of mitigation strategies as the calculated effectiveness of a strategy is farm-specific (Crosson et al., 2011; Del Prado et al., 2013; Rawnsley et al., 2016). Whilst benchmarking and hotspot analyses conducted using regional averages may present information of most interest to policy makers and to industry as scoping analyses, farm-specific analyses provide information directly relevant to producers and a targeted pathway to increased on-farm mitigation.

2.5.2 Methodological considerations

Whole-farm analyses are a preferred approach in the quantification of the environmental impact of a product or process. However, the calculated impact is not only determined by variations in primary data, such as specific farm characteristics and practices, or data quality as described above, but also by variations in methodological approaches (Basset-Mens et al., 2009). Despite attempts to streamline methodologies and data quality control across livestock carbon footprint and LCA studies (LEAP, 2015a, 2015b, 2016), there remains difficulty in comparing the results of whole-farm studies because of such diversity in approaches. Table 2.1 demonstrates the heterogeneity of approaches across Australian studies, with differences in the functional unit, systems boundaries, allocation method, emission calculations and data sources applied. In some cases, the goals of these studies overlap, however despite this, a direct comparison of results is fraught with risks of misinterpretation of findings. Yet studies continue to make such comparisons, despite wide recognition across the field that differences in results are only partly a result of the differences in farm characteristics and practices that they are examining, but mostly a result of divergences in methodological approaches and data quality control (Crosson et al., 2011; de Vries & de Boer, 2010; Del Prado et al., 2013; Zervas & Tsiplakou, 2012).

This Section outlines some of the primary methodological considerations when comparing results across studies. These considerations, particularly in the context of the literature reviewed, played an important role in the development of the research design in the present study and interpretation of findings.

2.5.2.1 Functional unit

Whilst the results of whole-farm analyses are typically presented in terms of EI, a measure of productivity which theoretically should enable comparison between

products and studies, its application as a metric of emissions efficiency of livestock production systems is limited as it is dependent on the methodological approach and data quality of the study itself (Flysjö et al., 2011).

Along with this, the adopted FU in livestock studies varies widely, with units of measurement ranging from livestock units sold (i.e. finished beef calf; Ogino et al. 2004, 2007), total liveweight sold (i.e. kg LW; Hyland et al., 2016; Wiedemann et al., 2016b, liveweight sold from a particular stock class (i.e. kg lamb sold; Toro-Mujica et al., 2017; Dougherty et al., 2018), carcass weight sold (i.e. kg CW sold; Beauchemin et al., 2010; Stanley et al., 2018), liveweight gain on-farm (i.e. kg LWG; Ruviaro et al., 2015); Cerri et al., 2016), protein sold (kg protein sold; Vergé et al., 2014), to area grazed (i.e. ha; Foley et al., 2011; Nieto et al., 2018). Such variation is also exhibited between Australian studies (Table 2.1). It is not possible to compare studies which have employed different EIs. Whilst conversion to comparable metrics is possible, it requires that the respective studies present their inventories (i.e. livestock characteristics) and emissions results in sufficient detail. However, the detail of inventory and emission information included in studies varies widely, making comparisons between FU and other results, such as hotspots, difficult.

A second consideration is the suitability of the FU chosen. For example, adopting a FU of carcass weight where the study system boundary is at the farm-gate represents a mismatch of metrics as the boundary is prior to slaughter (LEAP, 2016). This will increase EI as compared to studies which consider liveweight at the farm gate. Similarly, the consideration of only one stock class in the FU (i.e. lamb) without allocating emissions between stock classes sold can drive up EI as it does not consider total product sold (i.e. cull animals). A third example is the use of EIs based on animal product versus farm area. Foley et al. (2011) found that in terms of emissions/ha, low producing farms were more emissions efficient than intensive farms in the USA, however, when emissions efficiency was examined in terms of carcass weight produced, the intensive farms were between 4 and 18% more efficient. Nieto et al. (2018) compared backgrounding and cow-calf pasture systems in Argentina, finding that on a per hectare basis, backgrounding enterprises were 30% less efficient than pasture-based systems, yet in terms of liveweight production, backgrounding was 71% more efficient. Presentation of results in terms of farm area can thus imply that poorly managed pastures at low stocking rates are more efficient than productive pastures with higher carrying capacities. These examples highlight the importance of FU and the potential for misleading interpretation when comparing different FUs.

2.5.2.2 System boundary

A second methodological consideration is the acknowledged inconsistencies in system boundaries across studies (Desjardins et al., 2012). While some studies consider post-farm stages such as processing (Peters et al., 2010) and export (Wiedemann et al., 2015b; Ledgard et al., 2011; Lieffering et al., 2011) in line with their specific research questions, others can omit activities within stages which can impact the carbon footprint, such as the exclusion of breeding herd activities (Biswas et al., 2010; McAuliffe et al., 2018; Ogino et al., 2004), typically the largest source of on-farm emissions. Multiple European studies integrate surplus animals from dairy systems into their investigations of beef cattle carbon footprints (Casey & Holden; 2006; Nguyen et al., 2010; Mogensen, 2015). Importantly, studies may omit emission sources whose inclusion is required in line with good practice guidelines (LEAP, 2015a; 2015b; 2016), such as pre-farm emissions, soil emissions and emissions associated with LU and dLUC. While LU and dLUC are excluded by most studies or reported separately following such guidelines in acknowledgement of the high degree of uncertainty the exists regarding its calculation and data quality, other emission sources should be included as common practice. For example, few studies include pasture residue emissions (Eady et al., 2011; Brock et al., 2013; Cottle & Cowie, 2016) despite presented methodologies in IPCC and NGGI guidelines, while others exclude soil or certain pre-farm emissions (Casey & Holden, 2006; Foley et al., 2011; Ripoll-Bosch et al., 2013; Mogensen et al., 2015). Such omissions have been made across multiple whole-farm analyses conducted in Australia which have instead prioritised examination of animal emissions such as enteric methane (Browne et al., 2011; Browne et al., 2015; Harrison et al., 2015; Cottle et al., 2016; Cullen et al., 2006; Harrison et al., 2016; Taylor et al., 2016). It is important to consider such variations when comparing studies as few are alike in their choice of boundaries.

Comparison can be made where results are amended to reflect boundaries, through the exclusion of post-farm stages or alignment of emission sources included for example. However, this can only be performed where studies present findings in sufficient detail, for example inter- and intra-emission source breakdowns. However, as for FU considerations, the inconsistency in presented results across studies means that such interpretation is rarely possible

2.5.2.3 Allocation of co-products

Where livestock production systems produce multiple co-products (i.e. wool, meat, milk) the calculated environmental impact of the system must be allocated between

these co-products. The approach adopted to handle co-production must be appropriate to the specific farming system in question and carefully considered as the choice of methods can have a considerable impact on final distribution of impacts (ISO, 2006b). Whilst Chapter Three outlines the specific approaches, this Section presents key challenges and criticisms.

Dealing with co-production in LCA has been, and continues to be, a subject of debate and discussion amongst practitioners. The key issues raised are that of the inconsistencies in approaches to solve co-production; driven by the difficulty in the interpretation and implementation of ISO standards (Pelletier et al., 2015), particularly in complex biological systems where causality is not as easily identified as in industrial processes (Mackenzie et al., 2017). The interconnectivity between outputs in agricultural systems makes it difficult to apply consistent principles. On top of this, few published LCA studies provide justification for and details of, their specific chosen approach. This produces a range of largely incomparable results where different allocation methods can often contradict each other, and the choice of one method merely represents one solution out of many. To combat this, there have been numerous calls for greater clarity to be provided alongside the ISO standards and the provision of more detailed guidelines of which how and when to adopt different approaches for handling co-products (Curran, 2007; Mackenzie et al., 2017; Pelletier et al., 2015; Weidema, 2018). Though there have been efforts to combat this with publications such as LEAP (2015b, 2016), the issues with inconsistent co-handling methods as described above are still prevalent.

Where the handling of co-products is required, most studies opt to allocate environmental impact between products. Biophysical approaches are the preferred form of allocation; however, most studies refer to simpler forms of allocation such as economic or mass allocation to avoid the complexity of this method. Another criticism is misuse of biophysical allocation by practitioners who use common physical properties as a basis for the allocation without justification of how these reflect *causal* relationships in the system, which means the approach is misleading (Mackenzie et al., 2017). In other situations, practitioners may apply biophysical allocation as preference over economic allocation following ISO guidelines, to outputs that are only classified as co-products *because* of their perceived economic value (i.e. manure from feedlot). The basis of this allocation is thus purely economic, which is what they were avoiding in the first place (Mackenzie et al., 2017). Historically, economic allocation has presented as the most common form of allocation (Biswas, Graham, Kelly, & John, 2010; Brock et al., 2013; de Vries & de Boer, 2010) due to its simplicity.

However economic allocation factors are susceptible to market price volatility and because of their reliance on specific market prices, obtained results cannot be compared between regions, time periods, or livestock breeds, for example. Wiedemann et al. (2015a) argued that economic allocation also opens the possibility of burden shifting between different products (i.e. wool and meat) as economic value changes over time. For these reasons, biophysical allocation methodologies are the more preferred approach in current LCA and carbon footprint analyses.

The above reasons highlight that not only is the choice of allocation method critical, but the results of studies which have applied different allocation methods cannot be directly compared. Numerous Australian (Cottle & Cowie, 2016; Eady et al., 2012; Wiedemann et al., 2015a) and international studies (Casey & Holden, 2005; Cederberg & Stadig, 2003; Dougherty et al., 2018; Flysjö et al., 2011; Gac et al., 2014; Lieffering et al., 2011; Nguyen et al., 2012; Weiler et al., 2014) have examined the effect of different partitioning and allocation approaches in beef cattle and sheep production systems, highlighting the high variance (up to 50%) in results due to those different allocation assumptions. There can also be high variations within allocation types; for example, the price of wool is higher in Australia than in other countries where the primary focus of sheep production is meat (i.e. Canada). In such cases, the allocation of emissions to sheep meat will be lower in Australia and differences in respective EIs between the countries may not represent efficiency differences, but rather allocation. This is explored further in Chapter Five.

2.5.2.4 Emission calculations and emission factors

The choice of methodology to calculate emissions can have substantial impact on the carbon footprint of livestock production, particularly when the emission source is traditionally a hotspot such as enteric methane (Brock et al., 2013; Dougherty et al., 2018). For example, Tier one approaches to calculate enteric methane production apply a flat conversion rate, regardless of animal or feed characteristics (IPCC, 2006). Given that the primary uncertainties in the calculation of enteric methane lie with the characterisation of livestock diets and growth curves, it follows that Tier one approaches have assigned uncertainties of up to 50% (Crosson et al., 2011). In instances where Tier one approaches are adopted (Cerri et al., 2016; Edwards-Jones et al., 2009; Jones et al., 2014) the performance of livestock systems will not reflect the inherent heterogeneity between and within farms, regardless of the other farm characteristics considered.

By contrast, Tier two methodologies do consider animal and diet characteristics,

reflected in the assigned uncertainty of 20%, and are the most common approach of international studies. However, while detailed animal information is required for this approach, animal diet is considered only through a conversion factor to account for quality, with factors provided for roughage-dominant and concentrate-dominant diets only. Whilst this enables examination of strategies such as intensification through concentrate diets (O'Brien et al., 2016; Pelletier et al., 2010), without the adoption of specific factors there is no differentiation across countries and region, not does it account for different pasture types or intra-annual variations. The inability to consider the influence of pasture using these IPCC approaches could provide reason as to why so few studies have examined the potential impact of pasture or temporal changes on the carbon footprint of livestock enterprises.

The varying uncertainty in such approaches highlights why caution must be taken when comparing studies which have adopted different approaches. Comparing the results of studies to those with methodologies which attempt to reduce this uncertainty further, such as the NGGI which considers seasonal differences in animal and pasture characteristics or biophysical model outputs which present highly specific and daily timestep results, is likely to present even greater risk of misinterpretation. In fact, the differences in carbon footprints following the application of difference methodological approaches be as much as 15% (Dougherty et al., 2018) or 27% (Brock et al., 2013). In another example, Vergé et al. (2008) and (Legesse et al., 2016) both calculated the carbon footprint of the Canadian beef industry in overlapping time periods, obtaining different EIs (8-14% differences) for each of the periods examined. The differences, as noted by the studies, was not due to data sources, rather different methodological approaches, particularly for the calculation of enteric methane. Despite such uncertainties, most studies continue to make direct comparisons with no acknowledgement of the different approaches adopted, the uncertainty which accompanies such approaches and that the presented differences are, in part due to methodological differences, not differences in farming systems or employed mitigation.

Another consideration is the change in recommended methodologies to calculate emissions as improved approaches are developed. For example, prior to 2018 the NGGI recommended the calculation of cattle intake following the approach of Blaxter and Clapperton (1965). This has now been replaced by the methodology developed by Charmley et al (2016). Similarly, the NGGI removed its requirement for the calculation of emissions arising from the N-fixation of legume pasture and crops in 2018, alongside the introduction of the requirement to calculate pasture residue

emissions in line with IPCC methodologies. On a broader scale, the specific GWPs adopted by the IPCC have changed with the release of each report, altering the respective weightings of nitrous oxide and methane. In each case, these changes will influence the final carbon footprint of the livestock system under analysis. Comparison with studies conducted prior to such changes will reflect these differences in methodologies, masking any accompanying differences in system characteristics or management practices.

Table 2.1 - Summary of Australian beef cattle and sheep LCA and carbon footprint studies

Study	Country/ region	Goal	LCA/ carbon footprint	FU	Data source	System boundary	GHG methodology	Allocation	Mitigation analysis	EI
Beef cattle and sheep production										
(Peters et al., 2010)	Australia (sheep - WA; beef - NSW & Vic)	To examine Australian red meat production to improve accuracy of research approach	LCA – 4 impact categories	Kg HSCW	CSF (livestock & farm input data) NGGI average (pasture data)	Cradle to processing plant exit	NGGI	Mass	No	WA sheep – 7.2-8.3; Vic beef – 8.2-11.5 NSW beef – 9.8-10.2 kg CO_2-e/ kg HSCW
(Wiedemann et al., 2015b)	Australia (sheep – NSW, SA & Vic; beef – NSW & QLD); USA	To determine the impact of the production, processing and transport of Australian beef and lamb to the USA	LCA – 4 impact categories	Kg retail cuts of beef/ lamb	CSF and modelled RAF (data sourced from (Wiedemann et al., 2015c; Wiedemann et al., 2016b)) National average data (Feedlot, processing & transport data)	Cradle in Aust. to pre-retail distribution in USA	IPCC Tier 2 - cattle enteric CH_4 NGGI- sheep enteric CH_4 & all other sources	Biophysical (sheep on-farm stage) Economic (processing stage)	No	Sheep – 16.1 kg CO_2-e/kg lamb Beef – 23.4-27.2 kg CO_2-e/ kg beef
Beef cattle production										
(Eady et al., 2011)	Australia (QLD)	To conduct LCAs of two beef production enterprises in QLD	LCA - 2 impact categories	Kg LW	CSF (livestock and farm input data) NGGI average (pasture data)	Cradle to farm gate	NGGI	Economic - stock class division	No	14.4-20.8 kg CO_2-e/ kg LW*
(Ridoutt et al., 2011)	Australia (NSW)	To calculate and compare the water and carbon footprints of six beef enterprises in southern Australia	LCA – 2 impact categories	Kg LW	Simulated CSF using state average data	Cradle to farm gate	NGGI	Economic (for culls)	No	10.1-12.7 kg CO_2-e/ kg LW
(Wiedemann et al., 2015c)	Australia (NSW & QLD)	To conduct a benchmarking LCA of grass-fed beef production in eastern Australia	LCA – 4 impact categories	Kg LW	CSF (livestock and farm input data) Modelled RAF using state survey, CSF and NGGI data Sources for pasture data not stated	Cradle to farm gate	IPCC Tier 2 - Enteric CH_4 NGGI – all other sources	None	No	CSF - 10.6-11.9 RAF – 12.2-12.4 kg CO_2-e/ kg LW
(Wiedemann et al., 2015d)	Australia	To quantify the trend in the impact of beef cattle production from 1981-2010	LCA – 4 impact categories	Kg LW	National survey data (livestock & farm input data) NGGI average (pasture data)	Cradle to farm gate	NGGI	None	No	1981 – 15.3 2010 – 13.1 kg CO_2-e/ kg LW
(Taylor & Eckard, 2016)	Australia (QLD)	To quantify the carbon footprint of 3 beef backgrounding herds	Carbon footprint	Kg LW	CSF (livestock & farm input data) Sources for pasture attributes data not stated	Gate to gate	NGGI	None	Yes	Baseline- 4.3-6.3 Scenarios- 4.1-5.4 kg CO_2-e/ kg LW
(Wiedemann et al., 2016a)	Australia (NSW, QLD, Vic)	To conduct a benchmarking LCA of feedlot finished beef	LCA – 4 impact categories	Kg LWG	CSF (livestock & feedlot input data)	Gate to gate	IPCC Tier 2 - enteric CH_4 NGGI - all other sources	None	No	4.6-9.5 kg CO_2-e/ kg LWG

Sheep production										
(Biswas et al., 2010)	Australia (Vic)	To compare to the carbon footprint of wheat, sheep meat and wool	Carbon footprint	Kg sheep meat	Research trial (livestock & experimental plot data) Sources for pasture/crop stubble attributes data not stated	Gate to gate	Flat conv-ersion rates - enteric CH_4 Experimental data - N_sO emissions	Economic	No	5.1-5.6 kg CO_2-e/ kg sheep meat
(Eady et al., 2012)	Australia (WA)	To quantify the carbon footprint of agricultural system with multiple co-products	Carbon footprint	Animal unit	CSF (livestock and farm input data) Sources for pasture/crop stubble attributes not stated	Cradle to farm gate	NGGI	Biophysical & economic	No	Biophysical – 2.6-3.7 kg CO_2e-/kg LW Economic – 2.7 kg CO_2-e/kg LW cull ewe, 6.2 kg CO_2-e/kg LW lamb *
(Brock et al., 2013)	Australia (NSW)	To quantify the carbon footprint of wool production in the Southern Tablelands of NSW	Carbon footprint	Kg greasy wool	CSF (GrassGro model input data & farm input data) CSF-specific GrassGro modelled output (livestock & pasture data)	Cradle to farm gate	NGGI	Economic	No	24.9 kg CO_2-e/kg greasy wool 5.3 kg CO_2-e/ kg LW*
(Cottle & Cowie, 2016)	Australia (NSW & WA)	To quantify the carbon footprint of sheep production, through the examination of different allocation methods	Carbon footprint	Kg greasy wool	CSF (livestock & farm input data) Sources for pasture/stubble attributes not stated One CSF sourced from Wiedemann and Yan (2014)	Cradle to farm gate	NGGI	Protein mass, mass & economic	No	PMA – 20.7 Mass – 8.5 Economic – 35.8 kg CO_2-e/kg wool; PMA – 6.3 Mass – 8.5 Economic – 3.6 kg CO_2-e/ kg LW
(Wiedemann et al., 2016b)	Australia (SA, NSW & Vic)	To conduct a benchmarking LCA of the production of export lamb	LCA – 5 impact categories	Kg LW	CSF (livestock & farm input data) Modelled RAF using state survey data & NGGI data Pasture intake calculated to be residual from NGGI predicted feed intake less supplementary feed. Feed attribute data used to calculate these values not stated	Cradle to farm gate	NGGI	Protein mass	No	CSF – 6.0- 6.2 RAF – 6.5- 7.3 kg CO_2-e/ kg LW
(Wiedemann et al., 2016c)	Australia (NSW, SA & WA)	To conduct a benchmarking LCA of wool production in major Australian wool producing regions	LCA – 5 impact categories	Kg greasy wool	CSF (livestock & farm input data) Modelled RAF using state survey & NGGI data Pasture intake calculated to be residual from NGGI predicted feed intake less supplementary feed. Pasture attributes assessed visually on-farm	Cradle to farm gate	NGGI	Protein mass	No	CSF – 19.5-25.0 RAF – 20.1-21.3 kg CO_2-e/ kg greasy wool

2.5.3 Gaps in International and Australian research

2.5.3.1 Geographical distribution of livestock studies

The literature review revealed that, to-date, livestock LCA and carbon footprint research has predominantly been focussed on OECD countries. With the exception of smallholder dairy farming in Kenya (Weiler et al., 2014) and smallholder beef production in Thailand (Ogino et al., 2016), the only other studies conducted in non-OECD countries include a suite of beef cattle carbon footprint analyses conducted in Uruguay (Becoña et al., 2014; Modernel et al., 2013; Picasso et al., 2014), Argentina (Nieto et al., 2018) and Brazil (Bogaerts et al., 2017; Cederberg et al., 2011; Cerri et al., 2016; de Figueiredo et al., 2017; Dick et al., 2015a, 2015b; Ruviaro et al., 2015; Willers et al., 2017). The concentration of studies in these southern American countries reflects the importance of the beef cattle industry in the region (Gerber et al., 2013). Clearly there is a gap in research conducted in non-OECD, developing countries, where smallholder farming dominates livestock production. Given the contribution of smallholder producers to global livestock production and concurrent emissions, along with the distinct method of production in such systems as compared to larger scale systems (Desjardins et al., 2012), there is scope for further research beyond the existing global-scale estimates (Opio et al., 2013).

2.5.3.2 Sheep production

In addition to the differences in the global distribution of conducted LCA and carbon footprint research, there is a clear discrepancy in the research attention directed to beef cattle production as opposed to sheep production. The number of beef cattle studies reviewed outnumbered the number of sheep studies by almost four-fold. The studies reviewed only included those with primary focus on meat production, however similar observations were made for studies investigating the impact of dairy cattle as compared to dairy sheep production (Marino et al., 2016). Of the international sheep production studies examined, many were ether national benchmarking/commodity analyses which only considered national or regional averages (Dyer et al., 2014; Ledgard et al., 2011; O'Brien et al., 2016; Opio et al., 2013; Toro-Mujica et al., 2017), or applied case study farm data but utilised Tier one IPCC methodology in the calculation of animal emissions (Edwards-Jones et al., 2009; Hyland et al., 2016; Jones et al., 2014). The Tier one approach applies a flat conversion rate to animal emissions which doesn’t consider differences between stock classes, diet or intra-annual variations of either (IPCC, 2006). All international sheep production studies have specifically focussed on lamb production, with wool considered a by-product. The co-production of both meat and wool in these systems was dealt with through the

application of allocation, primarily on an economic or biophysical basis. Allocation method can significantly alter the results of carbon footprint analyses (Cottle & Cowie, 2016; Wiedemann et al., 2015a) and represents an additional reason as to why the comparison of studies is so difficult and fraught with misinterpretation.

Globally, most sheep production carbon footprint research conducted thus far has occurred in Australia where, unlike international observations, a similar level of attention has been directed towards Australian beef cattle and sheep production (Table 2.1). This reflects the importance of the sheep meat and wool industries in the Australian agricultural sector (MLA, 2019e) compared to other countries, for example Canada where the respective contribution is small (Dyer et al., 2014). A total of nine Australian sheep LCA or carbon footprint studies were sourced. However, only three considered meat as the primary product (Peters et al., 2010; Wiedemann et al., 2015b; Wiedemann et al., 2016b) with the remaining six centred on either wool production (Brock et al., 2013; Wiedemann et al., 2016c), a comparison of different farm-produced outputs (Biswas et al., 2010) or the investigation of allocation methodologies (Cottle & Cowie, 2016; Eady et al., 2012; Wiedemann et al., 2015a) , with meat considered a by-product of the process. The first sheep meat-centric study, Peters et al. (2010), collected two years of data from a Merino wool:meat enterprise in south-western Australia in their undertaking of the first livestock LCA conducted in Australia. This study considered pre-farm to processing stages and applied mass allocation of emissions. Wiedemann et al. (2016b) conducted a cradle-to-farm gate LCA to quantify the impact of the production of lamb for export in south-eastern Australia, using the approach outlined earlier of considering regional average data and individual case study farms. It was not possible to ascertain the number or specific characteristics of the case study farms applied in this study as all inventory data and results were presented as averages only. The other sheep meat study, Wiedemann et al. (2015b), used the same case study farm data, with the addition of meat processing, international transportation and warehouse distribution, in their consideration of the additional impact of export to the USA.

Regardless of primary product, of the Australian sheep studies examined research has been overwhelmingly concentrated in south-eastern Australia. Aside from the one sheep meat case study considered in Peters et al. (2010), only three other studies considered sheep production in south-western Australia (Cottle & Cowie, 2016; Eady et al., 2012; Wiedemann et al., 2016c). Eady et al. (2012) investigated the impact of different allocation methodologies on the carbon footprint of a mixed crop and Merino sheep enterprise in an undisclosed location in south-western Australia, with focus on

wool and crop production. Similarly, Cottle and Cowie (2016) also investigated the impact of allocation with the study separated into two components, the first a detailed carbon footprint analysis of two case study farms, one of which was a mixed crop and Merino enterprise in the great southern region of WA. The second involved a large-scale analysis of GrassGro modelled southern Australian enterprises obtained from another study (Moore, 2012), however this component was not a carbon footprint as only animal emissions and some soil emissions were considered. Wiedemann et al. (2016c) considered four sheep production enterprises in the Wheatbelt region of WA as part of their benchmarking analysis of wool production in Australia. As for the other benchmarking LCAs conducted by this author, the study applied both regional average farms using survey data and case study farms. The study did not state how the information from the case study farms was collected and, as per their other studies, presented the results only as an average so it as not possible to examine the particular characteristics of each case study farm.

The review of sheep LCA and carbon footprint studies revealed that along with a lack of in-depth analyses of carbon footprints in study findings, very few examined the factors influencing the carbon footprint of the production systems considered or the influence of potential mitigation strategies. For those which did, most considered such influencing factors qualitatively without any quantitative analysis. Instead, most sheep studies focus on benchmarking sheep production in a country or region, along with hotspot analysis. Considering that enteric methane production was overwhelmingly the dominant hotspot across the reviewed studies, there would be benefit in examining the factors influencing this hotspot.

Internationally, there has been greater focus on examining the influence of different farm practices than in Australia. For example, Ripoll-Bosch et al. (2013) compared the carbon footprint of three sheep production systems of varying degrees of intensification in Spain, while Jones et al. (2014) compared the carbon footprint of three typical sheep production systems in the UK before conducting an analysis to ascertain the key farm practices which influenced the resultant footprints (i.e. ewe fecundity, concentrate use, growth rate). Toro-Mujica et al. (2017) developed a comprehensive carbon footprint calculator which enabled the calculation of a range of simulated sheep production farms along with accompanying analysis to identify the factors influencing the carbon footprints of these farms (i.e. pasture yield, reproductive rates). O'Brien et al. (2016) also examined the influence of intensification in sheep farms in Ireland, finding that differences in pasture quality influence the carbon footprint. This is also one of the few studies to consider pasture differences.

By contrast, in Australia, sheep LCA and carbon footprint studies have largely focussed on benchmarking production (Wiedemann et al., 2015b; Wiedemann et al., 2016b; Wiedemann et al., 2016c) or analysing methodological issues (Cottle & Cowie, 2016; Eady et al., 2012; Wiedemann et al., 2015a). Whilst Peters et al. (2010) made comparisons between the carbon footprints of sheep and beef case study farms, determining that factors such as faster turnoff rate and purchased stock influenced these differences, these were made on qualitative comparisons of results. Wiedemann et al. (2016c) examined the effect of lamb age but only in the context of wool and on regional average farms. No other Australian study has investigated the influence of farm practices or mitigation. In addition to this, most studies do not breakdown the carbon footprint into individual emission sources, rather only in the context of a production stage such as on-farm or processing (Peters et al., 2010; Wiedemann et al., 2015b), an emissions type such as methane or nitrous oxide (Wiedemann et al., 2016b; Wiedemann et al., 2016c), or as a co-product total (Eady et al., 2012). This makes it difficult to conduct any further interpretation of such studies or make comparisons. The only studies which presented such results in detail were Biswas et al. (2010) and Brock et al. (2013), both of which presented emissions across all sources. However, Biswas et al. (2010) did not consider emissions from the breeding herd and applied a flat enteric methane conversion factor in line with IPCC Tier one methodology, while Brock et al. (2013) applied GrassGro output to calculate the carbon footprint, representing the most comprehensive analysis and presentation of results of all the Australian sheep studies. These differences further hinder comparison and highlight the variation in approach across studies. No studies have examined emissions across livestock classes or intra-annual variations.

Globally, sheep play a vital role economically, culturally and in terms of food security, alongside their contribution to GHG emissions. Despite this, it is clear through the review of existing sheep meat production studies, that this is a field requiring further research attention and has been highlighted by multiple studies (Jones et al., 2014; Marino et al., 2016; Zervas & Tsiplakou, 2012). In Australia, though the impact of wool production has received some attention, far less has been directed towards examining the impact of sheep meat production. There is also a clear gap in research conducted in south-western Australia. Most Australian sheep LCA and carbon footprint research thus far have focussed on benchmarking analyses of various sectors within the sheep industry or analyses of methodological approaches. Whilst these are invaluable bodies of research, if progress is to me made towards the reduction of emissions in line with national and international targets, going forward focus must be instead on

identifying and examining the effectiveness of farm strategies with mitigation potential. For this to be effective, it needs to be conducted at farm- or local-scale, not at regional or national levels which mask the diversity of farming systems in Australia.

2.5.3.3 Beef production

The review of beef production LCA and carbon footprint studies highlighted the diversity which exists between production systems. Despite the inability to directly compare results, the observed heterogeneity in impacts between the studies can at least partially be attributed to the range of production systems exhibited. For example, European and North American beef production typically occurs in more intensive systems than Australian or South American systems. This reflects the fact that European and North American cattle must be housed indoors for at least a portion of the year in these regions and are typically finished in feedlots for considerable periods prior to sale (Beauchemin et al., 2010; Mogensen et al., 2015; Pelletier et al., 2010). Such systems are input-, labour- and infrastructure-intensive (Desjardins et al., 2012). By contrast cattle in Australia and southern America are predominantly produced on pasture in extensive grazing systems which require low levels of management (Becoña et al., 2014; Dick et al., 2015a; Ridoutt et al., 2011; Wiedemann et al., 2015c). These exhibited differences are reflected in the distribution of emissions produced by the respective production systems. For example, manure emissions are typically a key emission source in more intensive systems, reflective of the requirement to maintain a liquid manure management system (Crosson et al., 2011). These systems also typically exhibit a higher contribution of emissions arising from the production of inputs, reflective of the requirement for large quantities of externally sourced feed (Ogino et al., 2004; Ogino et al., 2007). By contrast, these emission sources are typically negligible in extensive production systems, with enteric methane overwhelmingly the dominant hotspot in the carbon footprint of such systems (Desjardins et al., 2012).

Internationally, the review of beef cattle studies revealed that investigation of strategies influencing environmental impact, along with analysis of potential mitigation strategies, has been a primary driver of studies conducted to date. This is evident in even the early beef cattle studies, where focus was on investigating the influence of selected strategies such as finishing period, feedlot ration, turnoff periods and fecundity (Beauchemin et al., 2011; Casey & Holden, 2006; Lieffering et al., 2011; Ogino et al., 2004; Ogino et al., 2007; Stewart et al., 2009). Others conducted comparative analyses which examined the impact of different production methods on the carbon footprint of beef cattle systems (Nguyen et al., 2010; Pelletier et al., 2010;

Veysset et al., 2010). Though some of these early studies employed methodological approaches now not recommended for mitigation analysis, such as the use of regional or average data, Tier one IPCC methodology, or narrower system boundaries, such as the exclusion of breeding herd emissions, they were important foundational studies for beef cattle carbon footprint analysis. More recent beef cattle studies have built on the level of analyses conducted in these earlier studies, for example through the examination of farm practices influencing the carbon footprint across a suite of representative case study farms (Bragaglio et al., 2018; Nieto et al., 2018) or examining the effect of temporal changes (Hyland et al., 2016). A recent carbon footprint analysis examined the influence of intra-farm emissions distributions through an approach which considered the heterogeneity of animal performance on an individual animal scale and considering intra-annual variations (McAuliffe et al., 2018). This represents a deviation from other studies which typically aggregate results through the production year or across the herd and aligns with the callings for detailed farm-level approaches in examining carbon footprint and mitigation strategies going forward (Rawnsley et al., 2016).

In Australian LCA and carbon footprint studies by contrast, little attention has been directed towards investigating the mitigation potential of strategies in beef production systems. Instead, most research conducted to date has focussed on the benchmarking of beef production or examination for the purpose of hotspot analysis only. Of the seven Australian beef cattle LCA and carbon footprint studies reviewed, the primary goals of three were the benchmarking of broadacre beef production and feedlot beef production in eastern Australia (Ridoutt et al., 2011; Wiedemann et al., 2016a; Wiedemann et al., 2015c), while another conducted a comparison of the environmental performance of the national beef industry between 1981 and 2010 (Wiedemann et al., 2015d). The remaining three studies employed detailed case study records to quantify and compare the impact of specific beef production systems. For example, Peters et al. (2010) investigated and compared the environmental impact of a premium export beef supply chain and an organic beef supply chain, both located in eastern Australia. Similarly, Eady et al. (2011) quantified and compared the carbon and water footprints of two broadacre beef enterprises in north-eastern Australia, differing in their primary product, weaners and finished steers, respectively. Taylor and Eckard (2016) examined the carbon footprint of three beef herds in north-eastern Australia. Though each study had conducted analyses utilising detailed case study data, the level of interpretation afforded by each differed. Whilst each of these studies identified either stages (i.e. feedlot; Peters et al. (2010)), emission sources

(i.e. breeding herd; Eady et al. (2012)) or specific management practices (i.e. grazing management; Taylor and Eckard (2016)) influencing the carbon footprint, through the conducted comparative hotspot analyses, few presented findings beyond this. Taylor and Eckard (2016) was the only study to conduct an additional layer of analysis, by modelling the potential impact of alternate herd management strategies on the carbon footprint of the examined case study farms. Whilst Eady et al. (2012) conducted a preliminary analysis of the potential influence of carbon sequestration from tree plantings, along with Peters et al (2010), any discussion of potential abatement strategies was restricted to qualitative discussions.

The review of beef cattle LCA and carbon footprint studies highlighted that while more research attention has been directed towards the identification and investigation of strategies with mitigation potential than for sheep production, it has thus far been more concentrated in the more intensive systems of Europe and northern America. Whilst studies centred on extensive beef production in southern America have prioritised quantitative investigations into improving the carbon efficiency of such systems, in Australian broadacre systems more focus has instead been directed toward the benchmarking and primary hotspot analysis of beef production the south-eastern and north-eastern regions of the country. Furthermore, the inconsistency in the presentation of assumptions, data sources and results analysis between studies makes it difficult to draw further interpretation beyond the acknowledged findings of the study and extrapolate this to comparisons with other conducted or proposed studies. As for Australian sheep production, there is clearly a lack of LCA or carbon footprint research dedicated to the quantitative analysis of potential mitigation strategies applicable to beef production. In addition, to date no beef cattle studies have been conducted in south-western Australia. Given both the diversity of beef cattle production systems across Australia and the drive for carbon neutral beef, these are critical knowledge gaps.

2.5.3.4 Consideration of pasture systems

As one of the primary considerations of the present study was the investigation of the potential influence of pasture, specifically annual versus perennial pasture systems, the review of existing LCA and carbon footprint studies set out to ascertain the level of prior research attention that has been directed towards such investigations.

Internationally, consideration of pasture has primarily focussed on the comparison of grass- and concentrate-based production methods (Casey & Holden, 2006; Pelletier et al., 2010; Foley et al., 2011; Capper et al., 2012; Ogino et al., 2016; Bragaglio et

al., 2018). In line with the constraints of the IPCC Tier two methodologies as described in Section 2.5.2, it follows that perhaps analyses were restricted to examination of only pasture versus concentrate because methane conversion factors are only available for each, preventing further breakdowns within pasture types. More recently, the depth of analyses has expanded to the comparison of degraded, unmanaged and improved pastures, primarily in southern American extensive systems. The recent focus on different pasture systems in the region likely reflects the strong economic contribution of the livestock sectors, particularly beef, driving the advancement of methodologies to enable such analyses, either through specific methane conversion factors (Modernel et al., 2013; Ruviaro et al., 2015; Nieto et al., 2018) or the development of empirical, stochastic whole-farm models (Toro-Mujica et al.,2017). One study (de Figueiredo et al., 2017) however, employed flat enteric methane conversion rates, despite a research goal of comparing different pasture systems, while another (Dick et al., 2015) did not identify the specific methane conversion factors adopted, highlighting the importance of considering adopted methodology in the interpretation of results. In addition to these southern American studies, a recent UK study examined the intra-annual variations of pasture quality through the integration of fortnightly measurements into carbon footprint analyses, the only international study examined to make such intra-annual and comprehensive calculations (McAuliffe et al., 2018). No international study examined however, has assessed perennial versus perennial pasture systems in the context of an LCA or carbon footprint analysis.

In Australia, no LCA or carbon footprint study conducted to-date has considered the potential influence of pasture, whether it be in the context of overall attributes or as a comparison between annual and perennial systems. Enteric methane production, the primary hotspot of all the reviewed studies, is determined by feed intake, which in turn is a function of livestock physical and physiological characteristics along with the attributes of the pasture or feed consumed by that animal (DISER, 2020b). Whilst most of the studies provided detailed explanations regarding the assumptions and data sources behind livestock characteristics, in few instances was such information provided for the feed attributes applied in the emission calculations. In fact, of all the Australian studies considered, only Brock et al. (2013) considered detailed pasture data specific to the case study farm, applying farm-specific daily time-step GrassGro modelled annual pasture data. They were also the only study to present details of the pasture attributes applied. Most other studies have applied seasonal state average pasture data obtained from the NIR (DISER, 2020b). This data does not distinguish

between perennial or annual pasture, the influence of grazing management on pasture, or the diversity in pasture attributes evident across farms or even regions (DISER, 2020b).

Whilst the adoption of seasonal state average pasture data may be deemed suitable for studies which are conducting benchmarking analyses using state averages for other calculation data, the appropriateness when conducting analyses using farm-specific or regional averages for other data sources must be questioned given the potential impact on final hotspot results. For example, the NIR states that, for beef cattle, pasture dry matter digestibility (DMD) and crude protein (CP) content in summer and autumn in southern WA is between 58-50% and 7-10%, respectively. By contrast, validated modelled mixed annual: perennial grass pasture in the same region for the same period yielded DMD of between 67-68% and CP content of 13-14% (Thomas et al., 2012). Similarly, field observations throughout southern WA found that the average DMD and CP of various perennial grass species during the non-growing season ranged from 63-68% and 12-15%, respectively, whilst the attributed for annual species ranged from 44-52% and 5-9%, respectively (Moore et al., 2009b). This diversity is not captured in the adoption of average data. In fact, Brock et al. (2013) examined the influence of different calculation methods and data sources on calculated enteric methane output, finding that the NIR averages underestimated methane output by 16% as opposed to when farm-specific seasonal pasture data was applied. In both instances only annual pasture was considered. It follows that the magnitude of such differences could be greater if the farm-specific data considered perennial pasture, which can make use of out-of-season rainfall to continue to provide quality feed through the dry season (Descheemaeker et al., 2014; Monjardino et al., 2014). Clearly there is a gap in current LCA and carbon footprint research regarding the consideration of pasture data and the consideration of pasture system differences. The push for farm-level analyses and examination of mitigation strategies should be accompanied by a concurrent improvement in the accuracy of pasture and feed data, the primary influences of animal emission hotspots

There are, however, a number of Australian whole-farm system studies, which have considered the influence of pasture on emissions. Whilst not considered LCAs or carbon footprints due to the exclusion of certain emission sources, they predominantly conducted these analyses through biophysical modelling. These studies examined the impact of improved pasture quality and grazing management through the introduction of legumes (Cottle et al., 2016), comparisons of different pasture swards (Harrison et al., 2014a), the introduction of pasture species with enteric methane-

inhibiting properties (Doran-Browne et al., 2015), consideration of perennials grasses (Bell et al., 2012a; Thomas et al., 2012) and the consideration of perennial shrubs (Harrison et al., 2016; Harrison et al., 2015; Taylor et al., 2016). Whilst these studies did not follow the LCA principles adopted in the present study, they were critical in informing the methodological approach of the study in the context of pasture considerations.

2.6 Mitigating emissions in livestock production systems

The previous Sections outlined that widespread mitigation through the southern Australian livestock sector is required if emissions reduction targets are to be met. However, multiple barriers to the uptake of mitigation have hampered such progress (Herrero et al., 2015; Kragt et al., 2017), including;

- Economic or productivity implications
- High costs of implementation
- Accessibility by farmers
- Farmer uncertainty regarding potential benefits; and
- Lack of policy incentives and low returns from carbon prices.

Reducing the carbon footprint of a farming enterprise is in most cases considered secondary to the primary motivators of productivity and profitability. As such, the most promising strategies for adoption by livestock producers are those which also improve productivity (Hyland et al., 2016; Nieto et al., 2018). As productivity is typically examined at a whole-farm scale, it follows that the most appropriate method for assessing a strategy for mitigation is also at farm-scale, enabling the capture of all interactions between farm components (Rawnsley et al., 2016). In addition to this, just as livestock production systems are incredibly diverse in their individual characteristics and employed management practices, there will be no one size that fits all when it comes to mitigation. The actual impact of strategies will be farm-specific and the best approaches are those which consider this through locally and regionally appropriate analyses based on local research.

2.6.1 Opportunities for mitigation

There is a plethora of studies which have reviewed in detail the mitigation strategies available for ruminant production systems (Beauchemin et al., 2020; Eckard et al., 2010; Eckard & Clark, 2020b; Henry et al., 2012; Herrero et al., 2016; Hristov et al., 2013a; Hristov et al., 2013b; Leahy et al., 2019). It is not the purpose of this Section to repeat the findings of these reviews, specific strategies are examined in later

Chapters, but rather to highlight promising mitigation pathways in the context of what has been outlined above.

Investigation into the most appropriate mitigation strategies in livestock production has primarily centred in the reduction of methane and nitrous oxide emissions, particularly those arising from livestock. In the context of broadacre livestock systems, such strategies typically fall into two key categories; animal management and grazing management. In most cases, these strategies act to reduce emissions by one of two paths, a direct reduction of emissions or an indirect reduction of emissions through improved productivity.

The first pathway, the direct reduction of emissions, typically occurs through practices such dietary supplementation, rumen modifiers and breeding for low-emitting livestock (Eckard & Clark, 2020b). Whilst such technologies have demonstrated potential in the reduction of emissions, many are typically suited to intensive systems, are not currently accessible to producers for economic or feasibility reasons and require further research investment before their promotion for widespread adoption (Zervas et al., 2012; Beauchemin et al., 2020). To-date, research into these strategies has been concentrated at a component level, however some analyses have been conducted at whole-farm scales, with mixed results. For example, Harrison et al (2016) examined the impact of nitrate supplementation on broadacre beef production in northern Australia, finding that whilst EI reduced by 4%, farm productivity remained unchanged while farm gross margins increased by 37%. Similarly, while dietary supplements can reduce animal emissions, on a whole-farm scale they do not necessary reduce EI as increased pre-farm emissions, along with potential negative implications of the supplement on production, may offset such reductions (Williams et al., 2014). While genetic improvements such as traits for fecundity, methane yield and feed conversion efficiency demonstrate potential in reducing modelled EIs, many are not yet commercially available in Australia and interactions with other animal characteristics are yet unknown (Alcock & Hegarty, 2011a; Alcock et al., 2015; Harrison et al., 2014b). For example, the effect of breeding for low methane emitting livestock has unknown impacts on other parameters such as meat quality. While technologies which operate through a direct reduction of emissions are promising on a component level, their performance at whole-farm scales are mixed, and at present they often do not provide the co-economic and productivity benefits that drive on-farm adoption.

The second mitigation pathway in livestock systems, the indirect reduction of emissions through improvements in the efficiency with which liveweight is produced,

often provides concurrent economic and productivity benefits for the system .(Herrero et al., 2015). Because of these co-benefits, they are more likely to align with farmer motivators and thus have increased potential for uptake by producers in the shorter-term. As such these mitigation strategies, typically occurring through increased reproductive or feed conversion efficiencies, have been the focus of more LCA and carbon footprint analyses than strategies operating through the first pathway.

Mitigation through improved reproductive efficiencies occurs through a shift in emissions away from breeding livestock to livestock produced for sale. In doing so, it increases the quantity of emissions that are offset by saleable liveweight production. Strategies to increase reproductive efficiencies have been found to reduce the carbon footprint of livestock production internationally. Such strategies include increased fecundity, increased weaning rates, breeding animal longevity and reducing joining ages (Ledgard et al., 2010; Beauchemin et al., 2011; Lieffering et al., 2011; Becoňa et al., 2014; Jones et al., 2014; Dick et al., 2015; Toro-Mujica et al.; 2017; Dougherty et al., 2018; Nieto et al., 2018). In Australia, while no carbon footprint or LCA studies have explicitly examined the impact of such strategies, it has been the focus of numerous whole-farm studies typically through the biophysical and economic modelling of simulated production systems (Alcock & Hegarty, 2011a; Alcock et al., 2015; Browne et al., 2015; Browne et al., 2011; Cullen et al., 2016; Harrison et al., 2014a; Harrison et al., 2016; Harrison et al., 2014b). These strategies are explored further in later Chapters.

Mitigation through improved feed conversion typically operates through higher growth rates and turnoff rates resulting from improved feed quality and grazing management. Improvements in pasture quality and grazing management were the focus of Section 2.5.3.4. Intensification through increased concentrates in diets and increased feedlot finishing periods are another recommended mitigation strategy through improved diet quality. This strategy has received the most attention by international LCA and carbon footprint studies (Nguyen et al., 2010; Beauchemin et al., 2011; Foley et al., 2011; Pelletier et al., 2011; Ogino et al., 2016; Bragaglio et al., 2018) and has been found to be effective in the reduction of the EI of livestock production. Whilst Australia has observed an increase in commercial feedlot finishing (Wiedemann et al., 2016a), unlike international systems where livestock production is typically more intensive, increased commercial grain-finishing may have limited potential in Australia where there is a greater dependence on broadacre livestock production. In addition to this, consideration must be directed to other impacts associated with such intensification, such as increased arable land requirements for feed crop production (Wiedemann et

al., 2016a), increased emissions from the production and transportation of the additional grain (O'Brien et al., 2016), and that the increased allocation of grain to livestock production can divert supply from human consumption as opposed to livestock on pasture which has no feed value to humans (Desjardins et al., 2012). Despite this, there may be opportunities for smaller scale, on-farm targeted feedlot finishing of Australian livestock in conjunction with improved herd/flock productivity (Alcock & Hegarty, 2011a; Cottle et al., 2016; Harrison et al., 2014a).

A third, less explored option for mitigation, is the offset potential of increased soil carbon through sequestration and improved farm management. Agricultural soils are important stores for carbon (Soussana et al., 2010) and this has been promoted as a promising option to mitigate emissions (Lal, 2004; Lal et al., 2007). In Australia however, the results of the sequestration potential of pasture are mixed (Chan et al., 2010; Lam et al., 2013; Meyer et al., 2016; Sanderman et al., 2013; Wilson et al., 2011) and inclusion in whole-farm analyses limited to-date because of the high uncertainties associated with methodological approaches, understanding of the long-term persistence of sequestration, and regionally-relevant data availability. Driven by its inclusion as an ERF-recognised abatement strategy, more attention has been directed towards investigating whole-farm abatement potential of tree planting. Whilst reforestation and afforestation can offset whole-farm emissions, at current carbon prices the strategy is economically unfeasible for producers (Doran-Browne et al., 2017; Doran-Browne et al., 2016; Mayberry et al., 2019). Whole-farm studies such as these and others (Henry et al., 2015a; Henry et al., 2015b), recommend caution when calculating the impact of and recommending such sequestration strategies and clearly further research and policy investment needs to be directed towards improving its potential as a mitigation option.

Reducing the carbon footprint of livestock enterprises by improving the efficiency of liveweight production is a promising option for mitigation. In the context of on-farm adoption, such strategies can overcome the prevailing barriers to on-farm adoption as they can be accompanied by concurrent economic and productivity benefits, are accessible by producers in the short-term, and importantly, have demonstrated uptake and success by other producers. International studies which have examined the factors influencing the carbon footprint of case study farms found in all cases, farms with lower footprints employed either improved reproductive, feed conversion or grazing management strategies, or a combination of the three (Becoňa et al., 2014; Jones et al., 2014; Veysset et al., 2014; Toro-Mujica et al., 2017; Nieto et al., 2018). These studies also highlight the opportunities for mitigation through the adoption of

strategies with productivity benefits. In fact, it is estimated that widespread adoption of such strategies could reduce emissions from the global livestock sector by 30% (Gerber et al., 2013). Whilst many whole-farm studies examine mitigation potential on modelled farming systems and present options for adoption based on these simulations, the exhibited variation in farm characteristics, practices and resultant carbon footprints across case study farms emphasise the importance of conducting analyses which consider that carbon footprints and thus the effect of mitigation strategies will be farm-specific.

2.7 Conclusion

This Chapter explored the challenges facing the international and southern Australian livestock industries. Uptake of mitigation strategies by producers has been slow, hampered by inconsistent and ineffective policies, a lack of cost-effective strategies for uptake and uncertainty regarding such strategies. Overcoming these barriers is essential if emissions reduction targets are to be met and farmers progress towards enhanced resilience in the face of climate change and other risk factors.

There is opportunity for the adoption of strategies with both productivity and mitigation benefits, however the effect of such strategies will be farm-specific. Farm-specific, whole-farm system carbon footprint analyses are promising options for the examination of the carbon footprints livestock enterprises and targeted identification of strategies with mitigation potential. Despite this, while considerable effort in Australia has been directed toward benchmarking the carbon footprint of livestock and the investigation of associated methodological approaches, there remains a distinct gap in research conducted in south-western Australia, in farm-scale carbon footprints and in the investigation of practices which influence these footprints, such as animal and grazing management. Furthermore, there is a need to develop approaches which conduct in-depth, intra-farm carbon footprint analyses of livestock production systems to enable such farm-specific and targeted assessments.

3 METHODOLOGY

3.1 Introduction

The previous Chapter highlighted multiple gaps in existing livestock carbon footprint research, including a lack of intra-farm analyses, consideration of the influence of different pasture systems and widespread investigation into strategies with mitigation potential. It also highlighted the lack of farm-level carbon footprint research pertinent to livestock production in south-western Australian. From a methodological perspective the review demonstrated a need for research that integrates farm-specific data with detailed carbon footprint analyses to consider intra-farm scale analyses and application of potential mitigation strategies.

This Chapter outlines the methodological approach developed to address these gaps and achieve the goal of the present study. There are seven identified stages (Figure 3.1), each addressing one or more of the research objectives of this study.

Objective one - Develop a comprehensive tool that allows the calculation of the carbon footprint of sheep and beef cattle enterprises and examination of ensuing mitigation strategies.

And;

Objective two - Quantify the carbon footprint of livestock production systems in south-western Australia, with focus on perennial versus annual pasture systems.

Stages one to four: Following LCA guidelines, the study goal and scope were established to inform the methodological approach. In particular, the development of two carbon footprint calculators, tailored to sheep and beef cattle respectively. The calculators are referred to as “Frameworks” for the remainder of this thesis. These Frameworks were developed to address the specific objectives of this study. For example, they enabled the integration of livestock, pasture, crop and feedlot subsystems, permitted monthly calculations and the capture of the characteristics of farm-specific livestock classes. This meant that the biophysical model GrassGro could be employed to give a more tailored estimate of emissions from specific locations. Importantly, it also enabled a detailed examination of practices influencing the carbon footprint of a livestock production system and the mitigation potential of other practices. See Sections 3.3 and 3.4 for further detail.

Objective three - Examine the mitigation potential of identified strategies on the carbon footprint of livestock production systems and provide regionally appropriate recommendations for application.

Stages five and six: Using the results obtained in the previous stages of this study, the potential of selected practices to mitigate the carbon footprint of the livestock farming systems was examined. Examined practices were selected based on a number of criteria, including productivity, profitability and ease of adoption on farm.

Following these analyses, stage seven involved the provision of recommended strategies for potential application in livestock production systems in south-western Australia. Section 3.5 outlines this in greater detail.

3.2 An integrated approach to whole-farm carbon footprint analysis

In the review conducted in Chapter Two, the necessity for an integrated approach when conducting whole-farm system analyses was highlighted. Furthermore, it unified the call for farm-level analyses when examining potential mitigation strategies. At the inception of this research, it became obvious that the only way to address the study objectives was to adopt such an approach. This Section briefly outlines the rationale behind the integrated approach adopted in this study.

The three essential components of the study objectives were the;

- analysis of livestock production systems in south-western Australia
- differentiation between the impact of annual and perennial pasture systems on the carbon footprint of livestock production; and
- examination of the impact of potential mitigation studies.

Though the study always intended to follow an LCA approach, it became clear that to successfully complete the above components, a dedicated carbon footprint calculator would have to be developed. This was because existing Australian calculators and biophysical models did not permit one or more of the following, all of which were deemed necessary to complete the study objectives.

- Monthly calculations
- Examination of multiple feedbases (i.e. pasture, crop stubble, feedlot)
- Calculation of individual dietary components (i.e. pasture vs supplement)
- Differentiation between physiological status of a stock class (i.e. growing vs maintenance, dry vs lactating)
- Integration of livestock, pasture, crop and feedlot systems in a farming system

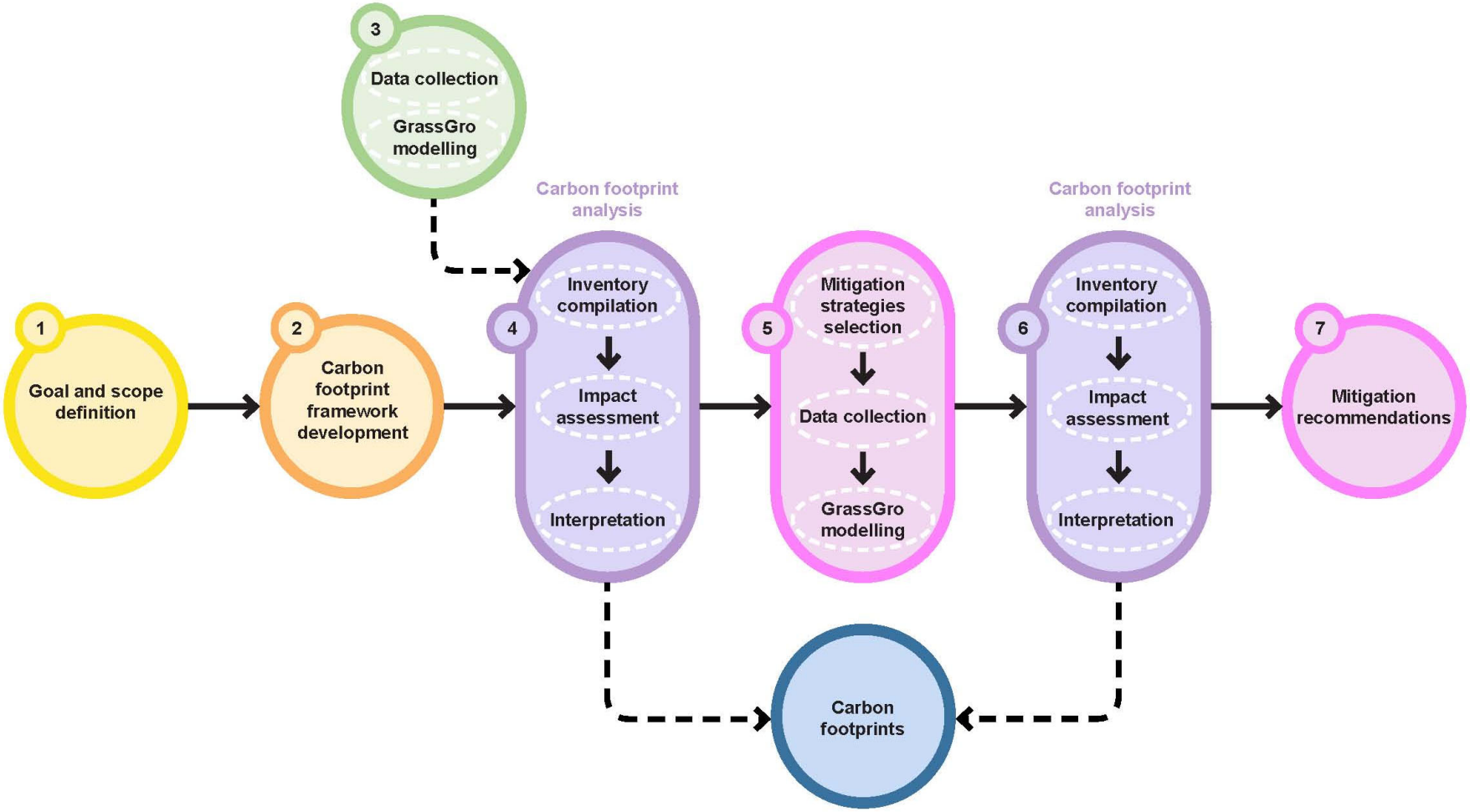


Figure 3.1 - Schematic representation of the methodological approach adopted in this study. .

- Calculation of pre-farm emissions (i.e. input production and transportation)
- Adoption of the 2018 Australian National Inventory Report (NIR) methodology (DISER, 2020b) for calculating emissions in accordance with IPCC guidelines (IPCC, 2006)

The requirement for monthly calculations can be applied as an example to demonstrate the importance of the above factors. Seasonal or annual calculations are insufficient where the relative performance of pastures is required. Annual growth patterns and nutritive attributes vary considerably over a production year and between pasture species. Such differences will have flow-on effects to other components of the farming system, including livestock productivity and emissions output. Annual and seasonal calculations mask these differences. For example, pasture growing seasons do not always align with the traditionally defined starting or end months of a season. Growing seasons differ across regions or pastures and so conducting seasonal calculations which do not align with the respective growing months can skew results. Monthly calculations also allow the impact of farm practices to be examined, such as time of lambing. As such, in this study monthly calculations were required to examine differences between pasture systems, the employed management practices and the effect of potential mitigation strategies.

Existing Australian GHG calculators calculate emissions on seasonal or annual time-steps with limited flexibility with regards to the differentiation between feedbases and stock classes. For example, the considered calculators did not permit the examination of more than one feedbase (i.e. pasture, crop stubble) on an enterprise simultaneously and or the calculation of multiple dietary components (i.e. pasture vs supplementary feed). This made it difficult to make comparisons between different pastures or to model the effects of rotational grazing across feedbases. The inability of these calculators to quantify supplementary feed requirements, a performance indicator of a feedbase, prevented the calculation of revised animal emissions where supplementary feed was provided alongside pasture, for example. It also made it difficult to examine the effect of implementing of a potential mitigation strategy and the calculation of emissions associated with the production and transportation of supplementary feed.

While biophysical models such as GrassGro do enable monthly calculations, they are limited in terms of the feedbases they can model. They also calculate animal emissions only, omitting other emission sources such as soil and pre-farm, and thus cannot undertake a full carbon footprint analysis. Animal emissions are also calculated using different approaches to those recommended by the 2018 NIR.

Given the above considerations, it was decided to develop whole-farm carbon footprint calculators for sheep and beef cattle. Whilst GrassGro was unsuitable for the purpose of conducting the carbon footprint analyses, its ability to capture the complex interactions between climate, soil, pasture and livestock, meant that it could provide farm-specific modelled monthly pasture data which could be integrated into the calculators. This integrated approach permitted detailed carbon footprint analyses of livestock production systems, considering multiple pasture systems and the examination of potential mitigation strategies.

3.3 Development of a whole-farm carbon footprint framework

This Section outlines the steps undertaken to develop the sheep and beef cattle carbon footprint Frameworks adopted in this study.

To conduct the carbon footprint analyses in this study, LCA methodology was followed. While a carbon footprint examines one environmental impact category, GHG emissions, LCAs consider multiple impact categories. Despite this difference, the approach is fundamentally the same, guided by sets of core standards (ISO, 2006a, 2006b, 2018). These standards separate the approach into four phases. Each phase is reflected in the methodology outlined in the following Sections of this Chapter.

1. Goal and scope definition (Sections 3.3.1-3.3.2)
2. Inventory analysis (Sections 3.3.3-3.3.4)
3. Life cycle impact assessment (Sections 3.3.5-3.3.6)
4. Interpretation (Section 3.3.7)

Following this, Sections 3.4-3.5 of this Chapter outline the additional tasks conducted to enable the successful completion of the above four phases.

3.3.1 Goal

The goal of the carbon footprint analyses conducted in this study was to investigate the whole-farm carbon footprint of livestock production systems in south-western Australia and to identify strategies which have potential to reduce the footprint of such systems. Specifically to;

- Develop a comprehensive tool which allows the calculation of the GHG emissions produced by a sheep or beef cattle enterprise and the analysis of potential mitigation strategies
- Quantify the carbon footprint of livestock enterprises located in major farming regions in south-western Australia, including an analysis of the impact of different pasture systems; and

- Investigate the mitigation potential of selected strategies as applied to the considered enterprises.

The functional unit (FU) adopted in this study was kilograms of saleable liveweight at the farm gate. The results of the carbon footprint analyses were presented as carbon dioxide equivalent per kilogram of liveweight produced for sale (kg CO_2-e/kg LW).

Following the approach of other carbon footprint studies, as described in Chapter Two, this FU is also referred to as “emissions intensity” (EI) and serves as a measure of comparison between and within the examined livestock production systems.

To address the goal and objectives of this study, first and foremost the standards outlined in ISO 14044 and ISO 14067 were adhered to. As a good practice guide for the carbon footprint calculation of livestock systems, LEAP (LEAP, 2015b, 2016) was also followed where appropriate and has been cited accordingly. Where decisions were also guided by other sources, these are specified separately.

3.3.2 Scope

This study conducted “cradle-to-gate” carbon footprint analyses of sheep or beef cattle production. As a carbon footprint analysis, one environmental impact category, “GHG emissions”, was considered.

Total GHG emissions produced on a livestock production system over a period of 12 months, a calendar year, were calculated using multi-year averaged data obtained directly from the farm or from secondary sources such as GrassGro modelled output. Each carbon footprint considered the GHG emissions associated with the production saleable liveweight, from resource extraction to the farm-gate.

Some of the livestock production systems examined in the present study produced co-products alongside the production of liveweight for meat (i.e. mixed crop-livestock enterprises, wool production). It was necessary to allocate the calculated whole-farm emissions between these co-products before examining the carbon footprint of saleable liveweight production. Section 3.3.6 details the specific approach adopted.

3.3.2.1 System boundaries

The system boundary of this study included the inputs and associated emissions resulting from all upstream and on-farm processes and required inputs associated with the production of liveweight sold at the farm gate (Figure 3.2). These can be separated into pre-farm and on-farm processes.

3.3.2.1.1 Pre-farm processes

The pre-farm processes considered in this study include the production and transportation of all externally sourced inputs used on-farm including;

- lime and chemicals such as pesticides, herbicides and fertilisers applied to pasture and feed crops
- veterinary products such as drenches and vaccinations
- on-farm machinery such as tractors and spraying implements
- fuel used during on-farm operations
- seed or seedlings used to sow pasture or supplementary feed crops; and
- purchased supplementary feed such as molasses, mineral licks and grain.

Pre-farm emissions arose primarily from the manufacture of these inputs and their transportation to the farm. The methodologies adopted to calculate these emissions are detailed in Section 3.3.4.5. Notable pre-farm assumptions applied in this study are outlined below.

- Where income crops, that is crops produced for off-farm sale, were also produced in an enterprise, the emissions associated with the production and transportation of inputs specific to these crops were excluded.
- In the case of the sheep enterprises, emissions resulting from the production and transportation of inputs used specifically for wool production were excluded.
- Following other studies, the packaging of inputs was excluded in line with its to its minor contribution (Brock et al., 2013).
- Emissions associated with on-farm buildings were excluded, following the approach of other studies and acknowledging the difficulty in quantifying such inputs (Foley et al., 2011; Jones et al., 2014; O'Brien et al., 2016; Ridoutt et al., 2011; Wiedemann et al., 2016b).
- Where emission factors (EFs) for inputs were unavailable, EFs from comparable products were applied. The contribution to the overall farm emissions of livestock production systems by these inputs is typically minor and as such the substitution of surrogate data from similar products was determined to have a minor influence on the final results.

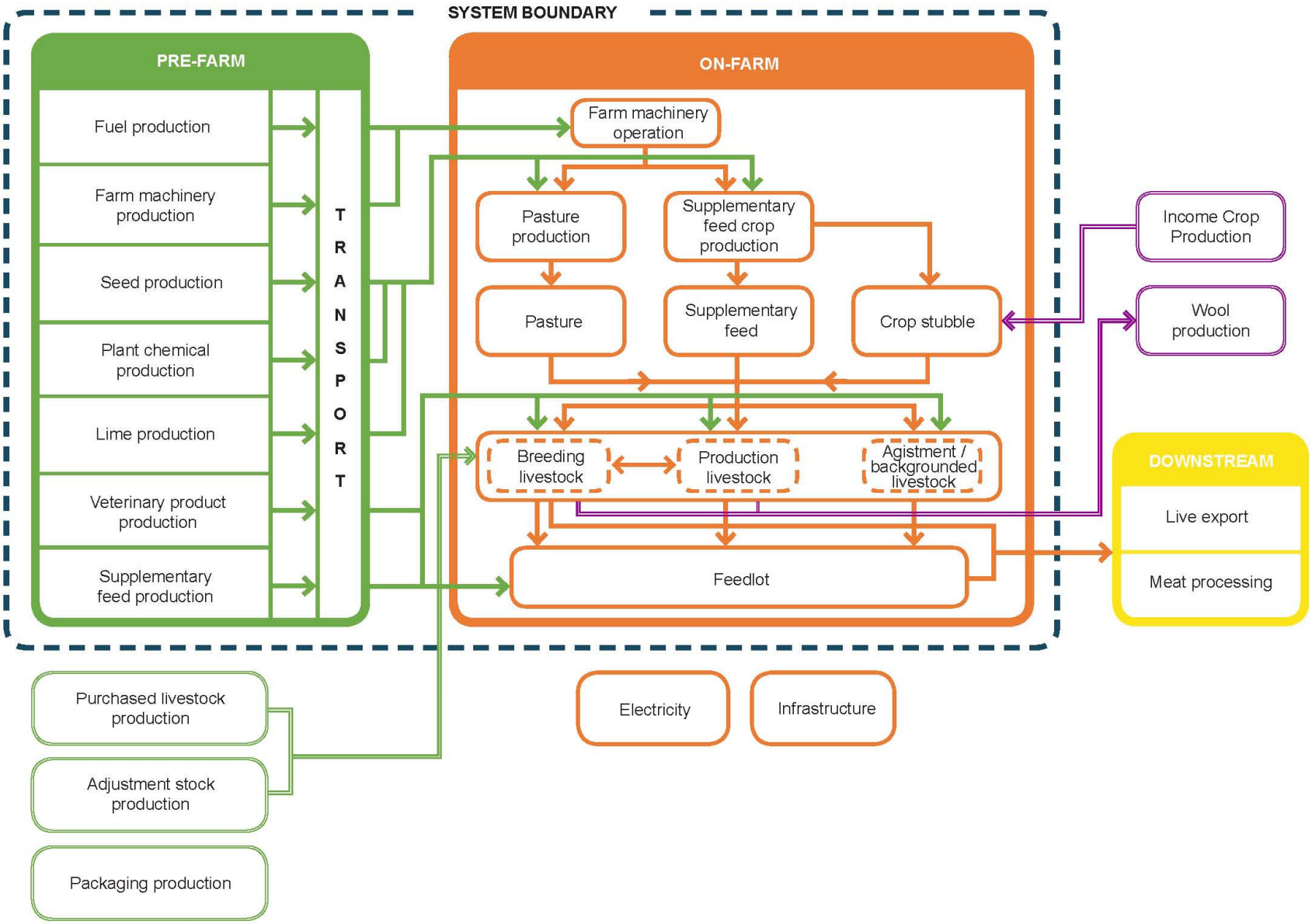


Figure 3.2 - Cradle to farm-gate system boundary for the sheep and beef production systems

3.3.2.1.2 On-farm processes

The on-farm processes considered within the system boundaries included;

- pasture and supplementary feed crop production; and
- livestock production, including the breeding flock/herd, purchased stock; and agisted/backgrounded stock on pasture, crop stubble and within the feedlot.

Multiple emissions were produced during these on-farm processes, including;

- carbon dioxide emissions produced through the on-farm combustion of fuel during on-farm machinery operation and following the application of either lime, dolomite, or urea fertiliser to soil
- methane emissions were produced directly through enteric fermentation and indirectly from animal manure
- direct nitrous oxide emissions from soil following the application of nitrogen (N) fertilisers, animal excreta and the decomposition of crop and pasture residues; and
- indirect nitrous oxide emissions from the leaching and runoff of N from soil and through atmospheric deposition following the volatilisation of N applied to soil.

Notable on-farm assumptions applied in the present study are outlined below.

- Emissions associated with the production of income crops were excluded, including all emissions arising from machinery operation and chemical application, along with soil emissions.
- Emissions from the production of crops purpose-grown to provide on-farm supplementary feed were included. This was because the product produced (i.e. grain and hay) could be attributed entirely to livestock, the focal product of the carbon footprint analysis.
- Where livestock grazed crop stubble, the animal emissions produced during the period grazed, such as enteric methane and nitrous oxide from excreta, were considered within the system boundaries of the study.
- The inputs and processes associated with the production of purchased or agistment livestock were only considered once the stock entered the farm. Inputs and emissions from the production of the stock prior to this were excluded in acknowledgement of the difficulties encountered when collating such detailed upstream data. Most studies do not explicitly state how or whether they have considered the pre-farm emissions associated with purchased or brought in livestock (Hyland et al., 2016; Wiedemann et al., 2016b; Dougherty et al., 2018; Nieto et al., 2018; Bragaglio et al., 2018, for

example). Only three of the studies reviewed for this thesis acknowledged this emission contribution. The first, an Australian red meat LCA study, made the assumption to exclude this emission source, citing the same reasons as this present project (Peters et al., 2010). The second, an Australian benchmarking study (Wiedemann et al., 2016a), included the pre-farm contribution of Queensland cattle to feedlot finishing, however they drew these values from a previous benchmarking paper published by the same author which quantified the farm stage impacts of cattle production in the same region. (Eady et al., 2011) also considered the pre-farm contribution of purchased cattle, with the quantified impact cited to be sourced from an LCI generated by the author's research organisation. No further detail was provided to enable further context regarding the specifics of this obtained impact. In practice, the quantification of the pre-farm contribution of purchased or brought in cattle requires a carbon footprint analysis of that upstream stage, another farm. This would require another study with scope similar to the present study, centric to cattle production in northern WA. As this is outside the reasonable expectation of the present study it has been excluded, but however warrants further investigation for future carbon footprint analyses of livestock systems.

- The GHG emissions associated with land use (LU, i.e. soil sequestration) and direct land use change (dLUC, i.e. C change following conversion of land use type) were not included in this study due to scope and data availability constraints. As discussed in Chapter Two, these emission sinks and sources are often omitted, due to a lack of appropriate and relevant data at both regionalised and finer scales, along with the many uncertainties which coincide with the calculation of these emissions. The few Australian studies which do consider dLUC conducted analyses at state or regional scales, utilising regional standardised datasets (Henry et al., 2015a; Wiedemann et al., 2015d). These studies acknowledged the considerable uncertainty inherent in their results and recommended caution in their interpretation given the lack of available data in a spatially and temporally disaggregated format. A unified call has been made for comprehensive and transparent data at a finer resolution, particularly with regards to soil carbon fluxes associated with both LU and dLUC, in order to provide confidence in quantified impacts (LEAP, 2015b). Translating such results to farm-scale is not possible at present. However, Chapter Six explores the potential impact of C sequestration from LU using data sourced from literature.
- Emissions associated with on-farm electricity use were excluded. Common

sources of on-farm electricity use include; water irrigation, livestock housing, milking and shearing (LEAP, 2015b). Of these, the only source applicable to the enterprises assessed in this study was that from shearing. However, as emissions from wool were excluded this data was not included in the inventory. According to LEAP (2015b) other, more significant, contributions from electricity occur in stages further down the supply chain, such as meat processing. As these stages fell outside the system boundary of the present study, these emissions were excluded.

- Fuel usage from on-farm machinery operation was included for both farm-owned machinery and contractors involved in on-farm activities (LEAP, 2015b).

3.3.3 Farm selection and data collection

3.3.3.1 Case study farms

Case study farms were selected on the basis that their participation would enable a range of livestock enterprises to be represented. The following factors were considered during the selection process:

- Location (within key agricultural zones in south-western Australia)
- Annual rainfall (low, medium and high rainfall zones)
- Enterprise type (sheep, beef, mixed system)
- Pasture species (annuals, perennial grasses, perennial shrubs)
- Availability of farm records

Each selected case study farm maintained a perennial pasture component within their pasture system which enabled the study to investigate the potential impact of perennials on the carbon footprint of broadacre livestock systems. The establishment of perennials in each system meant that, in some instances, the farms were able to adopt some productivity-enhancing practices, enabled by the increased productivity of maintaining perennial pasture (explored in later Chapters). This, in some instances, set them apart from some of the approaches of more traditional annual pasture dominant livestock production systems in the region. In the present study, it enabled close examination of the impact of such practices on the carbon footprint.

In total, four case study farms, located in the Northern Agricultural Region (NAR), the Wheatbelt and the Great Southern were analysed. Six farms were initially selected; Lancelin, Dongara, Moora, Wickepin, Bremer Bay and Manypeaks, each denoted by the town they were in closest proximity to. These farmers were recommended by various contacts made in the beginning months of the research. However, two farms

were omitted during the process of the study; Moora and Manypeaks. Moora was eliminated in the first year of the study as the farmer sold the farm. In the third year of the project, Manypeaks was also removed due to study workload constraints and data availability issues. Ideally, to obtain a more representative range of sites, it would have been desirable to include these two farms. However, the scope of the project was already substantial and it was determined that the existing farms enabled sufficient analyses to address the study objectives. The carbon footprint Frameworks provide opportunity for further locations to be modelled into the future.

Figure 3.3 - Locations of each case study farm in south-western Australia (locations are approximate)

3.3.3.2 Primary sources of data

The primary sources of data for this study were the participant farmers. Every possible attempt was made to incorporate as much of the data provided by the farmer and as accurately as the Frameworks would permit. In some cases, primary data was also obtained from external organisations, such as DPIRD or universities, who had conducted on-farm trials and collected information relevant specific to the property.

3.3.3.2.1 Questionnaire development

Two farmer-orientated questionnaires were developed in the opening months of

research; one tailored to sheep production and another tailored to beef cattle production (Appendix B). The purpose of these questionnaires was to obtain sufficient information to calculate the carbon footprint of each case study farm. The information collected included;

- property information (location, soil type, rainfall)
- pasture information (species, area, seeding rates)
- crop information (varieties, area, seeding rates)
- livestock information (stock numbers, liveweights, breeding information)
- grazing management
- production calendars
- chemical use (animal, pasture and crop); and
- machinery operation.

Each participant farmer was emailed a questionnaire to complete prior to any farm visits and interviews.

3.3.3.2.2 Ethics approval and consent forms

Ethics approval for this project was granted by the Curtin University Human Research Ethics Committee in 2014. The research was considered “low risk” as participant involvement was restricted to surveys, interviews, verbal and written communications.

Prior to commencing data collection, all participants were required to sign a consent form (Appendix A).

3.3.3.2.3 Interviews and fieldwork

Field visits to each case study farm were conducted throughout 2014. Each farm visit included a face-to-face interview with the participant farmer and a tour of the property. During the interviews, the questionnaires were discussed in depth and any outstanding information collected. The interviews were designed in such a way as to facilitate open-ended discussion between the farmer and researcher. This provided a greater understanding behind each farmer’s decision to introduce perennials and any ongoing benefits following their introduction.

In some case, farms were visited more than once if additional detail or clarification was required. Alternately outstanding information required following the farm visit was obtained through phone and email conversations with the participant farmers.

3.3.3.3 Secondary sources of data

Not all the required information could be obtained directly from the participant farmers or from farm-specific field trials conducted by other organisations. In such cases, the

missing information was obtained from other sources. The most important of these sources was the GrassGro software. This program was used to model pasture growth, biomass and nutritive properties of the annual pasture at each of the case study farms. Section 3.4 outlines this stage in greater detail.

Other secondary sources included peer-reviewed journal publications, industry publications, websites and personal communications with industry and academic personnel. In all cases, the quality of data was ensured through its technological, geographical and temporal representativeness. The following are examples of information collected from secondary sources;

- Livestock emission data
- Crop yields, stubble nutritive properties and decomposition rates
- Plant and animal chemical manufacturing and transportation data
- Supplementary feed nutritive properties
- Weather data
- Farm machinery specifications; and
- Pasture growth rates, biomass, nutritive properties (those not modelled n GrassGro).

3.3.4 Carbon footprint Frameworks

This Section describes each component of the developed Frameworks, including key assumptions, information sources, calculations and outputs. Each component can be assigned to the three key stages of the carbon footprint calculation.

1. Data collection for inventory input.
2. Calculation of inventory outputs in the form of pre- and on-farm emissions (i.e. kg CH_4, kg N_2O).
3. Conversion of these outputs to a common metric (i.e. kg CO_2-e/kg LW produced for sale).

The development of these Frameworks represented a significant body of work and a major stage of the project, enabling the quantification of all inputs and processes on a livestock production system, and subsequently the carbon footprint of that system.

The flows of information between Framework components are presented schematically in Figure 3.4. The “Farm specifications” component acted as the collection point for all inventory inputs. This inputted data was then fed into the respective components, each representing a GHG emission source (i.e. enteric fermentation, transportation). Following the calculation of the emission output within

each respective component, the GHG emission information flowed into the “Impact assessment” component for consolidation and calculation of the carbon footprint.

For each of the Frameworks components described in the below subsections, the individual input parameters requiring data input and the source of each input can be found in the respective tables of Appendix C. The information in these appendices can be examined in conjunction with the information provided in this Chapter.

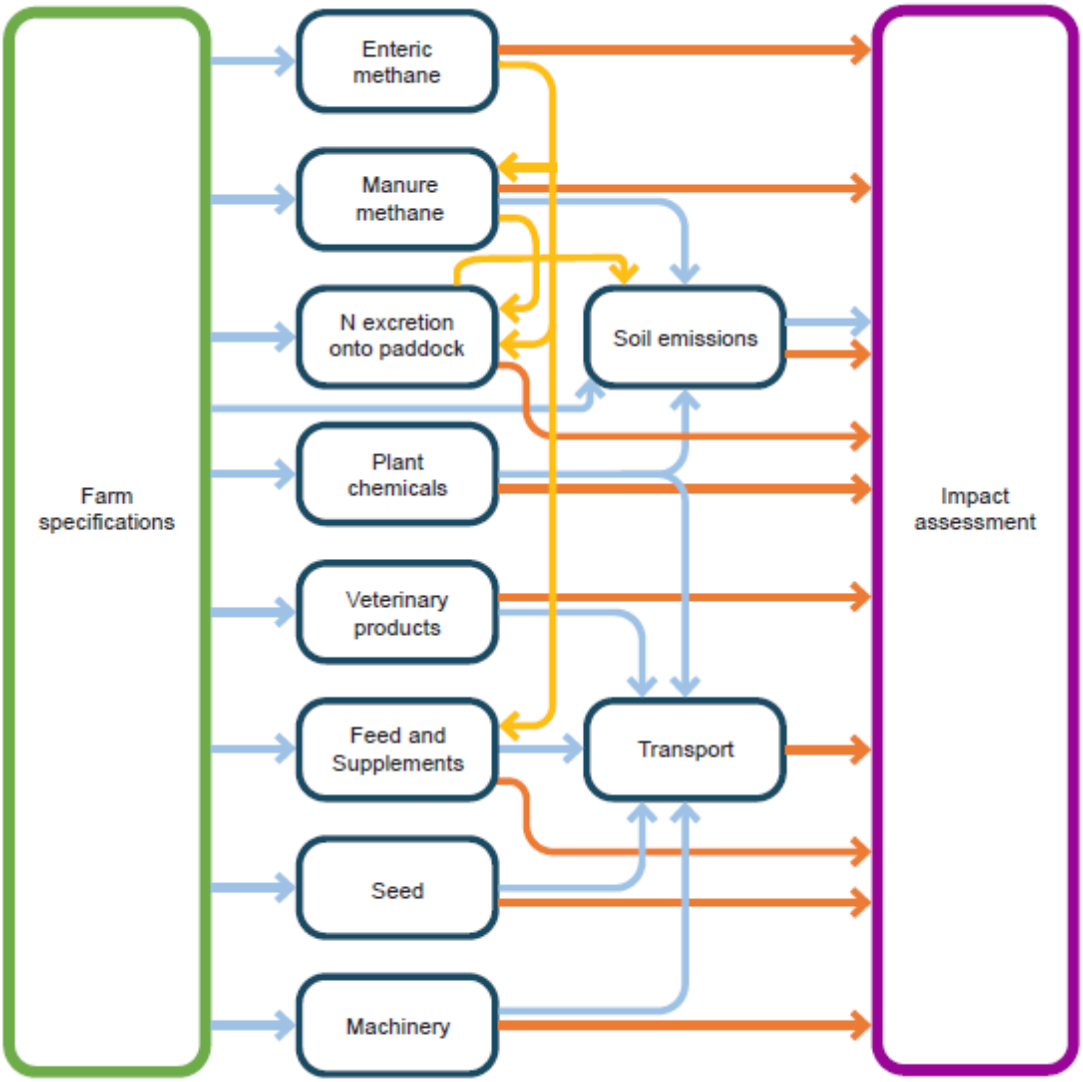


Figure 3.4 - Schematic representation of input flows and calculation linkages between the components of the carbon footprint Frameworks

Note: Blue lines represent flows of input information, yellow lines represent calculated variables also used in the calculation of other inventory outputs and orange lines represent the calculated inventory outputs (CH_4, N_2O and CO_2) prior to conversion to carbon dioxide equivalent (CO_2-e).

3.3.4.1 Farm specifications

The purpose of the "Farm specifications" component was to act as a collection point for all farm information required to calculate the carbon footprint. The information required for this component can be separated into four sections; livestock, feedbase (i.e. pasture or crop or feedlot) and supplementary feed. For each parameter within "Farm specifications", farm-specific data and GrassGro output was preferred to other, non-location-specific sources. In cases where this information was unavailable, regionally specific information was obtained from literature and industry reports.

The data inputted into the parameters were then used in calculations throughout the Frameworks, via automatic links, to calculate GHG emissions of sources across the farm (Figure 3.4).

3.3.4.1.1 Livestock parameters

Livestock parameters were required for the calculation of most emissions sources, including animal emissions, soil emissions and emissions associated with the production and transportation of veterinary products and supplementary feed. This Section outlines the livestock information required by the Frameworks and any relevant assumptions made for the present study. Appendices C and D provide further information regarding the required information and example calculations.

General livestock information

The first step of the analysis involved the identification of the feedbases (i.e. pasture, crop, feedlot) in the farming system. The farm's flock or herd were separated into livestock classes at this initial stage. Once inputted into "Farm specifications" this information automatically populated into the other Framework components. This initial step was essential for the analyses, facilitating the examination of emissions between feedbases and stock classes.

The next parameters requiring input were annual lambing/calving, marking and weaning rates, along with the information necessary to calculate livestock growth rates. Such information included birth, weaning and sale weights and ages. Growth rates were used to calculate monthly liveweight and energy requirements which, in turn, were required to calculate enteric methane excreta nitrous oxide emissions. Growth rates were calculated for all immature livestock, such as unweaned and weaner lambs and calves, heifers and wethers. Sheep over two years and cattle over three years were assumed to have reached their mature liveweight (Peters et al., 2009). The Frameworks were able to capture the growth rate changes of immature animals as they transitioned to different stages (i.e. suckling, post-weaning or feedlot)

and the ensuing effect on emissions. The actual growth rate of an animal was dependent on factors such as breed, the quality of supplied feed or weaning status. This information was sourced from the farms. Growth rates (kg/week) were calculated using equation 3.1.

$$\text{Growth rate} = \frac{(LW_{t=n} - LW_{t=0})}{(Age_{t=n} - Age_{t=0})} \quad (3.1)$$

Where:
Growth rate = liveweight gain (kg/week)
LW = liveweight (kg)
Age = age of animal (weeks)
t=0 = liveweight/age at start of considered stage of growth
t=n = liveweight/age at completion of considered stage of growth

Though the growth rates (and thus liveweight) of both immature and mature animals are also dependent on feed availability and quality, such fluctuations were not considered in the Frameworks due to the complexity of calculating these changes (CSIRO, 2007). This is consistent with other studies (Peters et al., 2009) and good practice recommendations (IPCC, 2006), where it is assumed that liveweight gains and losses due to feed intake changes balance out over a production year. An adjustment to the Frameworks to account for this could be made in the future.

Once growth rates were calculated, the monthly stock count and liveweight for each stock class across each feedbase was inputted. Livestock moved between stock classes as their physiological status changed and moved across feedbases because of grazing management strategies. Monthly liveweights of immature livestock were populated automatically from the calculated growth rates. Production events such as lambing or calving and weaning were noted, important for the subsequent carbon footprint analyses and identification of emission influencing practices. The number of animals sold from each stock class and average liveweight at sale were recorded. These values were required for the calculation of allocation factors for co-products as well as the FU.

Specific National Inventory Report information

The Frameworks followed the recommended methodologies of the most recent 2018 Australian National Inventory Report (NIR)(DISER, 2020b) for the calculation of livestock emissions. The additional livestock information required by the NIR methodologies, along with study-specific assumptions are outlined below. The

specific equations followed can be sourced directly from the 2018 NIR.

Monthly proportion of ewes/cows lactating.

The study used farm-specific information for this variable rather than state-specific NIR recommendations. For ease of feed intake calculations, in each Framework mature breeding livestock were separated into two sub-stock classes; lactating animals and dry/pregnant animals. Livestock were moved into the lactating sub-stock class when they commenced lactating and were moved back into the dry/pregnant sub-stock class following weaning. Separating these physiological states improved the accuracy of intake and emission calculations. Immature breeding stock (i.e. heifers, maiden ewes) were unable to be separated based on physiological status due to stock class restrictions. As such, the proportion of animals lactating was calculated, reflecting the lambing/calving rate and weaning rate.

Monthly feed adjustment (FE).

This applied to the Beef Framework only. Following the NIR, feed intake of lactating cows was assumed to increase by 30% for three months after calving and then by 10% for three months following. This factor was utilised in the enteric methane emission calculations (Section 3.3.4.2).

Monthly milk intake (MC) and milk production (MP)

NIR default values for milk intake and production were adopted in this study. For sheep, milk intake and production were assumed to be 1.6 kg/day. In the case of cattle, milk intake and production were assumed to be 4.0 kg/day in the first three months after calving and 3.0 kg/day from three to six months after calving for Brahman cross breeds. For Shorthorn breeds, these values were 6.0 and 4.0 kg/day, respectively. These values were utilised to calculate enteric methane from suckling lambs and calves, and nitrous oxide from manure for all stock (Sections 3.3.4.2 and 3.3.4.4).

Monthly greasy wool production and clean yield percentage.

This applied to the Sheep Framework only. Instead of NIR default values, farm-specific values were recorded for greasy wool production and clean yield percentage of the wool. This enabled the calculation of clean wool production (WP, kg/head; equation 3.2), a parameter required for the calculation of excreta nitrous oxide emissions (Section 3.3.4.6).

$$WP = GW \times CY \quad (3.2)$$

Where:

WP = clean wool production (kg/head)

GW = greasy wool production (kg/head)

CY = clean yield percentage, the proportion of clean wool after impurities are removed (%)

Standard reference weight (SRW)

CSIRO (2007) defines standard reference weight (SRW) as "…the liveweight of an animal (excluding fleece and conceptus) when skeletal development is complete and the condition score is middle of the range". NIR provides state defaults values for SRW. However these values are specific to large Merino sheep and short-horned cattle (CSIRO, 2007).

To improve the accuracy of the Framework calculations, the SRWs recorded were farm-specific and were inputted as the mature male and female liveweights of the breed produced on each farm. Where possible, this information was obtained from the farm, and in other cases, breed-specific mature liveweight information was obtained from secondary sources. This value was then utilised in the subsequent calculation of nitrous oxide emissions from excreta (Section 3.3.4.6).

Livestock energy requirements

The final livestock parameters required for the "Farm specifications" component were the energy requirements of each stock class. The daily metabolisable energy requirement of an animal (ME MJ/head/day) was required to calculate the supplementary feed component of total feed intake. Section 3.3.4.1 explains these calculations in detail.

The ME value represented the total daily energy required for maintenance, growth and lactation by an animal within a stock class. This was obtained by converting dry standard equivalent (DSE) ratings into ME equivalent, using a conversion rate of 8.3 MJ ME per 1 DSE. For growing animals, the additional ME required for growth was calculated as 35 MJ ME per kg liveweight. These conversion rates, along with specific DSE ratings, were obtained from the Prograze manual and Lifetimewool program (Bell & Allan, 2000; Lifetimewool, 2007).

3.3.4.1.2 Grazed feedbase

A key goal of the Frameworks was the capture of the intra-annual fluctuations of feedbase attributes, particularly pasture. These attributes are critical in determining

animal and soil emissions. It was necessary to develop a tool that enabled the use of information more comprehensive than NIR default values or the regional averages adopted in many carbon footprint studies. The use of monthly data, farm-specific feedbase information and biophysical modelling all enhanced the output of the carbon footprint analyses. This Section outlines the feedbase information required by the Frameworks to achieve this.

Each Framework allowed for the inclusion of up to eight different pasture and/or crop types. This was particularly useful when comparing different annual and perennial pastures and when livestock grazed multiple crop stubbles. The total area (ha) and years since establishment was recorded for each pasture and crop. Whether a the feedbase was grazed was also recorded for each month, based on grazing management information obtained from each farm. These values were automatically linked to ensure zero values during months without grazing.

Monthly dry matter availability (DMA), ME[2], dry matter digestibility (DMD) and crude protein content (CP) values were obtained for each farm. This information was necessary for the calculation of monthly animal and soil emissions, along with emissions associated with supplementary feed production. The method by which these monthly values were obtained differed for annual pasture, perennial species and crop stubble.

Annual and select perennial grasses

For all farms, annual pasture attributes such as DMA, DMD and CP were modelled in GrassGro with the monthly output then inputted directly into the Frameworks.

Other perennial grasses and perennial shrubs

At the time of this study, with the exception of a beta kikuyu (*Pennisetum clandestinum*) parameter set, there were no existing GrassGro parameter sets for subtropical grasses and perennial shrubs. Pasture attributes were instead sourced from secondary sources and manual calculations. Though other studies such as Thomas et al. (2012) and Doran-Browne et al. (2015) have instead modelled substitute parameter sets in lieu of the pasture species under investigation, this approach is accompanied by a high degree of uncertainty, particularly when modelling temperate C3 species in lieu of subtropical C4 species.

To calculate the monthly DMA (kg DM/ha) of perennials, the first step was to obtain the DMA of the month grazing commenced. For perennial grasses, this value was obtained directly from the Department of Primary Industries and Regional

[2] In the Frameworks, ME was derived from DMD using ME= 0.1604 x DMD – 1.037 (Minson & McDonald, 1987)

Development (DPIRD)(Moore et al., 2009b) or from regionally appropriate literature for the corresponding month or season. For perennial shrubs, where plants are sown in rows with annual pasture in the inter-row, the DMA had to consider both the perennial and annual components of the sward. The perennial component was obtained from regionally appropriate literature adjusted for planting density. The annual component was assumed to be the GrassGro modelled annual pasture DMA for that month, adjusted for the area the annual pasture occupied. For example, if the GrassGro modelled annual pasture DMA was 1,800 kg DM/ha and the annual species occupied 60% of the paddock, while the perennial shrubs occupied the remaining 40%, the annual pasture contribution to total DMA would be 1,080 kg DM/ha. The specific approaches adopted for each farm are outlined in Chapter Four.

There are limitations to calculating the DMA of mixed perennial shrub and annual pasture paddocks in the manner described above as it does not consider the influence of factors such as perennial-annual species interactions and selective grazing by livestock. However, without measured data or biophysical models able to account for these interactions, it was deemed to be the most appropriate manner of incorporating the annual pasture into the DMA values. The only other whole-farm analyses considering perennial shrubs (Harrison et al., 2015; Taylor et al., 2016), modelled the DMA of the perennial pasture to match the energy requirements of the grazing livestock, a reverse approach. Such an approach underestimated DMA and pasture residue in the present study, so this approach was deemed unsuitable.

Once the DMA of the first month of grazing was obtained, the DMA of the following month was calculated using calculated feed intake values for that starting month, along with pasture growth rates and a wastage factor. Section 3.3.4.2 details the adopted approach to calculate feed intake by livestock. Daily feed intake per stock class was converted to monthly intake per pasture type. Regionally specific daily perennial pasture growth rates were obtained from DPIRD or relevant literature. A pasture wastage factor of 20% was adopted to account for the loss of pasture through trampling and fouling during grazing (LEAP, 2015b; MLA, 2019d). This calculation was performed for each month grazed, using equation 3.3.

$$DMA_t = (DMA_{t-1} \times (G \times days)) - (Intake_{t-1} \times (1 + W) / Area) \quad (3.3)$$

Where:

DMA_t = dry matter availability of the pasture for the month considered (kg DM/ha)

DMA_{t-1} = dry matter availability of the pasture for the previous month (kg DM/ha)

G = plant growth rate (kg DM/ha/day)

days = number of days in the month considered

$Intake_{t-1}$ = total intake by livestock on the pasture during the previous month (kg DM)

W = wastage factor (%)

Area = total area of pasture (ha)

For perennial grasses, monthly DMD and CP content were obtained directly from DPIRD or from regionally specific literature. For pasture with perennial shrubs and annual species, these were calculated as a weighted average between the two.

Crop stubble

Crop residue, also known as stubble yield, is the plant material remaining after harvest. In this study, where livestock grazed crop stubble they produced animal emissions which were included in the carbon footprint analysis. The decomposition of crop residue remaining after grazing also produces nitrous oxide emissions.

Where stubble was grazed, the quantity of crop residue remaining at harvest was used to represent the starting DMA. Crop residue (kg DM/ha/year) was calculated following Unkovich et al. (2010a) as indicated in equation 3.4.

$$\text{Crop residue} = Y_g \times \left[\frac{(1 - HI)}{HI}\right] \quad (3.4)$$

Where:

Crop residue = quantity of plant material remaining after harvest (kg DM/ha/yr)

Y_g = total grain yield of the crop adjusted for dry matter content (kg DM/ha/yr)

HI = harvest index, a measure of the productivity of a crop.

Farm-specific grain yields were obtained from farmers while HIs for each crop were obtained from an Australian review (Unkovich et al., 2010a) rather than the national default averages in the 2018 NIR.

Following the calculation of crop residue, monthly DMA was calculated in the same manner as the perennial pasture (Equation 3.3), but with growth rates adjusted to zero. The DMA remaining at the end of the grazing period was applied in the calculation of nitrous oxide emissions from crop decomposition described in Section

3.3.4.6. Stubble DMD and CP content were obtained from relevant literature.

3.3.4.1.3 Feedlot

In the present study, feedlots in the case study farms presented as small paddocks where stock were fed high quality rations for a short period before being sold. In the Frameworks, emissions associated with the feedlots were calculated separately from those arising from pasture and crop.

Along with monthly stock numbers and liveweights in the feedlot, information regarding the provided feedlot ration was also required. As livestock were fed a minimum of two feed types (i.e. hay and grain) as part of their feedlot ration, using the proportion of the total ration each comprises, the weighted average of ME, DMD and CP content of the ration was calculated.

In some cases, feedlot grain was fed as a mix with other components, such as molasses and urea, so the first step was to calculate the total nutritive properties of this mix. The next step was to calculate the overall nutritive attributes of the ration using the respective proportions of total ration made up by grain and hay.

Step 1: Calculation of the nutritive properties of the grain mix (if applicable)

a) Calculation of the mass of each ingredient within a kg of mix on a dry matter basis ($Mass_{ingredient}$; kg DM/kg mix):

$$Mass_{ingredient} = Q \times \rho_{ingredient} \times DM_{ingredient} \quad (3.5)$$

Where:

$Mass_{ingredient}$ = mass of the ingredient in a kg of mix (kg DM/kg mix)

Q = mass (kg) or volume (l) of ingredient per kg of grain mix

$\rho_{ingredient}$ = density (kg/l; where ingredient is a liquid)

$DM_{ingredient}$ = dry matter content (%)

b) Calculation of the weighted average of each attribute (DMD_{mix}, ME_{mix} and CP_{mix}; %) of mix:

For example;

$$DMD_{mix} = \frac{\Sigma(DMD_{ingredient} \times Mass_{ingredient})}{\Sigma Mass_{ingredient}} \times 100\% \quad (3.6)$$

Where:

DMD_{mix} = weighted DMD of the grain mix (%)

$DMD_{ingredient}$ = DMD of each ingredient in mix (%)

$Mass_{ingredient}$ = mass of each ingredient per kg of mix (kg DM/kg mix)

Step 2: Calculations of the weighted value of each attribute (DMD_{ration}, ME_{ration} and CP_{ration}; %) of the total feedlot ration

For example;

$$DMD_{ration} = (DMD_{mix} \times Proportion_{mix}) + (DMD_{hay} \times Proportion_{hay}) \quad (3.7)$$

Where:

DMD_{ration} = weighted DMD of feedlot ration (%)

$DMD_{mix/hay}$ = DMD of the grain mix/hay (%)

$Proportion_{mix/hay}$ = proportion of grain mix or hay in feedlot ration (%)

3.3.4.1.4 Supplementary feed

Supplementary feed was provided to maintain animal condition, usually during the non-growing season. In the Frameworks, the monthly occurrence and type of supplementary feed provided was recorded for each feedbase and stock class. These were linked to the feed intake calculations, methane emissions from enteric fermentation and manure, along excreta nitrous oxide emissions. Four different supplementary feeding options were available.

1. Supplementary feed type 1 only (i.e. grain).
2. Supplementary feed type 2 only (i.e. hay).
3. Supplementary feeds type 1 and 2 together.
4. No supplementary feeding.

The option recorded determined how monthly feed intake values were calculated later in the Framework. For example, if the first option was recorded for ewes grazing annual pasture in February, then a mixed diet of pasture and the supplementary feed nominated in that option was used to calculate the feed intake of those ewes. Once the monthly supplementary feeding regime was nominated, the nutritive properties, including ME, DMD, DM content and CP content (or weighted values if option three is selected) of the feed supplied were recorded.

Along with the two supplementary feed types described above (i.e. one and two), the Frameworks also recorded a third supplementary feed representing the supply of protein-only supplements (i.e. mineral blocks and urea), necessary to calculate

excreta nitrous oxide emissions.

3.3.4.2 Methane emissions from enteric fermentation

The “Enteric methane” component of each Framework calculated enteric methane emissions from livestock, through two key stages;

a) The calculation of monthly feed intake by each stock class on each feedbase, accounting for supplementary feeding where applicable; and
b) Applying the feed intake values derived in stage (a), the calculation of enteric methane production.

The calculated intake values were also required for the calculation of other emissions, such as nitrous oxide and methane emissions from animal waste and pre-farm emissions from the production of supplementary feed (Figure 3.4).

Enteric methane emissions were calculated by applying the methodologies outlined in the 2018 NIR for sheep and beef cattle (DISER, 2020b). The 2016 NIR (DEE, 2018) amended the recommended methodology for estimating cattle enteric methane emissions to that proposed in Charmley et al. (2016), replacing the previous method outlined by Blaxter and Clapperton (1965). As such the Beef Cattle Framework was amended in 2018 to reflect this change. The specific equations followed are not replicated here, only descriptions. Appendices C and D provide further information regarding the required information and example calculations.

For sheep, the calculation of enteric methane production involved the following steps;

1. Calculation of potential intake (PI), defined as the maximum possible intake of feed by the animal, determined by the quality of feed available and liveweight of the animal considered.
2. Calculation of relative intake (RI), defined as the proportion of PI actually consumed by the animal. In NIR, this is a function of feed availability.
3. Calculation of additional intake for milk production (MA).
4. Calculation of predicted total feed intake using the product of PI, RI and MA.
5. Calculation of methane production (kg/head/day), dependent on total feed intake.
6. Calculation of total methane production by each stock class for each season.

The process, as defined by the NIR, was slightly different for cattle;

1. Calculation of PI, determined by liveweight and liveweight gain (LWG) of the animal considered.
2. Calculation of MA.

3. Calculation of predicted total feed intake using the product of PI and MA.
4. Calculation of methane production (kg/head/day), dependent on intake.
5. Calculation of total methane production in each stock class for each season.

3.3.4.2.1 Study-specific inventory assumptions

Within both Frameworks, additional assumptions were made in parallel to the above NIR methodologies. This was necessary to address the objectives of this study. These assumptions, along with the rationale behind each, are detailed in this Section.

Stock class considerations

The 2018 NIR separates sheep into the following stock classes when calculating emissions;

- Rams (>1 year),
- Wethers (>1 year),
- Maiden ewes (>1 year),
- Breeding ewes (>1 year),
- Other ewes (>1 year), and
- Lambs and hoggets (<1 year).

While beef cattle are separated into;

- Bulls (>1 year),
- Bulls (<1 year),
- Steers (<1 year),
- Cows (1-2 year),
- Cows (>2 year),
- Cows (<1 year), and
- Steers (>1 year).

Calculating emissions using these stock classes is useful when undertaking calculations on larger scales, such as state or national, or when modelling a theoretical farm to test a hypothesis. By contrast, the present study utilised case study farms. The livestock on these farms did not always fit conveniently within the NIR designated stock classes for the following reasons:

- There is a risk of oversimplifying calculations and not capturing the variations influencing emissions. For example, NIR groups lambs, bulls, cows and steers under one year of age. However, growth rate fluctuates through this first year, from suckling, to weaning, to feedlot if applicable. In the present study the primary product of both the sheep and beef enterprises were livestock under one year (see Chapter Four). However, livestock were also kept on as

replacements or for sale in the following year. Averages do not allow for examination of stock sold, for example, at six months, versus stock retained beyond a year, despite very different growth rates and ensuing emissions. Similarly, sheep have not reached mature weight by the age of one, nor cattle by two years. Yet the NIR groups stock of this age with mature stock, irrespective of different intake requirements and emission outputs.

- There is an opportunity to enhance the final analysis of emission results by dividing stock into classes in a way that targets the research question. For example, separating lactating and dry livestock, rather than grouping them into one stock class, allows for detailed examination of emission sources and effectiveness of mitigation strategies (Beauchemin et al., 2010; Beauchemin et al., 2011). It is still possible to group different classes during the final analyses if required. For example, the grouping of all replacement heifers once emissions have been calculated separately for unjoined heifers, first-calf heifers and second-calf heifers, ensures that the very different intake requirements (due to growth and lactating or dry status) and ensuing emissions from each are accounted for. It is not possible to breakdown the emissions in this manner using the NIR stock classes, potentially removing the ability to examine hotspots and identify mitigation opportunities beyond broadscale observations.
- The NIR stock class categorisation does not allow for allocation of farm emissions to co-products following impact assessment. For example, when using the method of economic allocation, stock classes need to be separated into the production and breeding classes specific to the farm as they return different market prices. Using the NIR stock classes could mean two products with very different market values could be grouped into the same class, resulting in a misallocation of emissions.

For the above reasons, stock classes presented in this study differed across farms and were specific to the collected farm information.

Monthly calculations

The NIR calculations are conducted seasonally. As discussed previously, it was determined that monthly calculations were more appropriate as they capture the temporal variation in pasture and stock. This allowed for a closer examination of pasture changes and resultant effects on overall emissions.

Grouping of production events and mid-month changes

In practice, on-farm production events such as lambing and calving, weaning and sale of stock occur over a period of days, weeks or even months. Due to the complexity of accounting for this in emission calculations, in whole-farm analyses, each event is usually grouped and assumed to occur on a set date (Beauchemin et al., 2010; Dougherty et al., 2018; Harrison et al., 2014a; Wiedemann et al., 2016c). This assumption was also made for this study. As the Framework calculations occur monthly, most events were assumed to occur at the beginning or end of a month. This approach was deemed to be more accurate than studies which conducted analyses on a seasonal basis, thus diluting the effect of such production events over a season, rather than a month.

Some of the participant farmers, however, provided information which made it necessary to incorporate mid-month changes. For example, in one case lambs were weaned at the beginning of July at three and a half months which meant that lambing had to be modelled in mid-March. In these cases, the Framework was manually adjusted to account for the mid-month change in stock numbers, feed intake and resultant emissions.

Adjustment of cattle intake calculations to consider feedbase attributes

The 2018 NIR calculates the feed intake (I; kg DM/head/day) of beef cattle using the equation of Minson and McDonald (1987), corrected for milk production:

$$I = (1.185 + 0.00454LW - 0.0000026LW^2 + 0.315LWG)^2 \times MA \quad (3.8)$$

Where:

I = feed intake (kg DM/head/day)

LW = liveweight of the animal (kg)

LWG = liveweight gain (kg/head/day)

MA = additional intake for milk production (kg DM/head/day)

This equation derives feed intake from LW and LWG and assumes that;

a) LWG is linearly related to the DMD of feed consumed; and
b) that feed availability (i.e. DMA) is non-limiting.

This linear relationship between LWG and DMD assumes that reliable growth rate information is available. In the present study, per the 2018 NIR, zero net growth over the production year for mature livestock was assumed. Growth rates for immature livestock were calculated using weight and age data provided by farmers, not

measured growth rates for each feedbase on each farm. Therefore, predicting intake based on these values would be insufficient.

The second assumption made by Minson and McDonald (1987) was not applicable in the present study where feed availability is invariably limited. CSIRO (2007) states that the intake of cattle is negatively affected by feed availability when DMA falls below 3 t DM/ha. After examining the assumptions behind the NIR recommended intake calculations and considering that the research questions of this study focus heavily on pasture, it was decided to predict intake by cattle using the methods outlined by CSIRO (2007). Similar to the NIR methodology for predicting sheep intake, these methods take into consideration the effect of DMD and DMA through the calculation of relative intake. Adjusting cattle intake to account for DMA allowed for a more considered approach when comparing pastures, while also enabling a more appropriate comparison between sheep and beef enterprises.

Adjustment of feed intake to incorporate supplementary feed

Throughout southern Australia, supplementary feed is routinely supplied to livestock during the dry season. The current NIR methodology does not consider supplementary feed intake, unless it is included in the seasonal DMD, CP and DMA values prior to the intake calculations. To do so requires prior knowledge of the proportion of feedbase intake versus supplementary feed in the animal's diet. Such specific information may be accessible in intensive systems where such control and close monitoring is possible, however this is not the case in extensive pasture systems with low management.

In the present study, the participant farmers were unable to provide adequate information regarding supplementary feed supplied per head. None were able to provide values specific to each stock class. The second difficulty was the quantification of supplementary feed in terms of DMA using NIR methodology, particularly when the animal is also consuming pasture with a modelled/measured DMA. Thirdly, it is not possible to separate animal emissions resulting from pasture intake from those resulting from supplementary feed intake. Being able to separate these emissions enhances the analysis and assessment of the effectiveness of mitigation strategies.

It was thus decided to develop the Frameworks to enable the calculation of the supplementary feed component of feed intake. This was conducted as a step following the initial feedbase intake calculations. Supplementary feed intake was calculated as the additional feed required to meet the energy requirement of the animal. This was

a multi-step process. For each month, for each stock class and on each feedbase the following calculations were undertaken.

Step 1: Calculation of the ME intake (MEI_f; MJ ME/head/day) obtained from the feedbase intake

$$MEI_f = I_f \times ME_f \quad (3.9)$$

Where:

MEI_f = ME intake (MJ ME/head/day)

I_f = calculated intake of the feedbase (kg DM/head/day)

ME_f = ME content of the feedbase (MJ/kg DM)

Note: If the animal was suckling (between 8 weeks and weaning) then an additional step was incorporated to include energy obtained from milk consumption (MC; MJ ME/head/day), calculated as:

$$MC = I_m \times ME_m \quad (3.10)$$

Where:

MC = milk consumption (MJ ME/head/day)

I_m = milk intake using 2018 NIR default values (Section 3.3.4.1)

ME_m = ME content of milk for sheep and cattle (4.7 and 3.1 MJ/kg milk, respectively) (Freer et al., 2012)

Step 2: Calculation of the energy deficit (if present; MEI_{def}; MJ ME/head/day) following pasture or crop stubble intake

$$MEI_{def} = MEI_f - ME_{req} \quad (3.11)$$

Where:

MEI_{def} = energy deficit of the animal grazing the feedbase (MJ ME/head/day)

MEI_f = energy intake from the feedbase (MJ ME/head/day) as calculated in equation 3.9

ME_{req} = daily energy requirement of the stock class considered (MJ ME/head/day)

Step 3: Calculation of the supplementary feed intake (I_s; kg DM/head/day) required to meet the energy deficit

If there was an energy deficit following pasture or crop stubble intake, supplementary feed had been nominated to be supplied for that month, then intake was calculated for the nominated supplementary feed type.

If only one type of supplementary feed was supplied, then intake was:

$$I_s = \frac{MEI_{def}}{ME_s} \quad (3.12)$$

Where:

I_s = supplement intake (kg DM/head/day)

MEI_{def} = energy deficit of the animal as calculated in equation 3.11 (ME MJ/head/day)

ME_s = ME of supplement (MJ/kg DM)

If more than one type of supplementary feed was supplied, then intake (I_s, kg DM/head/day) was the total of each supplementary feed type, adjusted for the relative percentage contribution of each to the overall supplementary feed ration provided:

$$I_s = \left[\frac{MEI_{def}}{ME_r}\right] \text{x supp feed}_1 + \left[\frac{MEI_{def}}{ME_r}\right] \text{x supp feed}_2 \quad (3.13)$$

Where:

MEI_{def} = energy deficit of the animal as calculated in equation 3.11 (ME MJ/head/day)

Supp $feed_{1\ \&\ 2}$ = proportion of supplementary feeds one and two in the total supplementary feed ration (%)

ME_r = weighted metabolisable energy of supplementary feed ration (MJ/kg DM)

Step 4: Calculation of total feed intake (I_t; kg DM/head/day)

$$I_t = I_f + I_s \quad (3.14)$$

This final intake value, considering both feedbase and supplementary feed, was used to calculate enteric methane production, manure methane production and excreta nitrous oxide emissions.

The use of energy requirements to calculate animal intake is widely adopted and recommended internationally and within Australia (CSIRO, 2007; IPCC, 2006). Australian programs such as Prograze™, Making More from Sheep, Lifetime Ewe Management and More Beef from Pastures all aimed to educate Australian farmers about the importance of managing feed supply to meet animal demand to improve the production efficiency of their enterprise (AWI & MLA, 2008; Bell & Allan, 2000; Lifetimewool, 2007; MLA, 2018). Farmers are encouraged to calculate supplementary feed supply as determined by animal energy and intake requirements, taking into consideration pasture intake.

In the present study, the use of energy requirements enabled examination of whether

a feedbase met livestock requirements and subsequent calculations of supplementary feed intake. It also assisted the examination of the effect of proposed mitigation strategies on feed supply and livestock intake, which was is not possible usingNIR methodology where strategies which show to reduce emissions may actually restrict animal intake and reduce productivity.

Intake of pregnant livestock

CSIRO (2007) and the NIR assume that feed intake of livestock does not increase during pregnancy. This assumption was followed in the Frameworks, whereby intake of dry and pregnant stock were calculated together.

Enteric methane production of lambs/calves

Methane is not produced from a milk diet. Following IPCC (2006), in the present study enteric methane production in lambs and calves is only considered once they are eight weeks of age. This is assumed to be the stage at which lambs and calves are dependent on milk for approximately half of their overall diet (Gibbs et al., 2002).

3.3.4.3 Nitrogen excretion onto paddock

The “N excretion onto paddock” component of the Frameworks calculated the mass of faecal and urinary N excreted onto pasture, crop stubble or feedlot. The calculated values were then applied in the “Soil emissions” component to calculate direct nitrous oxide emissions from deposited excreta as well as indirect nitrous oxide emissions from atmospheric deposition and from leaching and runoff (Figure 3.4).

The methodologies recommended by 2018 NIR were applied to the Frameworks (see NIR for detailed equations). The specific approach differs for beef cattle and sheep and the processes for calculating each are outlined below. Appendices C and D provide further information regarding the required information and example calculations. The 2016 NIR (DEE, 2018) recommended slight changes to the calculations of N excreted from beef cattle and the Beef Cattle Framework was amended to reflect these recommendations.

For sheep, the calculation of N excretion from excreta involved the following steps:

1. Calculation of CP intake (CPI; kg/head/day), as a function of feed intake, CP content of diet and, if applicable, protein content of milk consumed (MC).
2. Calculation of N retained by the body (NR; kg/head/day), determined by protein required for milk production (MP), wool production (WP) and growth.
3. Calculation of N excreted in faeces (F; kg/head/day), determined by the quantities of undegraded protein from solid feed microbial crude protein, milk protein and endogenous faecal protein,

4. Calculation of N excreted in urine (U; kg/head/day), determined by the difference between N consumed, NR and F.
7. Calculation of monthly total F and U in each stock class.

The calculations were slightly different for beef cattle:

1. Calculation of CPI, as a function of feed intake and CP content of diet.
2. Calculation of N retained by the body, as determined by the protein required for MP and growth.
3. Calculation of F, as per the sheep methodology.
4. Calculation of U, determined by the difference between N consumed, NR, F and dermal protein loss.
5. Calculation of monthly total F and U in each stock class.

3.3.4.3.1 Study-specific inventory assumptions

As for enteric methane production, the Frameworks included additional study-specific assumptions regarding the calculation of N excretion, as outline in this Section.

Calculation of CPI using total feed intake

It was necessary to calculate CPI based on total feed intake. In the case of the Frameworks, this meant calculating the proportion of CPI derived from feedbase and the proportion derived from supplementary feed, if applicable. CPI was thus a function of the monthly feed intake calculated in the “Enteric methane” component and the CP content values inputted in “Farm specifications”. As described earlier, in some instances a third, protein-only, supplement such as urea, was supplied. Where supplied, the CP contribution of this supplement was also considered.

Calculation of weighted DMD and ME

The DMD and ME content of the diet consumed by livestock was required to calculate N excreted in faeces (F). As for CPI, this required both feedbase and supplementary feed components of total intake. Using the monthly intake values calculated in the “Enteric fermentation” component, along with the DMD and ME values inputted in “Farm specifications”, the weighted values of dietary DMD and ME were calculated.

3.3.4.3.2 Methane emissions from manure

The purpose of the “Manure methane” component was to calculate methane emissions from manure. Consistent with other components of the Frameworks, calculations were undertaken monthly for each stock class on each feedbase.

Manure methane emissions were determined by applying the recommended methodology and default EF in the 2018 NIR. Appendices C and D provide further information regarding the required information and example calculations. The

methodology was the same for sheep and cattle and involved the following steps:

1. Calculation of methane production from manure (kg/head/day), dependent on feed intake and the weighted DMD of the diet consumed. The default NIR EF was obtained from the results of Gonzalez-Avalos and Ruiz-Suarez (2001).
2. Calculation of monthly total methane production by the stock class.

3.3.4.4 Pre-farm emissions

Pre-farm emissions considered the production of all inputs and their transportation to the farm, including; chemicals applied to pasture or supplementary feed crops, veterinary products, livestock supplementary feed and ration components, seed and diesel. Calculation of the on-farm consumption of each input occurred in the respective Framework component (i.e. “Plant chemicals”, “Seed”). The calculated totals were used to obtain the emissions associated with the production of the inputs in the “Impact assessment” component. Calculation of the transportation of each input was conducted in the “Transportation” component. Figure 3.4 highlights the linkages between the input and transportation components to the other components within the Frameworks.

The exceptions to the above were that of machinery and diesel. Inventory values associated with the manufacture of machinery and on-farm fuel consumption were both calculated in the “Machinery” components Frameworks. While the transportation of diesel was calculated in “Transportation”, transportation of farm machinery was outside the boundaries of this study and excluded.

The following Section details the calculations associated with the consumption and transportation of each of the farm inputs. Appendices C and D provide further information regarding the required information and example calculations.

3.3.4.4.1 Production of plant chemicals

To account for emissions arising from the production of chemicals applied to pasture or crops, the total application of all chemicals such as herbicides, pesticides, fertilisers during the study period was quantified in the “Plant chemicals” component. These values were then converted to the emissions. In line with the study system boundaries, chemicals applied to produce income crops were excluded.

Information such as the type and density of active ingredients in each chemical was sourced from its Material Safety Data Sheet (MSDS). Where it was not possible to find information pertaining to a chemical of a particular brand, surrogate data was used by substituting a chemical of another brand with similar properties. Chemicals were separated into five categories; herbicide, pesticide, fertiliser, lime and urea. This

was to facilitate the calculation of emissions (kg CO_2-e) in the "Impact assessment" component. Chemical application and emissions were calculated separately for each feedbase following the below steps.

Step 1: Calculation of the annual application of a chemical (C_p; kg or l/yr) to a feedbase

$$C_p = r_p \times Area \quad (3.15)$$

Where:
C_p = annual application of plant chemical to feedbase (kg or l/y)
r_p = plant chemical application rate (kg or l/ha/yr)
Area = total area of feedbase (ha)

For perennial pasture, the application rate (r_p) of chemicals applied at establishment was adjusted for the number of years since establishment. This annualised value was then totalled with the ongoing annual application of that chemical. For example, the total quantity of fertiliser applied ten years previously to establish a perennial grass pasture was divided by ten. This annualised value was then added to the amount of fertiliser applied yearly in an ongoing basis.

Step 2: Conversion of the annual chemical application from l/yr to kg/yr

In some cases, the EF of a chemical required the inventory value to be presented in kilograms. In such cases, the annual application of the liquid chemical (l/yr), as derived in equation 3.15, had to be converted to kg/yr using its density (kg/l).

This step was also undertaken when calculating the transportation emissions of the chemical as the transportation EF required inputs to be presented in kilograms (see Section 3.3.4.4).

Step 3: Conversion of annual chemical application to application of active ingredient

Some EFs were based on the quantity of active ingredient (AI) applied, not total chemical as calculated in equation 3.15. In such cases, annual chemical application (C_{pai}; kg or l AI/yr) was calculated as:

$$C_{pai} = C_p \times AI_p \quad (3.16)$$

Where:
C_{pai} = annual application of plant chemical AI to feedbase (kg or l AI/yr)
C_p = as calculated in in equation 3.15 (kg/yr or l/yr)
AI_p = mass of AI per unit of chemical (kg or l)

Step 4: Calculation of the annual GHG emissions (kg CO_2-e/yr) associated with the production of a chemical applied to a feedbase

The annual chemical application on each feedbase was converted directly to CO_2-e (E_p; kg CO_2-e/yr) in the "Impact assessment" component of the Frameworks using EFs obtained from Simapro databases.

$$E_p = C_p \times EF_p \quad (3.17)$$

Where:

E_p = annual emissions arising from the production of plant chemicals applied to feedbase (kg CO_2-e/yr)

C_p or C_{pai} = as calculated in equations 3.15 or 3.16 (kg/yr or l/yr)

EF_p or EI_{pai} = chemical EF (kg CO_2-e/kg or l plant chemical applied)

3.3.4.4.2 Production of veterinary products

Emissions arising from the manufacture of veterinary products used on-farm were considered. Total quantities of each product, such as vaccinations and drenches, applied to each livestock class during the study period were calculated. As for plant chemicals, these values were utilised to calculate emissions in the "Impact assessment" component.

Information such as the type and mass of AIs for each chemical was sourced from its MSDS, or a substitute chemical used if brand-specific information was unavailable. Chemicals were separated into three categories; vaccinations, drenches and other (i.e. blowfly treatment). Application rates were calculated separately for each livestock class.

Step 1: Calculation of annual veterinary product application (C_v; ml/yr) to a stock class

$$C_v = d_v \times Stock \quad (3.18)$$

Where:

C_v = annual application of veterinary product to stock class (ml/yr)

d_v = annual dosage per head (ml/head/yr)

Stock = number of livestock within stock class (head)

Step 2: Conversion of the annual chemical application from l/yr to kg/yr

As for plant chemicals, the EF for the manufacture of the product would sometimes require the inventory value in kilograms. In such cases, the annual application of the liquid chemical (l/yr), derived in equation 3.18 and converted from ml to l, was converted to kg/yr using it density (kg/l). This step was also required when calculating

emissions associated with the transportation of veterinary products (Section 3.3.4.4).

Step 3: Conversion of annual veterinary product application to application of AI

Some veterinary product EFs were based on quantity of AI applied, not product applied. In such cases annual veterinary product application (C_{vai}; kg or l AI/yr) was:

$$C_{vai} = C_v \times AI_v \quad (3.19)$$

Where:

C_{vai} = annual application of AI proportion of veterinary product (kg or l AI/yr)

C_v = as calculated in equation 3.18 (kg/yr or l/yr)

AI_v = the mass of AI per unit of veterinary product (kg or l)

Step 4: Calculation of the annual GHG emissions (kg CO_2-e/yr) associated with the production of a veterinary product applied to a stock class

The annual mass applied to each stock class was converted directly to CO_2-e (E_v; kg CO_2-e/yr) in the "Impact assessment" component using EFs obtained from Simapro databases.

$$E_v = C_v \times EF_v \quad (3.20)$$

Where:

E_v = annual emissions arising from the production of veterinary products supplied to stock class (kg CO_2-e/yr)

C_v or C_{vai} = as calculated using equations 3.18 or 3.19 (kg/yr or l/yr)

EF_v or EF_{vai} = chemical EF (kg CO_2-e/kg or l veterinary product applied)

3.3.4.4.3 Production of livestock feed and supplements

Earlier Sections outlined the calculation of supplementary feed and feedlot rations. Where farms purchased livestock feed supplies, emissions associated with their production are included within the study system boundary. The first stage, conducted in the "Feed and Supplements" component of the Frameworks, calculated the quantities of each supplement supplied. These were totalled for each stock class on each feedbase, including from the feedlot if applicable. The second stage involved the conversion of these values into emissions in the "Impact assessment" component.

Information regarding each supplementary feed or ration component was obtained from relevant literature, industry reports, or product information. Where brand-specific information was unavailable (i.e. for some mineral blocks), information from substitute products was applied.

Step 1: Calculation of the total intake of a supplement (TI_s; kg DM/month) by a stock class on a feedbase

$$TI_s = I_s \times days \times Stock \quad (3.21)$$

Where:

TI_s = total monthly intake of the supplement by the stock class on feedbase (kg DM/month)

I_s = daily intake of the supplement by an animal on feedbase, calculated in equations 3.12 and 3.13 (kg DM/head/day)

days = number of days in the month considered

Stock = number of livestock within the stock class (head)

The monthly intakes obtained in this step were then totalled across the production year to obtain the annual intake of the supplement by the stock class on that feedbase (kg DM/yr).

Step 2: Calculation of the annual intake of a supplement on an "as-fed" basis

Once the annual intake of the supplement was calculated, it was converted to an "as-fed" total (TI_{saf}; kg/yr). The as-fed value accounted for the DM content of the supplement and the wastage that occurred through processes such as stock trampling and fouling. A wastage factor of 20% was adopted.

$$TI_{saf} = \left[\frac{TI_s}{DM} \right] \times (1 + W) \quad (3.22)$$

Where:

TI_{saf} = total annual intake, on an "as-fed" basis, of the supplement by the stock class on feedbase (kg/yr)

TI_s = total annual intake of the supplement, as calculated in equation 3.21, by the stock class on feedbase (kg DM/year)

DM = D M content of supplement (%)

W = wastage factor (%)

The annual "as-fed" values for each stock class were then totalled across the feedbase to obtain the total annual quantity of supplement provided.

Step 3: Calculation of the annual GHG emissions (kg CO_2-e/yr) associated with the production of a supplement consumed on a feedbase

The annual quantity of the supplement was then converted directly to CO_2-e (E_{saf}; kg CO_2-e/yr) in the "Impact assessment" component using EFs obtained from Simapro databases.

$E_{saf} = TI_{saf} \times EF_s$ (3.23)

Where:

E_{saf} = annual emissions resulting from the production of supplementary feed provided on feedbase (kg CO_2-e/yr)

TI_{saf} = as calculated using equation 3.22 (kg/yr)

EF_s = supplement EF (kg CO_2-e/kg or l supplement provided)

3.3.4.4.4 Production of seed

Emissions were produced during the pre-farm production of seed used to sow pasture and supplementary feed crops. In the Frameworks, the "Seed" component calculated the total annual quantity of seed applied to each feedbase, which was then used to calculate the emissions associated with the production of the seed. Two assumptions were made to facilitate the calculations;

- Seed harvested on-farm and used in subsequent years following storage was excluded to avoid double-counting emissions; and
- Only seed used to sow supplementary feed crops were considered.

Step 1: Calculation of the annual seed application (S; kg/yr) for a feedbase

$S = r_s \times Area$ (3.24)

Where:

S= annual seed application on feedbase (kg/yr)

r_s= annual seed application rate on feedbase (kg/ha/yr)

Area= total area of feedbase (ha)

Per plant chemicals, for perennial pasture the application rate (r_s) was first adjusted for the number of years since establishment.

Step 2: Calculation of the annual GHG emissions (kg CO_2-e) associated with the production of seed applied to a feedbase

The annual quantity of seed sown on each feedbase was converted directly to CO_2-e (E_{seed}; kg CO_2-e/yr) in the "Impact assessment" component using EFs obtained from Simapro databases.

$E_{seed} = S \times EF_{seed}$ (3.25)

Where:

E_{seed} = annual emissions from the production of seed sown on feedbase (kg CO_2-e/yr)

S = as calculated in equation 3.24 (kg/yr)

EF_{seed} = seed EF (kg CO_2-e/kg seed applied)

3.3.4.4.5 Machinery manufacture and operation emissions

Emissions arising from the production and on-farm usage of farm machinery were calculated in the Frameworks. Farm machinery considered included all vehicles (i.e. tractors, harvesters) and farm implements (i.e. seeders, boomsprays) used on a paddock. Utility vehicles were excluded. Per the study scope, machinery use on paddocks with income crops were excluded.

The first stage in the calculation of machinery emissions was to obtain values for the;

- Cost of farm machinery (USD/ha @1998 price); and
- Farm machinery fuel consumption (l/yr).

These values were calculated separately for each type of machinery (vehicle and/or implement) used on each feedbase in the "Machinery" component. Following the calculation of machinery cost and fuel consumption, EFs were applied to calculate the emissions arising from the manufacture of farm machinery, the production of fuel consumed on-farm and the combustion of fuel. These emission calculations were conducted in the "Impact Assessment" component. A lengthy calculation process, the breakdown of the steps followed to calculate the inventory outputs and emissions for machinery manufacture and fuel consumption are presented in Appendix E.

The cost associated with the manufacture of machinery was calculated separately for each implement and vehicle used, following the methodology applied by Biswas et al. (2010) and Barton et al. (2014). Where implements that attach to the tractor were used, the tractor fuel consumption associated with the operation of those implements was calculated separately and then totalled for each feedbase to obtain total of fuel consumption.

3.3.4.4.6 Transportation of inputs

Unlike other the other pre-farm components of the Frameworks, the "Transport" component included the two calculation stages together:

- Calculation of the transportation of inputs (tonnes input per km travelled, tkm, for each mode of transport); and
- Calculation of emissions (kg CO_2-e) resulting from the transportation of each input.

All inputs transported to the farm were considered, including plant and animal chemicals, seed, feed supplements and diesel use on-farm. The impact assessment was calculated in this component to prevent overcomplicating the "Impact assessment" component, as each input comprised multiple modes of transport and multiple EFs.

All modes of transport were considered, from departure from the manufacturer (or farm if applicable) to arrival at the farm-gate. This included shipping, rail, trucks and smaller vehicles. Where ships transported inputs internationally, distance was calculated from the port closest to manufacturer to port closest to local distributer. Where the location of manufacturing facilities was unavailable, it was assumed that the local facility for the company was the departure point.

The Frameworks allowed the calculation of up to four different journeys for each input. In some cases this was necessary, for example where an input was manufactured overseas and journeys included; 1) transport via ship to local port, 2) transport via truck to main distributer, 3) transport via truck regional distributer, 4) transport via farm vehicle to the farm.

Step 1: Calculation of the tonne-kilometres (tkm_i; tkm/yr) associated with the transportation of an input to the farm

Consistent with the remainder of the Frameworks, transport and emissions arising from the transportation of these inputs were calculated separately for each feedbase. This was achieved by calculating the tonne-kilometres according to the quantity of input applied to or consumed on the feedbase considered.

For each journey, *i*, made by an input in its transportation to the farm:

$$tkm_i = \frac{Input}{1000} \times T_i \qquad (3.26)$$

Where:

tkm_i= tonne-kilometres associated with the transport of the input for journey *I* (tkm/yr)

Input= quantity of the input consumed on or applied to feedbase (kg/yr)

T_i= distance travelled to transport input for journey *i* (km)

To calculate tkm, each input quantity must be converted into kg. This is conducted in the input's respective Framework component (i.e. liquid fertiliser is converted to kg in the "Plant chemical" Component).

Step 2: Calculation of the GHG emissions (kg CO_2-e) associated with the transportation of input to the farm

a) The annual emissions associated with each journey (E_i; kg CO_2-e/yr) are calculated separately, to account for the different EFs which may be required.

$E_i = tkm_i \times EF_i$ (3.27)

Where:

E_i= annual emissions resulting from the transportation of the input for journey *i* (kg CO_2-e/yr)

tkm_i= as calculated in equation 3.38 (tkm)

EF_i= transport EF for journey *i* (kg CO_2-e/tkm)

b) The emissions from each journey were then totalled to obtain total emissions arising from the transport of the input from point of manufacture to the farm.

3.3.4.5 Soil emissions

The "Soil emissions" component calculated all direct and indirect emissions from agricultural soils. Direct sources included nitrous oxide emissions from the application of N fertilisers, from livestock excreta and from crop or pasture residue. Indirect sources included nitrous oxide emissions from leaching and runoff and from atmospheric deposition of N, along with carbon dioxide released via urea hydrolysis and following the application of lime.

The following Sections outline the approaches and assumptions followed in this study for the calculation of each of these sources. Figure 3.4 highlights the relationships between the "Soil emissions" component and other components of the Frameworks. Appendices C and D provide further information regarding the required information and example calculations.

3.3.4.5.1 Nitrous oxide emissions from N fertiliser application

Nitrous oxide emissions from the application of N fertiliser to pasture and crops were calculated using the 2018 NIR methodology and default EF. Emissions from N fertiliser applied to income crops were excluded. The calculation steps are as follows.

1. Calculation of mass of N applied (kg/yr) to each feedbase, determined by the total mass of fertiliser applied (kg/yr), the N content of the fertiliser (%) and the area of the feedbase (ha).
2. Calculation of N_2O-N emissions by applying the EF of 0.002 to the mass of N fertiliser.
3. Conversion of N_2O-N to N_2O emissions (kg/yr) using a factor of 44/28.

3.3.4.5.2 Nitrous oxide emissions from excreta deposited on paddocks

Nitrous oxide emissions following deposition of excreta on paddocks by livestock were calculated following the 2018 NIR methodologies and default EFs. In the present study, emissions from excreta were considered on all feedbases, including income

crop stubble. This was because livestock emissions on crop stubble are considered within the system boundary.

1. Calculation of total faecal and urinary N deposited (kg/yr) for each stock class on each feedbase, as calculated in Section 3.3.4.3.
2. Calculation of N_2O-N emissions by applying the default EF of 0.004 to the mass of N deposited.
3. Conversion of N_2O-N to N_2O emissions (kg/yr) using a factor of 44/28.

3.3.4.5.3 Nitrous oxide emissions from pasture and crop residue

The release of N to soil during the decomposition of plant material from crop stubble and pasture results in nitrous oxide emissions. These emissions were calculated IPCC methodology as recommended by the 2018 NIR. Emissions from decomposition of income crop stubble were excluded.

Prior to 2013, the NIR recommended the inclusion of calculations for nitrous oxide emissions from N-fixing crops and pasture, despite the failure of studies to demonstrate that biological N-fixation caused any significant emissions (Barton et al., 2011; Rochette & Janzen, 2005) in Australia and the removal of this emission source from IPCC (2006) guidelines. At that time the NIR also ignored emissions resulting from the decomposition of non-leguminous pasture and the contribution of below-ground plant matter decomposition. A difficulty of using NIR data to calculate the N output from decomposition and N-fixation was the lack of specific data for the types of crops and pastures considered in this study (i.e. lupins, annual grasses, annual legume pasture). Furthermore, the inputs required to calculate these emissions, such as carbon mass fraction and elemental N:C ratio, were not readily available in literature, despite extensive searches. A methodology was developed to account for the above and acknowledged the concerns raised by other Australian studies of the Australian methodology (Brock et al., 2013; Brock et al., 2016; Thamo et al., 2013). The methodology developed incorporated IPCC approaches for crop and annual pasture decomposition, using Australian-specific values. Emissions from the decomposition of N-fixing crops and pastures were calculated using the approach developed by Unkovich et al. (2010b) and adopted by Brock et al. (2013).

Subsequent to the incorporation of this alternate methodology in the Frameworks, the NIR removed emissions from N-fixing crops and pasture and also amended the methodology for calculating emissions from crop residue decomposition to reflect the IPCC methodology (to include below-ground residue and pasture). The methodology developed for the Frameworks was thus amended to this new approach, following the

overall preference of the study to follow country-specific methodology as it was not considered to interfere with the study goal and scope. The calculation steps for crop and pasture residue are outlined below.

1. Calculation of the mass of N returned to soils (kg/yr) for each feedbase, determined by the amount of stubble or pasture remaining after grazing (kg/ha; as calculated in Section 3.3.4.1), the ratio of root mass to shoots and leaf mass, the N content of roots (%), the N content of shoots and leaf matter (%) and the total area (ha) of the feedbase considered.
2. Calculation of N_2O-N emissions by applying the EF of 0.01 to the mass of N returned to soil.
3. Conversion of N_2O-N to N_2O emissions (kg/yr) using a factor of 44/28.

The 2018 NIR recommends that less-intensively managed pastures emissions are multiplied by a factor of 1/30. The present study follows these recommendations in the absence of an alternative, but also acknowledges the high uncertainty regarding this approach as it does not consider the multiple factors that may influence this value.

3.3.4.5.4 Nitrous oxide emissions from leaching and runoff

The first indirect nitrous oxide pathway considered was the leaching and runoff from land of N from fertilisers, livestock excreta and from pasture and crop residues. IPCC (2006) also considers leaching and runoff from resulting from sewage application and from N mineralised after a loss of soil C, however, these are not applicable to this study.

The present study followed the methodology and EFs of the 2018 NIR. Leaching was considered to occur where the ratio of evapotranspiration to annual rainfall (Et/P) was <0.8 or >1.0. In the Frameworks, annual rainfall and evapotranspiration (obtained from SILO weather files and GrassGro, respectively) were inputted and emissions from leaching and runoff calculated dependant on the ratio obtained. The process followed in outlined below.

1. Calculation of mass of N fertiliser (kg/yr) lost through leaching and runoff on each feedbase, determined by mass of N fertiliser applied (kg N/yr), the fraction of N available for leaching and runoff (FracWET= 0.508 for pasture and 0.223 for crops), fraction lost (FracLEACH= 0.300 for pasture and crops).
2. Calculation of mass of excreta N (kg/yr) lost through leaching and runoff on each feedbase, determined by total mass of excreta deposited (kg/yr), fraction of N available for leaching and runoff (FracWET= 0.510 for sheep and 0.826 for beef cattle), fraction lost (FracLEACH= 0.300 for sheep and beef cattle).

3. Calculation of mass of pasture and crop residue N (kg/yr) lost through leaching and runoff on each feedbase, determined by mass of pasture or crop residue N (kg/yr), fraction of N available for leaching and runoff (FracWET= 0.508 for pasture and 0.223 for crops), fraction lost (FracLEACH= 0.300 for pasture and crops).
4. Calculation of N_2O-N emissions of each source by applying the EF of 0.0075 to the mass of N lost through leaching and runoff.
5. Conversion of N_2O-N to N_2O emissions (kg/yr) using a factor of 44/28.

3.3.4.5.5 Nitrous oxide emissions from atmospheric deposition

The second indirect source of nitrous oxide occurred when N applied to agricultural soils was volatised as ammonia and oxides of N, which are then deposited onto soils and water bodies (IPCC, 2006). The country-specific methodology in the 2018 NIR was followed.

1. Calculation of mass of N fertiliser (kg/yr) volatised on each feedbase, determined by mass of N fertiliser applied (kg N/yr) and fraction volatised (FracGASF= 0.1).
2. Calculation of mass of excreta N (kg/yr) volatised on each feedbase, determined by the mass of excreta N deposited (kg/yr) and fraction of this amount volatised (FracGASM= 0.2).
3. Calculation of N_2O-N emissions of each source by applying the EF of 0.002 for N from fertiliser and 0.004 for N from animal waste to the fraction volatised.
4. Conversion of N_2O-N to N_2O emissions (kg/yr) using the factor 44/28.

3.3.4.5.6 Carbon dioxide emissions from urea hydrolysis

The application of urea to agricultural soils also results in the release of carbon dioxide and represents the carbon dioxide fixed during the industrial production process which is then released into the atmosphere through hydrolysis in the soil (IPCC, 2006). As for the other soil emission sources, the 2018 NIR methodology and default EF was followed. Emissions from income crops were included. The calculations included both the application of urea alone and when urea was supplied as a component of a fertiliser. The calculation process was as follows:

1. Calculation of CO_2-C emissions, as determined by mass of urea (kg/yr) applied to each feedbase applied and the EF of 0.2.
2. Conversion of CO_2-C to CO_2 emissions (kg/yr) using a factor of 44/12.

3.3.4.5.7 Carbon dioxide emissions from liming

Carbon dioxide is a by-product of liming; the process by which either lime or dolomite is applied to agricultural soils to reduce acidity and improve plant productivity. The calculation of these emissions followed the 2018 NIR. The NIR methodology altered from the initial development of the Frameworks, and so they were updated accordingly.

As lime was applied to soils intermittently, the annual amount of lime applied was calculated as total lime applied divided by the liming interval (i.e. mass lime/5 years). The application of lime to income crops were excluded from this study. The calculations, followed the steps outlined below:

1. Calculation of CO_2-C emissions, as determined by mass of lime applied (kg/yr, average) to each feedbase, the fractional purity (limestone= 0.90 and dolomite= 0.95) and the EF (lime= 0.12 and dolomite= 0.13).
2. Conversion of CO_2-C to CO_2 emissions (kg/yr) using a factor of 44/12.

3.3.5 Impact assessment

The third phase of the LCA approach, the impact assessment, aims at understanding and evaluating the magnitude and significance of potential environmental impacts for a product through its life cycle (ISO, 2006a, 2018). At this stage the results of the collated inventory are converted into meaningful indicators specific to the impact category under examination.

This study examined one impact category, “GHG emissions”, also referred to as “climate change”. For animal and soil emissions, inventory calculated end values were presented as individual GHGs (i.e. carbon dioxide, methane and nitrous oxide), while the impact assessment then converted these into the impact category indicator, or FU, (kg CO_2-e/kg LW produced for sale). This was achieved by applying GHG emissions characterisation factors, known as global warming potentials (GWP). As reported in the IPCC AR5, the three primary GHGs; carbon dioxide, methane and nitrous oxide were converted into CO_2-e using the 100-year GWPs of 1, 28 and 265, respectively (IPCC, 2014b).

For pre-farm emissions (Section 3.3.4.4), the inventory calculated total quantities for each input (i.e. kg fertiliser applied or L diesel consumed on-farm). For these emission sources, the impact assessment converted these absolute values directly to kg CO_2e/kg LW produced for sale. To do this, EFs for each input were sourced from Simapro databases (PRé Consultants, 2014). Wherever possible, EFs were sourced from Australian databases such as the Australian Life Cycle Inventory Database

(AusLCI; (ALCAS, 2018). Where this was not possible EFs from international databases, such as EcoInvent V3 (Wernet et al., 2016), were applied. Where an input comprised multiple elements, such as chemicals or supplementary feed products, product information was used to determine the relative contribution of each element and an EF sourced for each. For example, to calculate the EF of mineral blocks supplied to livestock, EFs were sourced for each ingredient and a weighted EF obtained by adjusted these for the relative proportion of each ingredient. For chemicals such as livestock vaccinations or multi-ingredient herbicides, the active ingredients were identified and EFs sourced for each, before calculating a weighted EF according to the proportion of active ingredient in the chemical.

In the Frameworks, the impact assessment stage was conducted within the "Impact assessment" component, involving the following steps;

1. Prior to the conversion of farm emissions to CO_2-e, the inventory input or process values calculated through the Frameworks were automatically populated into the "Impact assessment" worksheet. These were grouped into emission source sub-categories (i.e. herbicides, pesticides, fertilisers) within the key emission sources (i.e. plant chemicals).
2. There were two ways by which an inventory value was converted to CO_2-e:
   a) If one emission type was produced (i.e. methane from the enteric fermentation, nitrous oxide from crop residue decomposition) it was converted to CO_2-e using the applicable GWP.
   b) If multiple emissions were associated with an inventory value (i.e. the pre-farm manufacture of a fertiliser used on-farm produced numerous emissions) then an EF which accounted for all emissions was applied to convert quantity of input applied directly to CO_2-e. These EFs were sourced from databases in SimaPro software.
3. Continuing the approach adopted throughout the Frameworks, along with obtaining the total emissions produced by each emission source, emissions were allocated across feedbases according to usage.
4. Finally, the calculated CO_2-e values were allocated between co-products (i.e. wool, meat) if applicable. See the following Section.
5. The final results were then automatically populated in summary tables for the interpretation stage outlined in Section 3.3.7.

3.3.6 Handling co-production

Several of the livestock systems examined in the present study produced both primary and secondary products. This added a layer of complexity to the carbon footprint analyses as the calculated environmental impact of the system had to be allocated between these co-products. For example, at one enterprise the farmer ran Merino sheep for their wool with meat as a by-product, SAMM sheep for their meat with wool as a by-product and also produced income crops. Depending on the perspective taken, the various stock classes sold can also be considered as co-products. For example, the sale of cull ewes in the prime lamb production systems, or steers, heifers and cull cows in the bull-calf production system. However this was not conducted in the present study.

To address co-products, the present study adhered to the hierarchy of the ISO 14044 and ISO 14067 guidelines (ISO, 2006b, 2018) when handling co-production. That is:

Avoid allocation through;

1. Subdivision of the system; or
2. System expansion (expanding the system to include additional functions related to the co-products)

If allocation cannot be avoided then;

3. Allocation on the basis of a physical or biological relationship (i.e. mass, energy or protein); or
4. Another form of allocation, most commonly economic allocation.

Where possible methods used in recent Australian livestock studies were employed. The following Section outlines the specific approaches adopted.

3.3.6.1 Division into subsystems

In the present study, where farms comprised mixed enterprises (both income crops and livestock), the farm was first divided into income crop and livestock subsystems. This was achieved by attributing inputs and outputs specific to the subsystem which used them. For example, chemicals applied to income crops were assigned to the crop subsystem. If livestock grazed the stubble and remnant grain of these income crops, the animal emissions resulting from the stubble grazing are assigned to the livestock subsystem while the emissions from the production of the crops were attributed wholly to the crop subsystem. This is because following crop harvest, stubble and remnant grain are waste products of grain production. However, the weed control provided by sheep grazing stubble is simultaneously beneficial to the cropping

system, the net result being a negligible contribution by either subsystem (Cottle & Cowie, 2016; Wiedemann et al.). By contrast, where both income crops and crops for the purpose of on-farm supplementary feed were produced, inputs were divided between the crop subsystems and livestock subsystem accordingly. The Frameworks were designed in such a way that this division of subsystems occurs automatically, therefore avoiding the requirement for allocation at this initial stage.

3.3.6.2 Allocation of co-products

The next stage involved the handling of co-products, such as wool and meat, produced in the livestock production system. This was only applicable to the sheep systems in this study, as the beef cattle enterprises produced meat only.

The sheep production subsystem could not be divided as both wool and meat were produced by sheep and significant sources of farm income. The following option, as recommended in the ISO 14044 and 14067 guidelines, is system expansion. System expansion is preferred over allocation in the guidelines due to its ability to also consider effects of changes to supply and demand on co-product production in a way that partial analysis of a system from allocation cannot (Weidema & Schmidt, 2010; Wiedemann et al., 2015a). However, system expansion requires extensive data, complex modelling, a detailed knowledge of supply and demand forces with regards to substitution products and has a greater risk of error due to increased assumptions (Curran, 2007; Mackenzie et al., 2017). For these reasons, it was considered beyond the scope of this study's workload. Other studies have made the same decision (Brock et al., 2013; Casey & Holden, 2005; Ripoll-Bosch et al., 2013).

The next preferred option for partitioning impacts is allocation based on a physical or biological relationship. As proposed in Wiedemann et al. (2015a), allocation based on protein mass was adopted in this study. Termed protein mass allocation (PMA), this approach recognises that wool production is largely driven by protein intake not energy. Thus, while biophysical approaches using energy requirements are suitable for allocating impacts between milk and meat, which are both driven by energy input, this is not the case for wool. Wiedemann et al. (2015a) found that PMA provided similar results to the more complex biophysical allocation using protein requirements. Recent Australian (Cottle & Cowie, 2016; Cottle et al., 2016; Wiedemann et al., 2015a) and international (Dougherty et al., 2018; LEAP, 2015b) publications have used PMA when handling co-products. Following these studies, an allocation factor for wool (PMA_{wool}; %) using PMA was calculated through the following equation:

$$PMA_{wool} = \frac{P_{wool}}{(P_{wool} + P_{LW})} \quad (3.28)$$

Where;

PMA_{wool}= allocation factor for wool (%)

P_{wool}= protein content of wool, presented as a function of the protein content of clean wool on a dry matter basis (100%), the dry matter content of clean wool (84%) and the clean wool yield (%, obtained from farm data).

P_{LW}= protein content of liveweight, 18% per Sanson et al. (1993).

This formula was then rearranged to determine the allocation factor for liveweight (PMA_{LW}; %).

3.3.7 Life cycle interpretation

The final phase of the LCA approach or in the case of this study, the carbon footprint analysis, is the interpretation of results. This stage is essential for the analysis and interpretation of outcomes, with conclusions drawn in accordance with the original goal and scope of the study (ISO, 2006a, 2018). It also involves error checking and "hotspot" identification. In the present study, interpretation occurred at each stage of the project; from initial analyses conducted on the case study farms, to the assessment of potential mitigation strategies (Figure 3.1*)*. The following Sections detail the tasks involved in the interpretation stage and outline how each was achieved in this study.

3.3.7.1 Accuracy and robustness checks

A key role of interpretation is to ensure the accuracy and robustness of input data and the functioning of the model. To achieve this, consistency checks were undertaken to ensure that assumptions, theoretical approaches and methods were consistent with the goal and scope of the study. Completeness checks were undertaken to ensure that all input data was of high quality and level of detail, with particular attention directed at processes or stages in the product life cycle that contribute the most to the carbon footprint results. The above checks were conducted in accordance with LEAP (2016). They ensured that the analyses undertaken in this study were in fact iterative processes which strove for the continual improvement of the quality and accuracy of the carbon footprint results.

Checks were conducted throughout Framework development, during construction of each farm inventory and at the completion of each impact assessment. An iterative process, the motivator was to achieve the research goal and objectives. These checks ranged from as detailed as checking each data input and component of the model

against results, to high-level project decisions. For example, as described previously, after initial carbon footprints were generated using existing GHG calculator tools, it became obvious that the goals of the study would not be met. This led to the development of the Frameworks. Another example was the use of GrassGro to generate site-specific monthly input data rather than using annual or seasonal regional or national data. This followed the recognition that the key determinant of enteric methane, the largest contributor to the carbon footprint of livestock systems, was feed intake, which in turn was determined by feed and animal attributes. Incorporating GrassGro allowed for detailed data upon which mitigation strategies could be tested.

3.3.7.2 Hotspot identification

One of the most valuable outputs of the whole-farm analysis enabled by the LCA approach is the identification of "hotspots". That is, the inputs, processes or stages within the livestock life cycle that contribute the most to the carbon footprint. The conversion of individual GHG emission outputs (i.e. carbon dioxide, methane or nitrous oxide) into a common indicator (i.e. CO_2-e) during the impact assessment enables the direct comparison of the contribution of all emission sources to the carbon footprint.

In the present study, once a hotspot was identified in the overall carbon footprint, it was then possible to examine that hotspot in greater detail by breaking into down into sub-stages. This enabled the identification of the exact input or process responsible for this impact. In the developed Frameworks, the results of the impact assessment were automatically populated into summary tables which grouped emissions into categories of interest. For example, pre-farm and on-farm emissions, emissions according to feedbase, emissions according to stock class, as well as emissions by category (i.e. plant chemicals, enteric methane, transportation). This allowed a comprehensive examination of the livestock enterprise and identification of hotpots at multiple scales and levels of detail. Where a hotspot was identified, it was then possible to refer to the results detailed on the "Impact assessment" component to see a breakdown of emissions sources according to input or process type. This made it possible to adopt a targeted approach when identifying and examining the effect of potential mitigation strategies, but to also examine the effect at farm-scale.

3.3.7.3 Conclusions and recommendations

The final step of the interpretation stage of the carbon footprint analyses was to draw conclusions from the analysis and to provide necessary recommendations. Chapters

Five and Six present conclusion of the individual analyses while, final conclusions and recommendations of the present study are presented in Chapter Seven.

3.4 Simulation of pasture systems

This Section outlines the simulations of the annual grazing systems of the livestock production systems conducted in the present study, as modelled using the GrassGro biophysical model (Donnelly et al., 1997; Freer et al., 1997; Moore et al., 1997).

3.4.1 Historical weather data

Site-specific daily weather data were obtained from the SILO database (https://legacy.longpaddock.qld.gov.au/silo/; (Jeffrey et al., 2001) using farm coordinates. Other Australian studies have utilised a baseline period of between 30 and 50 years to account for inherent climate variability (Bell et al., 2013; Brock et al., 2013; Doran-Browne et al., 2015; Eckard & Cullen, 2011; Ghahramani & Bowran, 2018; Harrison et al., 2014a; Moore & Ghahramani, 2013). Following these studies, a 30-year period (1985-2014) was adopted for all simulations.

This historical weather data had four purposes in the present study:

1. To simulate the baseline annual grazing systems in GrassGro for each of the case study farms.
2. Through the use of annual rainfall and evapotranspiration data in the Frameworks, to determine whether leaching, and therefore indirect nitrous oxide emissions from leaching and runoff, occurred at each farm.
3. To obtain long-term monthly averages for rainfall, temperature, evaporation and radiation. This was achieved through the manipulation of the daily data output using Excel pivot tables.
4. To simulate changes to the baseline case study farms during the investigation into the impact of selected farming practices on the carbon footprint.

3.4.2 Simulation of baseline pasture systems

To simulate the baseline annual pasture systems of the case study farms, information obtained directly from the enterprises was inputted into the respective parameters in GrassGro. Where farm-specific information was not available, parameters were completed using relevant information from literature or from GrassGro defaults. Such information included site-specific climatic data from SILO, soil data, pasture characteristics, livestock characteristics, livestock management and grazing management.

For each farm, pasture was initialised and then simulated with 30 years of historical weather data (1985-2014) sourced from the SILO database. The pasture quality and growth rates generated in these simulations were compared to regionally appropriate data and farm information before being determined as acceptable. The information generated from these simulations was then inputted into the Frameworks to calculate carbon footprint for the baseline scenarios.

3.4.3 Simulation of grazing systems in the analysis of potential mitigation strategies

In the examination of each practice for mitigation potential in the present study, changes to the livestock production systems were simulated in GrassGro to reflect the anticipated changes to the enterprise. These changes were made to the baseline scenarios developed in Section 3.4.2 and simulations run using the same historical SILO weather data applied during the simulations of the baseline scenario. Information regarding the selection and examination of potential mitigation strategies can be found in the following Section.

3.5 Examination of potential mitigation strategies

As is evident in the preceding sections of this Chapter, livestock enterprises are highly integrated systems with many complex interactions between climate, soil, animals, pasture, crops and applied inputs. Along with the detailed carbon footprint analyses, the approach adopted in this study enabled the examination of the influence of strategies with mitigation potential on the carbon footprint. The following Sections outline the process followed during the identification, selection and assessment of potential mitigations.

3.5.1 Selection of potential mitigation strategies

A specific approach was adopted by the present study to account for some of the often-overlooked factors preventing the on-farm adoption of mitigation strategies. Along with the adoption of perennials, the farms in the present study had adopted a number of better practice management practices, some enabled by the increased productivity of their enterprise afforded by perennials, others driven by profit or other productivity motivators. Each farm had successfully overcome any potential barriers to adoption of such practices and had implemented them on their farm successfully. Importantly for the present study, some of these implemented practices were also those recommended as strategies with mitigation potential in existing research

Strategies were chosen for further examination during the carbon footprint analyses

of the case study farms. The Frameworks enabled comparisons to be drawn between farms and emission sources, but also between feedbases (i.e. perennial, annual, crop or feedlot), stock classes and months of the production year. This detailed breakdown of results allowed the targeted identification of farm practices influencing the carbon footprint. Identified practices employed by the case study farms which were also recommended in literature as potential mitigation options were selected for closer analysis. The examination of strategies already implemented by an enterprise is an approach that has also been adopted by other studies (Hyland et al., 2016; Nieto et al., 2018; Veysset et al., 2014). In each of these studies, it enabled the identification of practices already implemented by farmers for non-mitigation purposes such as increased farm productivity, which also had a quantified impact on the carbon footprint of the enterprises. This alignment between adaptation and mitigation is one of the premises of climate-smart agriculture and deemed an appropriate approach to identify regionally appropriate mitigation strategies for consideration by other livestock enterprises in south-western Australia.

3.5.2 Assessment of potential mitigation strategies

Two approaches were adopted to model the impact of a practice with mitigation potential on the carbon footprint of a livestock production system.

The first approach was applied to examine the impact of a practice already implemented on a case study farm. The impact of the practice on the whole-farm carbon footprint was examined by modelling the livestock system without the practice and was particularly valuable when examining the impact of the perennial pasture systems at each enterprise. Such an approach has been applied by other Australian whole-farm studies (Cottle et al., 2016; Taylor et al., 2016) and enables the impact of the practice to be investigated through a comparison of the baseline and the modelled scenario. In the present study, the scenario without the investigated farm practice would be modelled such that the same level of annual saleable liveweight production was maintained. This assumption was made following that gross margins, and thus the productivity, of an enterprise are key farm drivers. The carbon footprints of the “baseline” and the modelled scenario without the selected practice were then compared through the changes to net GHG emissions (CO_2-e) and emissions intensity (EI; CO_2-e/kg saleable liveweight produced). For example, when considering the effect of accelerated joining, the required changes to the enterprise under annual joining to produce the same quantity of liveweight as accelerated joining would be modelled (i.e. increased breeding herd numbers, increased supplementary feed). Following this, the resultant change to net GHG emissions and EI would be examined.

The second approach involved the modelling of the impact of implementing a selected practice on enterprise where it was not already employed. There were instances where a practice with mitigation potential was employed by only one or two of the case study farms. Along with modelling scenarios whereby those enterprises had not employed that strategy as described in the first approach, the effect of implementing the practice was also modelled on those farms who had not adopted the practice. This provided an indication across the enterprises of the respective impact of the practice as a mitigation strategy. This was deemed an appropriate approach given that the present study applied case study farms rather than modelled systems which can be initiated and simplified to suit the purposes of the analysis.

For each potential mitigation strategy, regardless of the approach adopted, the theoretical carbon footprint was compared against the baseline carbon footprints and the effectiveness of that strategy in reducing net emissions and EI quantified. For all simulations investigating the influence of such strategies, pasture groundcover was not permitted to fall below annual threshold levels (70%). This is considered good and common practice to avoid soil erosion and has been applied to other whole-farm studies examining mitigation strategies in southern Australia (Cottle et al., 2016; Harrison et al., 2014b; Moore, 2012). This, along with any additional supplementary feed requirements under the revised scenario, acted as additional measures to the carbon footprint, upon which the viability of a strategy could be assessed.

A limitation of the developed Frameworks was that they did not have the capacity to automatically calculate changes to livestock growth rates and liveweights following changes to feedbase attributes. The supplementary feed calculations did account for such changes, calculating the changes to supplementary feed requirements following such changes. This limitation meant that the effect of certain practices on the growth of livestock, for example, could not be examined. However, it could model the additional supplementary feed required to maintain the growth rates and turnoff requirements of the base scenario. This was assumed to serve as a proxy measure of the effect of applying certain strategies. Into the future, the Frameworks could be enhanced to include this additional functionality, enabling the examination of the effect of a practice in either accelerating or slowing the production of livestock and the ensuing effect on the whole-farm carbon footprint.

3.6 Conclusion

This Chapter outlined the methodology developed to address the goal of this study. To address the goal, it was necessary to integrate methodological approaches and

tools, such as LCA and biophysical models, in an innovative manner which allowed comprehensive analyses to be conducted.

The foundations of the methodological approach were the Sheep and Beef Cattle Frameworks developed to calculate the carbon footprint of livestock production systems located through south-western Australia. Importantly, whilst these Frameworks enabled a whole-farm analysis of a livestock enterprises, they were also developed to permit in-depth intra-farm analyses, comparing pasture systems, stock classes and monthly variations through the production year. It was this capability of the model that allowed two components of the study goal to be met; “the investigation of different pasture systems” and “the identification of regionally appropriate mitigation strategies”.

The data collection stage of this study was conducted to maximise the accuracy and detail of data used in the ensuing analyses. As such this stage included the selection of case study farms which met the research criteria, in-depth farm interviews, the inclusion of farm-specific pasture data derived from the biophysical modelling software GrassGro, and emissions information from SimaPro. This stage is presented in Chapter Four, where the inventories of each enterprise are detailed and initial analyses conducted. The following stage involves the carbon footprint analyses of the four livestock production systems, enabled by the development of the Frameworks in the first stage of the methodology and the data collected in the previous stage. Chapter Five presents and analyses the calculated carbon footprints in detail. The final methodological stage, the examination of farm practices with mitigation potential, was developed in such a way to best address the final objective of this study and to highlight the importance of farming practices already implemented by farmers in south-western Australia for productivity purposes and the potential for wide-spread adoption in a dual role with mitigation. The outcomes of this approach are presented in Chapters Six and Seven.

It is important to consider the limitations of a modelling approach such as the present study. In the absence of measured farm-specific data for all inputs (i.e. monthly pasture attributes and individual stock class characteristics), there will always be uncertainty. There is also uncertainty associated with GHG emission calculation methodology. However, every attempt has been made to address and reduce this uncertainty. Overall, the integrated methodological approach developed presents a detailed picture of the diverse carbon footprint of livestock production in south-western Australia, the role of different farming practices in the emissions output of an enterprise and how to maintain productivity in a carbon-constrained future.

4 INVENTORY DEVELOPMENT OF THE EXAMINED LIVESTOCK PRODUCTION SYSTEMS

4.1 Introduction

This Chapter presents the inventories for the pre- and on-farm activities of the four case study farms analysed in this study. Farm information required by the Frameworks, including livestock, pasture, crop and feedlot characteristics, are summarised. The calculated inventory outputs generated using this information, such as animal and soil emissions, along with emissions resulting from on-farm activities are presented. These outputs are converted to carbon dioxide equivalent (CO_2-e) in Chapter Five to generate the carbon footprint of each enterprise.

In recognition of the extensive data collection requirements of the livestock carbon footprint analyses conducted in this study, the inventory inputs for each case study farm are presented in this Chapter in sub-sections entitled; farm overview, livestock information, pasture information, crop information (where applicable) and grazing management. Every input applied in the carbon footprint calculations is included in tables in the relevant sub-section. In some instances, such as the presentation of chemical inputs and input transportation, input data is summarised to avoid lengthy presentation of data. Each input type, in the context of the farming system under consideration, is explained in the text, outlining justifications for inclusion where necessary, along with relevant sources where it was not sourced directly from farm records.

The final sub-section of each case study farm inventory as presented in this Chapter, is the calculation of on-farm inventory outputs associated with on-farm processed. Using the inventory input data outlined in the previous sections these calculations yield values for methane emissions from enteric fermentation and manure, nitrous oxide emissions from excreta and various other soil emission sources, and carbon dioxide from liming or urea hydrolysis. The presentation of these calculated on-farm outputs is accompanied by preliminary analyses and interpretations drawn by applying the input and farm management information presented in the preceding sub-sections.

The goal of this Chapter is not only to present the inventory information of each farm prior to the subsequent carbon footprint analyses, but to also provide context for the adopted decision-making processes and farm practices. Decisions made by farmers

are influenced by many factors, such as economic or environmental. Understanding the rationale behind these decisions is important for the ensuing carbon footprint analyses and identification of strategies with mitigation potential. For example, perennial species in the pasture system may increase the productivity of the enterprise and alleviate environmental issues such as soil degradation but this may not necessarily translate to an improved carbon footprint. Further to this, if the scope of the mitigation analysis conducted in Chapters Six and Seven is broadened beyond the enterprise to regional levels, the applicability of strategies in different contexts needs to be understood. The interactions between the multiple factors influencing farmer decisions are complex and this study attempts to consider these factors throughout.

Every effort was made to represent each farm accurately and to prioritise farm information over secondary sources. In some cases, provided farm information was insufficient to meet the Framework requirements and so farm-specific assumptions were made or surrogate information sourced from elsewhere. Wherever this occurred, assumptions and clear explanations of the alternate source are provided.

4.2 Inventory of the Bremer Bay sheep production enterprise

4.2.1 Farm overview

The Bremer Bay case study farm[3] was a self-replacing prime Dorper lamb operation located in the Bremer Bay region of southern WA (19°E, 34°S) on a total area of 610 ha. Of this, 410 ha was dedicated to pasture production and 130 ha to crop production for supplementary feed, with the remaining area a mix of farm infrastructure, dams, shelter belts and uncleared vegetation. The climate was temperate; the 30-year (1985-2014) historical minimum and maximum monthly average temperatures at the site were 7°C in July and 27°C in January to February, respectively. Average annual rainfall was 514 mm, with the maximum average monthly rainfall of 64 mm recorded in May and the minimum average monthly rainfall of 19 mm in February. (Figure 4.1, SILO data)(Jeffrey et al., 2001). The average annual growing season was seven months, from May to the end of November. This region also received the highest amount of out-of-season rainfall out of all the regions examined, with 139 mm or 27% of annual rainfall received between December and April.

[3] For simplicity, following the initial introduction of each case study farm it is thereafter referred to by its location.

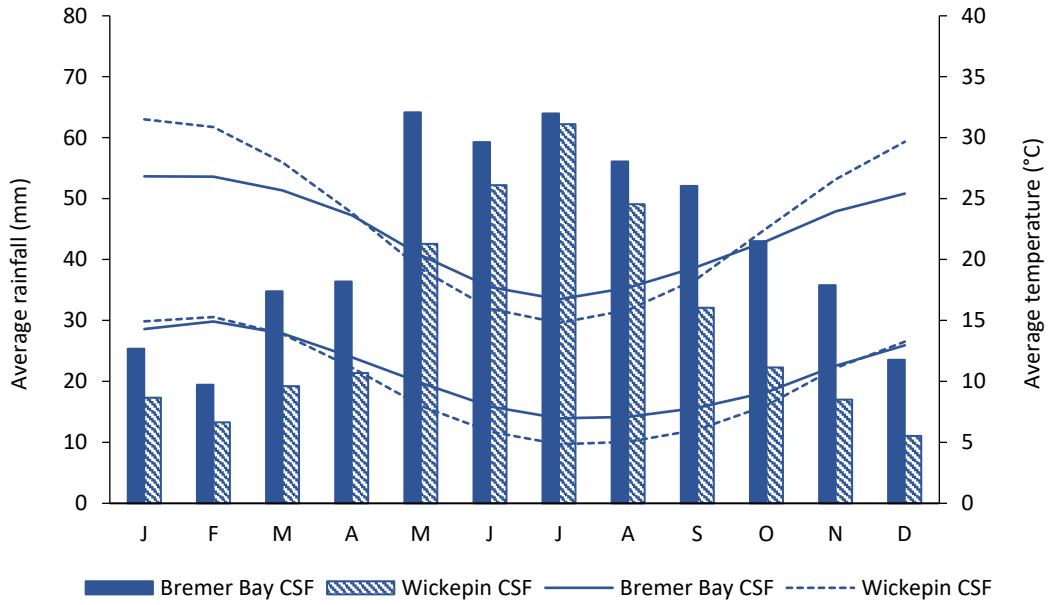


Figure 4.1 - The 30-year (1985-2014) average monthly rainfall (mm), minimum and maximum temperatures (°C) at the two sheep production enterprises.

Where: average monthly rainfall (mm, bars), minimum and maximum temperatures (°C, lines) at Bremer Bay (blue, solid) and Wickepin (blue, striped). Site-specific historical weather data was sourced from SILO (Jeffrey et al., 2001).

The primary product was prime Dorper lambs. Cull breeding ewes and rams were also sold for meat production. As Dorper sheep are self-shedding, considerations of wool production were not required. Dorper sheep are less seasonal breeders, capable of multiple joinings each year. Breeding ewes were thus mated twice a year; in April and October, resulting in two lambing periods, September and March, respectively. The March lambing allowed the farmer to take advantage of the out-of-season high lamb prices. As a self-replacing herd, lambs destined for sale were finished in feedlots for a month prior to sale, while lambs destined for the breeding herd remained on pasture. Cull ewes and rams were sold directly off pasture or stubble.

The grazing system was comprised of two pasture types; annual and perennial grass pasture. The annual pasture system (170 ha) was legume-dominant (60%), with a mix of sub clover (*Trifolium subterraneum*), French serradella (*Ornithopus sativus*) and 40% annual ryegrass (*Lolium rigidum*). Pasture was resown as required to maintain an optimal mix of species. The perennial pasture system (240 ha) was comprised of the subtropical perennial grass kikuyu (*Pennisetum clandestinum*), sub clover, serradella and annual ryegrass. Kikuyu was sown over annual pasture, initially in 2006, and then in 2010. Kikuyu had not been resown since initial establishment. Annual pasture was grazed through winter and spring and rested in the summer-autumn period, while kikuyu was grazed year-round.

Lupin (40 ha) and oat (90 ha) crops were grown for the purpose of supplementary feed for on-farm use. Crop stubble was grazed in the summer months.

In line with common practice in south-western Australia, no irrigation occurred on the Bremer Bay case study farm, or any of the other farms considered in this study.

4.2.2 Livestock information

Dorper sheep are valued for their self-shedding properties that reduce labour costs through the elimination of requirement for shearing, crutching or mulesing (Chadwick & Pearce, 2013). The ability to direct more energy towards liveweight gain means that Dorper lambs grow more rapidly than other sheep breeds such as Merinos (Kleemann et al., 2000; Schoeman, 1990), allowing them to be sold off earlier or at higher liveweights. Dorpers are an early-maturing breed and can be mated as early as seven months as long as they weigh at least 40 kg (Chadwick & Pearce, 2013). As non-seasonal breeders, Dorpers perform well under accelerated joining systems, allowing farmers to benefit from high lambing rates and the production of out-of-season lambs.

The livestock characteristics and management information at Bremer Bay are outlined in Table 4.1. The enterprise's Dorper breeding flock consisted of 800 ewes with a mature weight of 70 kg and 12 rams with mature weight of 80 kg. Annually an average of 150 cull ewes and three cull rams were sold. As a self-replacing flock, 150 replacement maiden ewes were sourced on-farm annually and joined for the first time at seven months at an average liveweight of 50 kg. It was assumed that 75 replacement ewes joined the breeding herd (and thus 75 cull ewes sold) prior to joining in April and the remaining 75 (and 75 culls sold) prior to the October joining. Replacement rams were purchased at an average age of 18 months weighing an average of 65 kg. It was assumed that cull rams were sold at the end of March and replacement rams were purchased at the beginning of April to coincide with the April joining and the growing season. All breeding stock were drenched in November.

Farm-specific weaning rates were 150%, resulting in an average 1,200 lambs weaned annually over the two breeding cycles. As 150 weaner ewes were retained to join the breeding flock, 1,050 weaner lambs were sold annually. Though lambing occurs over number of weeks, for the purposes of this analysis lambing was assumed to occur in mid-March and mid-September. Lambs were marked and vaccinated at eight weeks and then weaned at 14 weeks at a target liveweight of 28-30 kg. After one month on pasture, lambs destined for sale were moved to the feedlots, while lambs destined to become replacement ewes remained on pasture. Lambs remained in the feedlot for one month on average (at a target growth rate of 2 kg/week) before being sold at 22

weeks of age at a target liveweight of 45 kg. Lamb sales were assumed to take place at the beginning of March and September.

Table 4.1 - Characteristics of the Dorper flock and annual veterinary inputs at the Bremer Bay sheep enterprise

Input^	Unit	Ram[a]	Ewe	Replace-ment ewe	Lamb[b]	Total
Breed			Dorper			
Joining date(s)			15 Apr, 15 Oct			
Lambing date(s)			15 Mar, 15 Sep			
Weaning date(s)			1 Jan, 1 Jul			
Stock count	hd	15	800	150	1200	-
Age	months	18->24	>24	4.5-24	0-7	-
LW	kg	60-80	70	35-70	4-45[c]	-
Growth rate[d]	kg/hd/day[e]	0.00-0.12	0.00	0.04-0.21	0.21-0.36	-
			Sale information			
Sale date(s)		31 Mar	31 Mar, 30 Sep	-	1 Mar, 1 Sep	-
Sale count	hd	3	150	-	1050	-
Sale LW	kg	75	70	-	45	-
Total LW sold	kg	225	10500	-	47250	57975
			Veterinary product application			
Vaccination	ml	-	-	-	3627	3627
Drench	ml	720	34500	13500	-	48720
			Veterinary product transportation			
Vaccination	tkm					16
Drench	tkm					379

Hd = Head LW = liveweight tkm = tonne-kilometres

^ Inputs are presented on an annual basis.

[a] Includes mature (>24 months) and replacement rams (<24 months).

[b] Includes lambs sold and lambs destined to become replacement ewes.

[c] Where birthweight of 4 kg was sourced from Schoeman and Burger (1992).

[d] Required growth rates of growing animals to meet farm specifications were calculated from farm-specific information. As detailed in Chapter Three, mature sheep (>24months) were assumed to have a zero net liveweight change.

4.2.3 Pasture information

Annual pasture was first sown in 1975 and at the time of data collection, contained a mix of sub clover, serradella and annual ryegrass. At establishment, paddocks were sown with 5 kg/ha sub clover, 5 kg/ha serradella and 2 kg/ha annual ryegrass. All seed was purchased. It was assumed that the pasture was resown every ten years in the month of April. Prior to sowing, pre-emergent herbicide was applied. At establishment and in May and June annually following establishment, fertilisers containing P, K and Ca were applied. Table 4.2 provides details of the pasture characteristics and inputs applied.

Perennial pasture was first established in 2006 with the introduction of kikuyu

oversown on existing annual pasture. More area was converted in 2010. Just before sowing, annual pasture was sprayed twice in a double-knock approach, with glyphosate and then Spray.Seed 250 herbicides. Kikuyu was then sown at a rate of 1 kg/ha with 4 kg/ha of seconds comprised of sub clover and annual ryegrass. All seed was purchased. At establishment and then every May and June following, fertilisers containing P, K and Ca were also applied to kikuyu pasture.

Table 4.2 - Characteristics and annual inputs of the two pasture types and two supplementary feed crops at the Bremer Bay sheep enterprise

Input^	Unit	Annual pasture	Perennial grass pasture	Lupins	Oats	Total
Predominant soil type		Sand over gravel over clay				
Area	ha	170	240	40	90	540
Years since establishment	yr	10	6	1	1	-
Month sown		Apr	Sep	Apr	Apr	-
Month harvested		-	-	Dec	Dec	-
Grain yield	kg/ha	-	-	1200	2000	-
Sowing rate	kg/ha	1.2	1.0	100.0	85.0	-
Chemical application						
Herbicide						
Glyphosate	l	17	58	40	90	205
Spray.Seed	l	-	48	-	-	48
Simazine	kg	-	-	40	-	40
Fertiliser						
P & K fertiliser	l	935	1440	200	450	3025
Liquid C	l	-	-	500	1125	1625
Ca fertiliser	l	935	1440	200	450	3025
Chemical transportation						
Herbicide						
Glyphosate	tkm	80	271	188	424	963
Spray.Seed	tkm	-	1009	-	-	1009
Simazine	tkm	-	-	428	-	428
Fertiliser						
P & K fertiliser	tkm	411	634	88	198	1331
Liquid C	tkm	-	-	224	505	729
Ca fertiliser	tkm	626	964	134	301	2025
On-farm fuel consumption & transportation						
Tractor diesel	l	154	336	230	480	1200
Harvester diesel	l	-	-	165	372	537
Fuel transportation	tkm	6	14	16	35	71
Farm machinery production						
Tractor	USD	271	580	389	807	2048
Seeder	USD	48	134	112	252	546
Boomspray	USD	172	284	135	228	819
Harvester	USD	-	-	1050	2363	3413

tkm = tonne-kilometres

^ Inputs are presented on an annual basis. Inputs applied at establishment x years ago are also considered in this table by adjusted the input value for the number of years since establishment.

4.2.3.1 Pasture attributes

Pasture attributes specific to Bremer Bay were obtained from multiple sources.

Annual pasture data sources

Monthly annual pasture attributes, including dry matter availability (DMA), growth rates, dry matter digestibility (DMD) and crude protein (CP) content were obtained from GrassGro software by modelling site-specific data over a 30-year period (1985-2014). Currently GrassGro does not include a pasture parameter set for serradella and, despite efforts, no published studies were sourced which had measured the intake by sheep of serradella in a mixed annual pasture in WA's southern region. As such, when developing the farming systems in GrassGro, the serradella pasture was substituted for the sub clover parameter set. This aligns with other studies which observed that the monthly nutritional properties of serradella spp. compared well to sub clover (Hackney et al., 2013). It also follows the approach of other modelling studies which used substitute pasture species in their chosen biophysical models (Doran-Browne et al., 2015; Thomas et al., 2012).

Kikuyu pasture data sources

The GrassGro database has a beta kikuyu parameter set available to users. The kikuyu pasture was initially modelled using this parameter set along with the sub clover set. However, the modelled kikuyu outputs consistently under-estimated DMA, DMD and CP content. This mirrors the findings of Moore (2012) who omitted the kikuyu parameter set from their climate change and adaptation modelling study. Instead, seasonal mixed kikuyu-annual pasture seasonal DMA and monthly DMD and CP values were obtained from trials conducted in Esperance and southern WA (Sanford, per comm 2014; (Moore et al., 2009b).

Calculated pasture attributes

Though the monthly analyses were conducted (Figure 4.2; Section 4.2.6.1), for simplicity feedbase results are presented as averages according to the annual growing and non-growing season of the region (Table 4.3). This was a preferred approach to the adoption of traditional seasonal averages (i.e. spring, summer, autumn and winter) which can yield misleading results due to varying climatic patterns and ensuing pasture production differences across regions. The traditional definitions of the four seasons will need to evolve as the projected impacts of climate change in southern Australia become more evident and historical definitions no longer applicable.

Table 4.3 – Productivity and feed quality attributes of the two pasture types and two crop stubbles grazed at the Bremer Bay sheep enterprise

Input^	Unit	Annual pasture		Perennial grass pasture		Lupin stubble		Oat stubble	
		Mean	SD	Mean	SD	Mean	SD	Mean	SD
Length of growing season		Late Apr – late Nov		Jan - Dec		-		-	
Annual NPP[a]	t DM/ha	8.32	-	11.05	-	-	-	-	-
Growing season NPP[b]	t DM/ha	7.34	-	7.31	-	-	-	-	-
Non-growing season NPP[b]	t DM/ha	0.98	-	3.74	-	-	-	-	-
Available DMA[c] (% green)									
Start of growing season	t DM/ha	1.65 (28%)	*0.00*	2.12	*0.00*	-	-	-	-
Peak growing season	t DM/ha	3.28 (68%)	*1.04*	3.19	*0.54*	-	-	-	-
End of growing season	t DM/ha	4.39 (7%)	*0.00*	3.86	*0.00*	-	-	-	-
Non-growing season	t DM/ha	2.63 (5%)	*2.56*	2.25	*0.10*	3.35	*0.27*	8.15	*0.50*
DMD[d]									
Start of growing season	%	75	*0.00*	72	*0.00*	-	-	-	-
Peak growing season	%	71	*2.56*	71	*1.45*	-	-	-	-
End of growing season	%	68	*0.00*	70	*0.00*	-	-	-	-
Non-growing season	%	56	*2.56*	68	*2.00*	52	*3.85*	45	*3.26*
CP content[d]									
Start of growing season	%	27	*0.00*	19	*0.00*	-	-	-	-
Peak growing season	%	24	*1.85*	24	*2.22*	-	-	-	-
End of growing season	%	22	*0.00*	20	*0.00*	-	-	-	-
Non-growing season	%	15	*1.62*	12	*1.78*	7	*0.82*	4	*0.82*

^ With the exception of the annual net primary productivity (NPP) of the pasture, all averages and standard deviations are calculated only from months feedbase is grazed within that period.

[a] GrassGro modelled output for annual pasture and regionally-specific data for perennial (Sanford, per comms 2013 and Moore et al. (2009b).

[b] Where growing season refers to the annual species' growing season at the case study farm location.

[c] GrassGro provides values for both total available DMA and green DMA, the regional perennial data only provided overall values. Lupin and oat stubble DMA calculated from stubble yields (a function of grain yield and harvest index, see Chapter Three) and calculated animal intake (Table 4.6).

[d] Annual pasture DMD and CP content obtained from GrassGro. Perennial pasture attributes sourced from Sanford (per comm 2014) and Moore et al. (2009b) and stubble attributes from DAFWA (2006a).

The GrassGro output revealed that the long-term average growing season for annual pasture at Bremer Bay ran from late April to late-November. The average net primary productivity (NPP) of annual pasture at Bremer Bay was 8.32 t DM/ha/yr, with 88% of this produced during the growing season and only 12% in the non-growing season months from December to April. Kikuyu pasture, by contrast, produced green biomass throughout the year, with an NPP of 11.05 t DM/ha/yr. The summer-active kikuyu, making use of summer rains in this region, produced over one third (38%) over its annual biomass outside of the annual growing season. This meant that while the DMAs of annual pasture and perennial pasture outside of the growing season were

comparable (2.63 and 2.25 t DM/ha, respectively), annual pasture was mostly senescent from late-November onwards (95%). The kikuyu, however, continued to provide green feed.

The high legume component of the annual pasture ensured high quality feed from the beginning of the growing season in April (75% DMD and 27% CP) through the peak growing season (71% DMD and 24% CP). This was also the case for the kikuyu pasture. In winter, kikuyu growth rates declined, allowing the annual component to establish and dominate the pasture mix through to early November (71% DMD and 24% CP). The quality of annual pasture sharply declined in late November, with an average non-growing season DMD of 56% and CP content of 15%. The pasture continued to decline until the following growing season (Figure 4.2a; Section 4.2.6.1), with the DMD and CP content of annual pasture unable to meet the energy requirements of lactating and growing livestock between February and April. By contrast, as the annual species senesced, the growth rates of kikuyu increased, resulting in an average non-growing season DMD of 68%, which remained consistent for this period. The CP content of perennial pasture declined as the annual components senesced, to an average 12%, reflecting the lower CP content of kikuyu.

4.2.4 Crop information

Bremer Bay grew lupin and oat crops for the sole purpose of providing supplementary feed on-farm. Crops were sown in April annually using seed retained from the previous years' crops. Pre-emergent herbicides were applied prior to seeding using the farm machinery. At seeding or soon after, multiple liquid fertilisers or soil ameliorants containing P, K, Ca and C were applied. In most years, harvest occurred in December. All harvested grain was retained on-farm. The average lupin grain yield was 1,200 kg/ha/yr and average oat grain yield 2,000 kg/ha/yr (Table 4.2). The harvested lupins were used in the feedlot ration provided to lambs. The harvested oats were provided to stock during the dry season and in the feedlot ration.

4.2.4.1 Crop attributes

Lupin and oat stubbles were grazed between harvest and sowing in April. The starting DMAs, or stubble yields, of lupin and oat stubble were calculated to be 3.68 and 8.76 t DM/ha (Table 4.3), respectively. Following the methodology described in Chapter Three, these were obtained using farm-specific grain yields and the harvest index of each crop, 0.30 and 0.21 for lupins and oats, respectively (Unkovich et al., 2010a). The starting DMD and CP contents of each stubble were assumed to be 56% and 49% (DMD) and 8% and 5% (CP content) for lupins and oats, respectively (DAFWA,

2006a, 2006b). Monthly stubble DMA, DMD and CP content of both crop stubbles declined during the months grazed by livestock (Figure 4.2, Section 4.2.6.1). The average non-growing season DMAs for lupins and oats were 3.15 and 8.15 t DM/ha, the DMDs were 52% and 45%, while the CP contents were 7% and 4%, respectively. Average values were calculated only from the months stubbles were grazed.

4.2.5 Grazing management

Livestock were rotated between pasture and stubble types through the production calendar as part of the farm's grazing management strategies.

Annual pasture

Livestock grazed annual pasture from April through to the end of December, when pasture quality declined to below that required to meet the energy requirements of multiple stock classes. Supplementary feed was provided in December to all stock on annual pasture. Both hay and oats were provided, along with a mineral lick. The hay was assumed to comprise 20% of overall feed provided, to ensure enough roughage was in the animal's diet to avoid problems such as acidosis (DAFWA, 2006a). The total supplementary feed provided on annual pasture was 22,187 kg/yr (Table 4.4).

Crop stubble

Mature livestock on annual pasture were moved to crop stubble in January after the December harvest, to take advantage of the spilled grain in stubble, while reducing the stubble load. Stock remained on stubble until the end of March when they were moved back onto annual pasture. Immature stock were not moved to stubble in December, rather to kikuyu pasture which provided high quality green feed during this period. Stock grazing stubble were not supplied supplementary feed in January as the combined quality of stubble and spilled grain was quite high. In February, oats were provided and in March, when stubble quality was very low, both oats and hay were provided. A mineral lick to combat the mineral and protein deficient dry feed was provided for the duration of stubble grazing. In total, 15,841 kg was provided annually to stock grazing stubble.

Kikuyu pasture

The kikuyu pasture was grazed all year. Livestock were supplementary fed oats, hay and mineral licks from December through to April, to ensure that lactating ewes were able to meet their requirements (43,605 kg). In addition to the stock already grazing perennial pasture, the September lambs from annual pasture were weaned directly onto perennial pasture in January to take advantage of the green feed. Replacement stock were also moved to kikuyu pasture in January from annual pasture, as younger

stock are less able to persist on lower quality stubble than mature livestock (DAFWA, 2006a, 2006b).

Table 4.4 - Feed quality attributes and annual inputs of the supplementary feed and feedlot ration supplied at the Bremer Bay sheep enterprise

Attribute or input^	Unit	Annual pasture	Perennial grass pasture	Lupin stubble	Oat stubble	Total
			Supplementary feed attributes[a]			
DMD						
Oats	%		73			
Hay	%		59			
Mineral lick	%		-			
CP content						
Oats	%		9			
Hay	%		8			
Mineral lick	%		30			
			Supplementary feed provided[b]			
Oats	kg	16000	29716	3325	9017	58058
Hay	kg	4089	7594	667	1533	13883
Mineral lick	kg	2098	6295	457	842	9692
			Supplementary feed transportation			
Oats	tkm	-	-	-	-	-
Hay	tkm	720	1337	117	270	2444
Mineral lick	tkm	353	1058	77	141	1656
			Feedlot ration attributes[b]			
DMD						
Grain mix	%		79			
Hay	%		59			
CP content						
Grain mix	%		20			
Hay	%		8			
			Feedlot ration provided[b]			
Grain mix	kg					19126
Hay	kg					21612
			Feedlot ration transportation			
Grain mix	tkm					1929
Hay	tkm					-

^ Inputs are presented on an annual basis.

[a] DMD and CP content of feed sourced from industry publications (DAFWA, 2006a; NSW DPI, 2016).

[b] As fed, considering DM content of feed and assumed wastage (20%).

Feedlot

After a month on kikuyu pasture post-weaning, September lambs destined for sale were moved to the feedlot in February for finishing prior to sale in March. During the month in the feedlot, lambs were provided a high-quality ration comprising of a grain mix (80% of total ration) and hay (20% of total ration) to achieve target sale weights. The grain mix included oats, lupins, molasses, urea and a buffer solution, with all ingredients purchased aside from the oats and lupins. The combined grain mix and hay ration had a DMD of 75% and CP content of 18%. The March lambs underwent

the same finishing process in the feedlot in August after being moved from both annual and perennial pasture following weaning. The annual feedlot ration provided over the two lambing cycles totalled 40,438 kg.

4.2.6 Calculated on-farm inventory outputs

The calculated on-farm inventory outputs associated with on-farm processes at Bremer Bay are presented in this Section, accompanied by preliminary analyses. Calculated outputs are presented in Table 4.5 and include methane emissions from enteric fermentation and manure, nitrous oxide emissions from excreta and various other soil emission sources, and carbon dioxide from liming or urea hydrolysis. These inventory outputs are then converted from individual GHG emissions to carbon dioxide equivalent in the impact assessment stage presented in Chapter Five.

As discussed in Chapter Three, unlike on-farm processes, the calculated inventory outputs of pre-farm inputs are presented as quantities (i.e. kg, l; in the case of the production of inputs) or tonne-kilometres (tkm; in the case of the transportation of inputs). The Bremer Bay pre-farm inventory outputs were presented in the earlier Sections of this Chapter and as such are not discussed in this Section. These pre-farm outputs are converted directly to carbon dioxide equivalent in Chapter Five, where they can be analysed alongside the on-farm outputs.

For each on-farm emission source detailed in Table 4.5, the quantity of emissions is presented both as an unadjusted total and according to the functional unit (kg GHG/kg LW produced for sale). Each also displays the contribution from each feedbase, as appropriate. Animal emissions across feedbases are presented using two approaches. Firstly, emissions produced according to total liveweight grazed on each feedbase (i.e. kg GHG/kg LW grazed) and secondly, emissions produced according to saleable liveweight production on that feedbase (i.e. kg GHG/kg LW produced for sale). Both serve as indicators of the emissions efficiency of a feedbase; in terms of its ability to support grazing livestock and in terms of its ability to support the production of saleable product. Per animal emissions, soil and other on-farm emissions from each feedbase are presented in terms of saleable liveweight production and also on a per hectare basis. It was important to include multiple measures as no one approach provided a full picture of emissions efficiency

<u>Enteric methane emissions</u>

Total enteric methane produced at Bremer Bay was 12,592.85 kg CH_4/yr, with 60% produced on kikuyu pasture, 29% on annual pasture, 6% on crop stubble and the remainder in the feedlot. These unadjusted total values are misleading however as

they do not consider area grazed, duration of grazing or intensity of grazing, for example. Considering instead the emissions produced per kilogram of liveweight grazed on each feedbase revealed that the feedlot produced the highest emissions of 2.29×10^{-2} kg CH_4/kg LW grazed, followed by annual pasture and kikuyu pasture, with crop stubble producing the lowest at 1.34×10^{-2} kg CH_4/kg LW grazed. Feed intake, the predictor of enteric methane production, is generally higher when an animal has access to sufficient quantities of high-quality feed. As such, it follows that intake by livestock in the feedlot, provided high quality rations would be high. Similarly, a combination of low feed quality and livestock grazing at maintenance levels (i.e. no lactating stock) would yield lower feed intake values on stubble.

Enteric methane emissions produced per kilogram of saleable liveweight was 2.17×10^{-1} kg CH_4/kg LW produced for sale. A comparison of the feedbases using this metric revealed that instead the feedlot was the most efficient in terms of enteric methane production, producing 5.85×10^{-2} kg CH_4/kg LW and crop stubble the least, producing $3.29\text{-}3.39 \times 10^{-1}$ kg CH_4/kg LW. Similarly, kikuyu pasture was more efficient than annual pasture. This measure highlights the importance of considering the multiple factors affecting emissions output. While livestock grazing feedbases of high quality may produce greater emissions than those grazing low quality feed, these emissions are offset by the increased productivity. Section 4.2.6.1 examines the relationship between enteric methane production, feed intake and feedbase attributes in detail, exploring the factors that produced the above results. This is important for the analyses conducted in Chapter Five.

Manure methane emissions

Manure methane output was a fraction of enteric methane (2.77 kg CH_4/yr). Despite this there was a 25% difference between oat stubble, which produced the most emissions in terms of livestock grazed, and kikuyu pasture which produced the least. This is unsurprising as manure methane is a function of intake and DMD (see Chapter Three), so feed sources with higher DMDs result in lower methane output. Though annual pasture was of high DMD through the growing season, across the production year there was less variability in the quality of kikuyu, resulting in the lower manure methane output. As with enteric methane emissions, the most productive feedbase in terms of saleable liveweight production was the feedlot, followed by annual pasture and kikuyu pasture, with crop stubbles least productive.

Table 4.5 - Calculated annual animal, soil and associated on-farm emissions produced on the Bremer Bay sheep enterprise

Output	Unit	Annual pasture	Perennial grass pasture	Lupins	Oats	Feedlot	Total
			Animal emissions				
Enteric CH_4	kg CH_4/yr	3616.45	7550.24	282.32	529.14	614.70	12592.85
	kg CH_4/kg LW grazed[a]	1.58E-02	1.57E-02	1.34E-02	1.34E-02	2.29E-02	1.58E-02
	kg CH_4/kg LW produced for sale[b]	2.39E-01	2.52E-01	3.29E-01	3.39E-01	5.85E-02	2.17E-01
Manure CH_4	kg CH_4/yr	0.80	1.60	0.09	0.18	0.11	2.77
	kg CH_4/kg LW grazed	3.49E-06	3.32E-06	4.17E-06	4.44E-06	3.97E-06	3.47E-06
	kg CH_4/kg LW produced for sale	5.27E-05	5.35E-05	1.02E-04	1.13E-04	1.02E-05	4.77E-05
			Direct soil emissions				
Excreta N_2O	kg N_2O/yr	38.25	66.24	1.11	1.56	4.30	111.45
	kg N_2O/kg LW grazed	1.68E-04	1.37E-04	5.25E-05	3.94E-05	1.60E-04	1.40E-04
	kg N_2O/kg LW produced for sale	2.53E-03	2.21E-03	1.29E-03	1.00E-03	4.09E-04	1.92E-03
N fertilisers N_2O	kg N_2O/yr	-	-	-	-	-	-
	kg N_2O/ha	-	-	-	-	-	-
	kg N_2O/kg LW produced for sale	-	-	-	-	-	-
Crop residue N_2O	kg N_2O/yr	-	-	29.52	123.89	-	153.41
	kg N_2O/ha	-	-	7.38E-01	1.38E+00	-	1.18E+00
	kg N_2O/kg LW produced for sale	-	-	3.44E-02	7.94E-02	-	2.65E-03
Pasture residue N_2O	kg N_2O/yr	9.09	7.15	-	-	-	16.24
	kg N_2O/ha	5.35E-02	2.98E-02	-	-	-	8.33E-02
	kg N_2O/kg LW produced for sale	6.01E-04	2.39E-04	-	-	-	2.80E-04
			Indirect soil emissions				
Atmospheric deposition N_2O	kg N_2O/yr	7.65	13.25	0.22	0.31	0.86	22.29
	kg N_2O/kg LW produced for sale	5.06E-04	4.43E-04	2.57E-04	2.00E-04	8.18E-05	3.84E-04
			Other calculated on-farm emissions				
Liming CO_2	kg CO_2/yr	-	-	-	-	-	-
	kg CO_2/ha	-	-	-	-	-	-
	kg CO_2/kg LW produced for sale	-	-	-	-	-	-
Urea hydrolysis CO_2	kg CO_2/yr	-	-	-	-	-	-
	kg CO_2/ha	-	-	-	-	-	-
	kg CO_2/kg LW produced for sale	-	-	-	-	-	-

LW= liveweight

[a] To enable comparison across feedbases, emissions were adjusted according to output per kg of liveweight grazed on each.

[b] Using the functional unit, overall emissions were adjusted according to output per kg of liveweight produced for sale on each pasture.

Nitrous oxide emissions from excreta

Excreta nitrous oxide emissions totalled 111.45 kg N_2O/yr, or 1.92×10^{-3} kg N_2O/kg LW produced for sale. In terms of livestock grazed, stock on crop stubble produced the lowest emissions (oats, 3.94×10^{-5}; lupins, 5.25×10^{-5} kg N_2O/kg LW grazed), followed by kikuyu pasture, the feedlot and finally annual pasture (1.68×10^{-4} kg N_2O/kg LW grazed). Considering CP content is a key determinant of excreta nitrous oxide emissions, it follows that stock grazing stubble with low CP content would produce the least nitrous oxide, while those consuming green feed with high protein contents produce more. According to total saleable liveweight production, both pastures produced more nitrous oxide than crop stubbles or feedlot. Of the two, kikuyu pasture was more efficient, producing 13% less nitrous oxide per kg LW sold than annual pasture. Again, this is indicative of the higher CP content of annual pasture during the months grazed than kikuyu.

Other soil emissions

Other nitrous oxide emissions from soil were sourced from crop and pasture residue. Crop residue was a significant source of nitrous oxide and contributed 2.65×10^{-3} kg N_2O/kg LW produced for sale, or 153.41 kg N_2O/yr, 38% more than excreta nitrous oxide emissions. Pasture residue emissions, annualised by a factor of 30 in accordance with the NGGI methodology (DISER, 2020b), contributed only 2.80×10^{-4} kg N_2O/kg LW, or 16.24 kg N_2O/yr. Breaking this down to a per hectare basis, emissions were higher on oat stubble compared to lupin stubble (1.38 and 7.38×10^{-1} kg N_2O/ha, respectively), while emissions were higher on annual pasture as opposed to kikuyu pasture (2.98×10^{-2} and 5.35×10^{-2} kg N_2O/ha, respectively). The quantity of pasture or stubble remaining after grazing, along with the N content of the above- and below-ground residue, determine the nitrous oxide released, providing explanation as to the exhibited differences in residue emissions between feedbases. For example, the stubble yield of oats was more than double that of lupin, which offset any emissions associated with the increased CP content of lupin stubble. The high quantities of pasture residue on annual pasture, coupled with the high proportion of legumes in the sward explained the higher emissions of annual pasture to kikuyu. Comparing the productivity in terms of saleable liveweight production on each feedbase yields the same exhibited trends.

Bremer Bay did not experience leaching or runoff, had no liming program or N fertiliser application and hence produced no emissions associated with these sources.

4.2.6.1 Analysis of the interactions between feedbase, animal intake and enteric methane production at the Bremer Bay sheep enterprise

As described in Chapter Three, enteric methane production by sheep is a function of feed intake by the animal. In turn, feed intake is determined by the availability and DMD of pasture, along with the liveweight and physiological status of the animal (i.e. lactating or dry). The previous Section presented emissions resulting from enteric fermentation as annual averages, however it is valuable to break down these emission averages further to investigate and compare the roles of farm practices. This Section presents the key factors influencing the enteric methane output of Bremer Bay, while Chapter Five explores these in the context of the carbon footprint analysis.

The first component of Figure 4.2 presents the attributes of each feedbase at Bremer Bay. The second presents the average monthly feed intake and enteric methane production by each stock class corresponding to each feedbase. It is evident that feed intake, and thus enteric methane production, varies considerably between; a) the feedbases, b) the stock classes, and c) the same stock class within a production year. The calculated intake and enteric methane production values are presented as annual averages for each stock class in Table 4.6. Though presenting this information as annual or seasonal averages can be useful to draw conclusions, the monthly values presented in Figure 4.2b support this information by demonstrating how management decisions within a production year can influence overall emission output of a livestock production system.

As expected, feedbase intake and enteric methane production was higher across the stock classes during the growing season when pasture quality and availability was higher. As pasture declined in December, so too did animal intake, resulting in the requirement to supplementary feed. The advantage of the summer-active kikuyu is obvious at this stage; supplementary feed comprised an average of 12% of total feed intake as opposed to 21% on annual pasture (Table 4.6). Livestock moved from annual pasture to graze crop stubble required increasing amounts of supplementary feed to meet their daily requirements as stubble quality declined. By the end of the non-growing season in March when lambing began, supplementary feed comprised close to half of daily intake (lupins, 45%; oats, 52%). By contrast, during this period kikuyu pasture, which retained its quality, was largely sufficient to meet the maintenance requirements of livestock. Livestock with additional requirements, for example, those lactating or growing, received supplementary feed. On average, in March, supplementary feed provided 23% of intake, around half of the crop stubble.

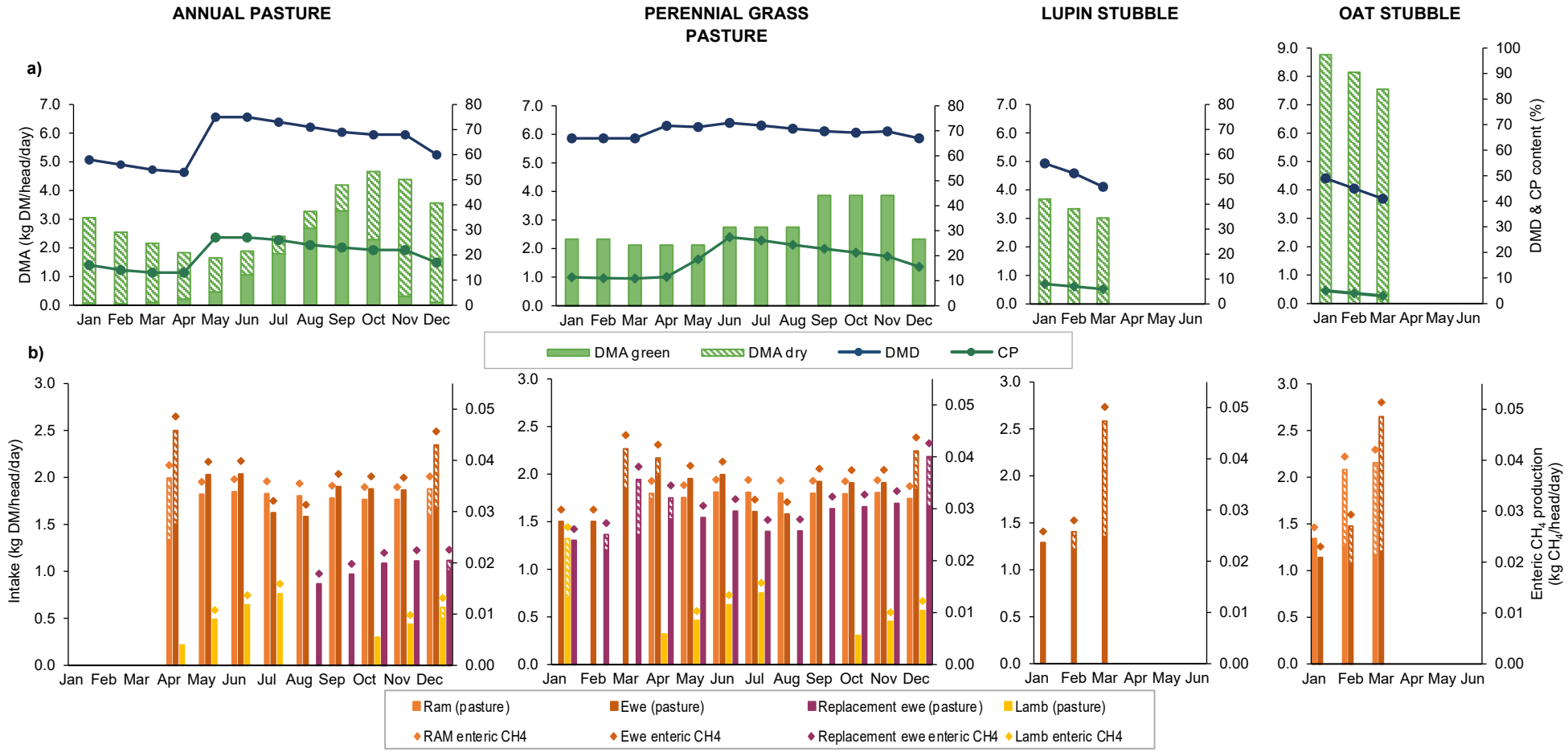


Figure 4.2 - Relationship between feedbase attributes, feed intake and enteric methane production of the breeding flock at the Bremer Bay sheep enterprise. (Where; a) the monthly feedbase attributes; dry matter availability (DMA, kg DM/ha/yr), dry matter digestibility (DMD, %) and crude protein content (CP, %); b) the average daily feed intake (kg DM/hd; solid colour is feedbase intake, patterned colour is supplementary feed intake) and average daily enteric methane production (kg/head, scatter plot) of each stock class. Note: Only GrassGro modelled annual pasture data could distinguish between green and dry DMA, perennial pasture values reflect total DMA only).

Livestock on annual pasture in April required four times the amount of supplementary feed than stock on kikuyu pasture. This resulted in higher enteric methane output than during the growing season when livestock graze high quality greed feed. Referring back to the previous Section, which found that annual pasture had the lowest EI of all the feedbases (2.39×10^{-1} kg CH_4/kg LW, Table 4.5), it is apparent that these overall averages do not reflect the employed farm management practices. Annual pasture was only most productive because it was grazed primarily during the growing season when pasture quality was high. When annual pasture was of poor quality, growing stock were instead grazed on kikuyu, whilst mature stock were moved to crop stubble. During this period stubble was of similar quality to senesced annual pasture (Figure 4.2). If livestock had instead been grazed on annual pasture through the non-growing season, the pasture's ability to produce saleable lightweight would have been much lower, as reflected in the values obtained for stubble. This highlights the primary benefit of running summer-active perennials and the implications of grazing management decisions such as stubble grazing, both from an emissions and an animal production perspective.

Across the stock classes at Bremer Bay, the greatest average daily intake and enteric methane production was attributed to rams and mature ewes, followed by replacement ewes and then lambs. Though not presented separately in Figure 4.2b, overall, lactating ewes consumed the most feed (2.31 kg DM/head/day) and produced the greatest daily enteric emissions (0.44 kg CH_4/head/day) across the farm (Table 4.6). By contrast, lambs, even with the high growth rates, consumed the least feed (0.55 kg DM/head/day) and produced the lowest emissions (0.014 kg CH_4/head/day).

Along whole-farm differences between stock classes, the intake and emission output of each stock class also varied between feedbases. For example, the average intake of a mature dry ewe ranged from 1.38 kg DM/head/day (0.028 kg CH_4/head/day) on the lower quality oat stubble to 1.56 kg DM/head/day (0.031 kg CH_4/head/day) on higher quality kikuyu, a difference of 12%. The results also demonstrate variations in feed intake and enteric methane production a particular stock class grazing a feedbase across the production year. For example, the feed intake and enteric methane production of a dry mature ewe grazing kikuyu ranged from 1.50 kg DM/head/day in January and February, to 1.63 kg DM/head/day in June when pasture quality was highest. This is an 8% difference on the same feedbase.

Table 4.6 - Daily feed intake and calculated enteric methane production across stock classes for the annual and perennial grass pastures, crop stubbles and the feedlot at the Bremer Bay sheep enterprise

Input	Unit	Annual pasture		Perennial grass pasture		Lupin stubble		Oat stubble		Total	
		Mean	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**
Months grazed		Apr-Dec		Jan-Dec		Jan-Mar		Jan-Mar			
Stocking rate[a]	hd/ha	3-6		3-6		3-6		3-6			
Proportion of LW produced for sale[b]	%	26		52		1		3			
					Daily intake[c]						
Rams	kg DM/hd	1.80	*0.12*	1.75	*0.11*	-	-	1.86	*0.37*	1.78	*0.16*
Ewes (dry)	kg DM/hd	1.54	*0.10*	1.56	*0.04*	1.39	*0.08*	1.38	*0.17*	1.51	*0.11*
Ewes (lact.)	kg DM/hd	2.25	*0.32*	2.20	*0.19*	2.91	*0.00*	2.97	*0.00*	2.31	*0.34*
Ewes (rep.)	kg DM/hd	1.35	*0.37*	1.60	*0.33*	-	-	-	-	1.54	*0.26*
Lambs	kg DM/hd	0.50	*0.18*	0.60	*0.31*	-	-	-	-	0.55	*0.36*
					Daily enteric methane production						
Rams	kg CH_4/hd	0.035	*0.002*	0.035	*0.002*	-	-	0.037	*0.007*	0.035	*0.003*
Ewes (dry)	kg CH_4/hd	0.031	*0.002*	0.031	*0.01*	0.028	*0.001*	0.028	*0.003*	0.030	*0.002*
Ewes (lact.)	kg CH_4/hd	0.044	*0.006*	0.043	*0.004*	0.056	*0.000*	0.057	*0.000*	0.044	*0.006*
Ewes (rep.)	kg CH_4/hd	0.027	*0.007*	0.032	*0.006*	-	-	-	-	0.031	*0.007*
Lambs	kg CH_4/hd	0.011	*0.002*	0.015	*0.006*	-	-	-	-	0.014	*0.004*
					Feedlot						
Months in feedlot		Feb, Aug									
Proportion of LW sold[b]	%	18									
Daily intake	kg DM/hd									0.97	-
Daily enteric CH_4 production	kg CH_4/hd									0.020	-
					Supplementary feed						
Months fed		Dec, Apr		Jan-Apr		Feb-Mar		Feb-Mar			
Proportion of total intake											
December	%	21		12		-		-		16	
January	%	-		25		-		-		25	
February	%	-		6		14		29		16	
March	%	-		23		45		52		40	
April	%	37		9		-		-		23	

Hd = head (livestock unit)

[a] As per farm information. Stocking rate increased post lambing.

[b] Proportion of total liveweight sold produced on each feedbase.

[c] Calculated average of combined feedbase and supplementary feed intake for the months grazed.

Influence of management practices on feed intake and enteric methane production

The differences in feed intake and enteric methane production observed between stock classes, between the same stock class grazing different feedbases and between months by a stock class grazing the same feedbase across a production year, highlights the importance of good management practices. Factors such as stock class distribution (i.e. higher proportion of mature stock to younger stock), grazing management (i.e. rotation of stock across feedbases) and timing of production events

(i.e. month of lambing, turn-period), will influence intake and thus the overall emissions of an enterprise.

An example of the influence of management practices at Bremer Bay is the two lambing events that occurred each production year. Of the two, the September lambing produced lower emissions than the March lambing, with an average combined daily intake by lactating ewes and lambs (from lambing to transfer to feedlot) of 2.82 kg DM/day (0.054 kg CH_4/day) as opposed to 2.92 kg DM/day (0.056 kg CH_4/day). A further breakdown across feedbases revealed that for the March lambing, where lambing occurred on crop stubble before being moved to annual pasture, the combined intake and emissions were the highest at 3.05 kg DM/day (0.059 kg CH_4/day), reflecting the high levels of supplementary feed supplied from March to April. By contrast, where the March lambing occurred on kikuyu, the combined ewe and lamb intakes and emissions were 8% lower, totalling 2.79 kg DM/day or 0.054 kg CH_4/day, reflecting the lower supplementary feed requirements. In terms of the September lambing, ewes and lambs grazing annual pasture yielded similar values to kikuyu, totalling 2.83 and 2.80 kg DM/day (0.054 and 0.055 kg CH_4/day), respectively. This included the higher intake of lambs which were supplementary fed in January for a month after weaning.

The above example demonstrates the importance of timing of lambing and grazing management. It showed that regardless of the month of lambing, livestock grazing kikuyu pasture produced lower emissions than annual pasture. Despite the higher quality of annual pasture compared to kikuyu pasture during the growing season, its inability to provide sufficient feed outside of these months reduced its overall ability to produce saleable liveweight. Secondly, it demonstrates that even though a March lambing allowed lambs to be finished on high quality pasture, the emissions resulting from having to supplementary feed lactating ewes at the onset of the lambing outweigh such benefits. The emission burden of having to supplementary feed weaned lambs following a September lambing is less than the emissions burden of supplementary feeding ewes in the March lambing.

A second example was the use of feedlots to finish lambs in February and August. It was calculated that the daily intake of lambs during the month in the feedlot was 0.97 kg DM/head/day (0.020 kg CH_4/head/day). This was lower than that of weaned September lambs whilst on pasture in January (1.32 kg DM/head/day, 0.026 kg CH_4/head/day), because of the supplementary feed requirements on pasture. Therefore, lambs in the feedlot were able to achieve higher growth rates with lower

intake and methane output to those grazing kikuyu in January. However, the feedlot values were higher than that of weaned lambs on either pasture type in July (average 0.76 kg DM/head/day, or 0.016 kg CH_4/head/day). Despite this, it was calculated that it would take 40% longer for lambs to reach the target liveweight of 45 kg on pasture than the month required to do so in the feedlot. The emissions implications of maintaining lambs on pasture for this additional period offset any potential benefit of removing the feedlot. The implications of feedlots on the carbon footprint of a livestock production system are further examined in Chapter Seven.

4.3 Inventory of the Wickepin sheep production enterprise

4.3.1 Farm overview

The second enterprise considered was a mixed crop-livestock enterprise located near Wickepin (117°E, 24°S), in WA's Wheatbelt region. Covering approximately 6,000 ha, 2,430 ha was dedicated to pasture production and 2,650 ha to income and supplementary feed crops. The remaining farm area was a mix of native vegetation, farm infrastructure and salinity-affected land. The climate was dry temperate, characterised by hot, dry summers and cool winters. The 30-year historical minimum and maximum average daily temperatures were below 5°C in July and above 31°C in January (Jeffrey et al., 2001). The region received relatively low average annual rainfall (360 mm) with almost 70% of this occurring between May and September (Figure 4.1). The growing season was two months shorter than at Bremer Bay, averaging five months, starting late-May and concluding mid-November.

The Wickepin case study farm operated a self-replacing 3,100 head Merino ewe enterprise, producing both prime lamb and fine wool. In 2004, to combat falling wool prices, the farm diversified from fine merino wool production to include prime lamb with the introduction of prime South African Meat Merinos (SAMM). At the time of the interviews, the farm produced first-cross prime SAMM-Merino lambs from SAMM rams and Merino ewes, and Merino lambs from Merino rams and Merino ewes. This resulted in two lambing periods each year; SAMM lambs in April and Merino lambs in June. The farmer did so to coordinate the annual feed demands of the flock with farm feed supply. SAMM lambs can achieve higher growth rates than Merinos and are efficient converters of feed to liveweight. This meant they could utilise the growing season pasture supply before being sold directly off the ewe. Merino lambs were unable to meet sale weights by weaning and were finished on lupin stubble with supplementary feed. Cull ewes and rams were also sold, directly off pasture or stubble. Fine Merino wool was produced by the Merino ewe breeding flock and

weaner Merino ewes prior to sale. SAMM wool is of lower quality, so whilst the SAMM rams were sheared annually, SAMM lambs were sold prior to shearing.

Annual pasture, covering 2,320 ha, was the primary feed source for livestock, grazed from April to December. The pasture was a legume-dominant mix comprised of sub clover, annual ryegrass, capeweed (*Arctotheca calendula*), barley grass (*Hordeum* spp.) and brome grass (*Bromus* spp.). A small area (110 ha) of the farm was also planted to the perennial shrub, old man saltbush (*Atriplex nummularia*), with annual pasture in the inter-rows. The first saltbushes were established in 1985 on salinity-affected land, with further plantings occurring in 2000 and 2014. The saltbush transformed marginal land to that which could support livestock and was grazed from November to April.

More than half of the arable area of the farm (2,350 ha) was dedicated to rotational cropping of annual crops for income such as wheat, canola, barley and oats. Lupin (300 ha) was also grown for the sole purpose of providing on-farm supplementary feed for livestock. All crop stubble was grazed during the non-growing season.

4.3.2 Livestock information

The characteristics of the Wickepin sheep flock are detailed in Table 4.7. The breeding flock comprised of 3,100 Merino ewes with a mature liveweight of 70 kg, 50 Merino rams at 84 kg and 20 prime SAMM rams at 110 kg. Each year, on average 600 cull ewes and 15 cull rams were sold from December to February. For calculation purposes it was assumed that the sale of cull stock occurred at the end of January to avoid excess supplementary feed requirements during the summer-autumn feed gap. The farm was expanding its breeding flock so annually on average 800 replacement Merino ewes were sourced from on-farm. These maiden ewes were first joined at a liveweight of 55 kg at 19 months to Merino rams during the Merino joining period in January. Replacement rams were purchased prior to each joining period at 20 months and liveweight of 80 kg or 110 kg depending on whether they were Merino or prime SAMM, respectively. All breeding stock were jetted for worms in September and drenched for worms in November.

Each year on average, 800 SAMM-Merino lambs and 1,600 Merino lambs were weaned. Merino ewes were joined with the SAMM rams in November and lambing occurred in the following April. The SAMM lambs were marked, crutched, tailed and vaccinated in July. The high growth rate characteristic of SAMMs meant that the lambs could be weaned and sold directly off ewes in October at six months at 45 kg. The SAMM average weaning rate was 92.5%. The second joining period, between

Merino ewes and rams, occurred in January. The Merino lambing occurred in June, followed by marking, mulesing, crutching and vaccinating in August. Merino lambs had not attained sale weight by weaning in December at six months of age and so remained on pasture until the end of December before being moved to lupin stubble. The average weaning rate of the Merino lambs was 85%. The 800 wether lambs were finished on lupin stubble prior to sale at the beginning of February at a target liveweight of 37 kg. Weaner ewes destined to become replacement ewes remained on the stubble. Along with vaccinations at marking, the SAMM and Merino lambs were jetted in September, while the Merino lambs were drenched in November with the breeding flock.

Table 4.7 - Characteristics of the Merino and SAMM breeding flocks and annual veterinary inputs at the Wickepin sheep enterprise

Input	Unit	Ram	Ewe	Replace-ment ewe	SAMM lamb	Merino lamb[a]	Total
Breed		SAMM, Merino	Merino	Merino	SAMM	Merino	
Joining date(s)		1 Nov (SAMM), 1 Jan (Merino)					
Lambing date(s)		1 Apr (SAMM), 1 Jun (Merino)					
Weaning date(s)		1 Oct (SAMM), 1 Dec (Merino)					
Stock count	hd	20, 50	3100	800	800	1600	
Age	months	>24	>24	8-24	0-6	0-8	
LW	kg	110, 80	70	29-70	5-45	4.5-45[b]	
Growth rate[c]	kg/hd/day[e]	0.00	0.00	0.07-0.11	0.24	0.15	
Wool production							
Greasy wool production	kg/hd/day	0.020, 0.014	0.015	0.015	-	0.006	
Clean wool yield	%	73	73	73	-	73	
Sale information							
Sale date(s)		31 Jan	31 Jan	-	1 Oct	1 Feb	
Sale count	hd	5	600	-	800	800	
Sale LW	kg	110, 80	70	-	45	37	
Total LW sold	kg	400	42000	-	36000	29600	108000
Veterinary product application							
Vaccination	ml	-	-	-	816	1632	2448
Drench	ml	4200	159600	48000	-	96000	48720
Other	ml	560	24800	6400	6400	12800	50960
Veterinary product transportation							
Vaccination	tkm						10
Drench	tkm						2550
Other	tkm						760

Hd = head LW = liveweight tkm = tonne-kilometres

[a] Includes wether lambs sold (800 head) and ewe lambs destined to join the breeding flock (800 head).

[b] Merino birthweight sourced from Oldham et al. (2011).

[c] Required growth rates of growing animals to meet farm specifications were calculated from farm-specific information. Mature sheep (>24months) were assumed to have a zero net liveweight change.

4.3.2.1 Wool production at Wickepin

Though wool production falls outside the system boundaries of the present study, as a co-product it was necessary to gather information to both calculate nitrous oxide emissions from sheep excreta and to calculate the protein mass allocation factor for the allocation of emissions between wool and liveweight production (see Chapter Three).

At Wickepin, Merino ewes produced on average 3.95 kg greasy wool annually (3.4-4.5 kg) while Merino rams produced on average 5.25 kg (4.5-6 kg; Table 4.7). In the absence of farm-specific data or species- and stock class-specific ABS data or other industry published data, the greasy wool yield of the SAMM rams and six-month merino weaners was sourced from the national SAMM breeders' association's published breed standards and an Australian study investigating shearing time in Merino flocks, respectively. As such, the greasy wool yield of SAMM rams was assumed to be 5 kg/yr (PSBSA, 2018) and from six-month Merino weaners to be 1.11 kg/yr (Campbell, 2006). Shearing of all stock on-farm occurred in November annually.

4.3.3 Pasture information

Annual pasture was last re-established 10 years prior to the study period. Sub clover was sown into existing annual grasses using farm machinery in April. Prior to seeding, pre-emergent herbicide and pesticide were applied. A targeted pasture fertiliser (Super-copper-zinc-moly) was applied at seeding. At the time of data collection, the farmer had not fertilised pasture for two years. Table 4.8 details the inputs to the pasture systems at Wickepin.

Old man saltbush was first planted in 1985, then in 2000 and 2014, on farmland affected by salt scald. This land had been cleared of native vegetation in the early 1900s and, after years of annual cropping, had become saline due to rising ground water. The goal of the planting had been to examine the effectiveness of saltbush in mitigating salinisation and associated land degradation. Once established, it became a source of out-of-season feed for mature livestock. Saltbush was established by planting rows of seedlings at a rate of 2,000 plants/ha, with annuals grown (assumed to be the same mix as the annual pasture) in the inter-rows. Saltbush seedlings were planted by a specialised implement hired off-farm, attached to a farm tractor. Assuming an average width of 1.20 m per established plant (Honeysett et al., 2004), saltbush comprised 28% and the annual pasture inter-row 72% per hectare. The saltbush was not fertilised. It is acknowledged that annual pasture grown with saltbush may not be as productive as annual pasture grown in other areas of the farm.

Annually, an average of 500 tonne lime was applied to the farm at a rate of 1 t/ha. It was therefore assumed that each paddock containing pasture or crop was limed every ten years.

Table 4.8 - Characteristics and annual inputs of the two pasture types and the supplementary feed crop at the Wickepin sheep enterprise

Input^	Unit	Annual pasture	Saltbush pasture	Lupins[a]	Total
Predominant soil type			Loamy, clay duplex soils		
Area	ha	2320	110	300	
Years since establishment	yr	10	14	1	
Month sown		May-Jun		Apr	
Month harvested		-	-	Dec	
Grain yield	kg/ha	-	-	1200	
Sowing rate	kg/ha	1.5	143[b]	100	
			Chemical application		
Herbicide					
Glyphosate	l	696	24	300	1020
Diflufenican	l	-	-	225	225
Trifluralin	l	-	-	900	900
Pesticide					
Alpha-cypermethrin	l	23	1	-	24
Fertiliser					
Potash	kg	-	-	12000	12000
N, P, S, Ca fertiliser	kg	-	-	19500	19500
Super cu zn & mo fertiliser	kg	6960	236	-	7196
Lime	kg	232000	11000	30000	273000
			Chemical transportation		
Herbicide					
Glyphosate	tkm	3271	111	1332	4714
Diflufenican	tkm	-	-	4025	4025
Trifluralin	tkm	-	-	236	236
Pesticide					
Alpha-cypermethrin	tkm	177	6	-	183
Fertiliser					
Potash	tkm	-	-	3091	3091
N, P, S, Ca fertiliser	tkm	-	-	4586	4586
Super copper zinc & moly fertiliser	tkm	1754	59	-	1813
Lime	tkm	77720	3685	10050	91455
			Fuel consumption		
Tractor diesel	l	2042	156	2573	4771
Harvester diesel	l	-	-	1844	1844
			Farm machinery production		
Tractors	USD	5080	617	6590	12288
Seeder	USD	2934	99	3794	6827
Boomspray	USD	2066	70	4008	6144
Spreader	USD	1042	15	2607	2048
Harvester	USD	-	-	13653	13653

^ Inputs are presented on an annual basis. Inputs applied at establishment x years ago are also considered in this table by adjusted the input value for the number of years since establishment.

[a] Income crop inputs fall outside the scope of this study. For income crop characteristics used in the calculation of animal emissions from grazing stubble see Table 4.10.

[b] Number of saltbush seedlings planted (2,000 per hectare at establishment).

4.3.3.1 Pasture attributes

Annual pasture data sources

As for Bremer Bay, the annual pasture system at Wickepin was modelled in GrassGro using farm information and site-specific SILO weather data. The modelled annual pasture mix included sub clover, annual ryegrass and capeweed.

Saltbush pasture data sources

GrassGro does not have parameter sets for perennial shrubs and so the attributes of the saltbush were obtained from other sources. The monthly average DMA of the perennial pasture at Wickepin was comprised of both the saltbush and annual inter-row components. The DMA of the saltbush component only considers edible dry matter (EDM), the leaf and stems of the shrub (Norman et al., 2010). The monthly EDM was calculated using the planting density and the EDM growth rates of old man saltbush. Like the kikuyu at Bremer Bay, old man saltbush follows a C4 photosynthetic pathway and so growth rates are higher during warmers months. Following a comparable study in south-western Australia, saltbush shrub growth rates were assumed to be 2 g EDM/shrub/day in summer, 5 g in autumn, 2.5 g in winter and 1 g in spring (Norman et al., 2010; Salem et al., 2010). The annual inter-row component was then included as described in Chapter Three to obtain monthly DMA values. DMD and CP contents were sourced from studies undertaken in southern Australia (Honeysett et al., 2004; Norman et al., 2010).

Calculated pasture attributes

Table 4.9 displays the average annual and perennial pasture attributes at Wickepin according to growing season, while Figure 4.3a (Section 4.3.6.1) displays this as monthly information.

The annual NPP of annual pasture was 5.28 t DM/ha, with 90% of this produced during the growing season, from June to November. The annual NPP at Wickepin was lower than Bremer Bay due to the shorter growing season and this is reflected in the differences in the monthly pasture DMAs of each farm (Figures 4.2a and 4.3a). Average DMA over the non-growing season at Wickepin was 1.70 t DM/ha, falling to 1.25 t DM/ha by the following growing season. These were low values, despite annual pasture rested from grazing during this period and highlights the importance of minimising erosion and soil degradation through managed out-of-season grazing, particularly on fragile soils such as at Wickepin. Despite the lower annual NPP, the average DMD (70%) and CP content (23%) of annual pasture during the peak growing seasons was comparable to Bremer Bay. However, Wickepin experienced a sharper

decline in quality at the end of the growing season with an average DMD of 52% and CP content of 12% during the non-growing season. By May, when all livestock were moved back onto annual pasture, DMD and CP content had fallen to 49% and 10%, respectively, below livestock maintenance requirements.

Table 4.9 - Productivity and feed quality attributes of the two pasture types and the lupin crop stubble grazed at the Wickepin sheep enterprise

Input^	Unit	Annual pasture		Saltbush pasture		Lupin stubble	
		Mean	**SD**	**Mean**	**SD**	**Mean**	**SD**
Length of growing season		Jun – mid Nov		Jan - Dec		-	
Annual NPP[a]	t DM/ha	5.28	-	5.63	-	-	-
Growing season NPP[b]	t DM/ha	4.70	-	3.38	-	-	-
Non-growing season NPP[b]	t DM/ha	0.59	-	1.89	-	-	-
Available DMA[c] (% green)						-	-
Start of growing season	t DM/ha	1.25 (35%)	*0.00*	-	-	-	-
Peak growing season	t DM/ha	2.53 (72%)	*0.60*	-	-	-	-
End of growing season	t DM/ha	2.75 (7%)	*0.00*	2.87	*0.00*	-	-
Non-growing season	t DM/ha	1.70 (2%)	*0.40*	1.99	*0.30*	2.33	*0.19*
DMD[d]							
Start of growing season	%	76	*0.00*	-	-	-	-
Peak growing season	%	70	*4.06*	-	-	-	-
End of growing season	%	58	*0.00*	55	*0.0*	-	-
Non-growing season	%	52	*2.62*	52	*1.28*	55	*0.82*
CP content[d]							
Start of growing season	%	27	*0.00*	-	-	-	-
Peak growing season	%	23	*2.69*	-	-	-	-
End of growing season	%	15	*0.00*	16	*0.00*	-	-
Non-growing season	%	12	*1.38*	15	*0.66*	7	*0.82*

^ With the exception of annual net primary productivity (NPP), all averages and standard deviations calculated only from the months feedbase is grazed within that period.

[a] GrassGro modelled output for annual pasture and inter-row component of saltbush pasture. Saltbush values calculated using regionally-specific information (Honeysett et al., 2004; Norman et al., 2010).

[b] Where growing season refers to the annual species growing season at the case study farm location.

[c] GrassGro provided both total available DMA and green DMA but the regional perennial data only provided overall DMA. Lupin stubble DMA calculated from stubble yield (see Chapter Three) and calculated animal intake (Table 4.13).

[d] Annual pasture and saltbush pasture inter-row DMD and CP content obtained from GrassGro. Saltbush DMD and CP content sourced from literature (Fancote et al., 2009; Honeysett et al., 2004; Norman et al., 2010). Stubble attributes obtained from DAFWA (2006a).

The annual NPP of saltbush pasture was 5.63 t DM/ha, of which 1.89 t DM/ha was calculated to be contributed by saltbush. Over 34% of the annual NPP was produced outside of the annual growing season, highlighting the value of saltbush as an out-of-season feed. It was assumed that the average monthly DMD of old man saltbush was

52% and CP content 17.5% (Honeysett et al., 2004; Norman et al., 2010). Sheep will only consume up to 200 g salt per day (Masters et al., 2005) which means that despite other nutritional attributes, the high salt concentration in saltbush prevents stock from consuming the quantities required to meet maintenance energy requirements. Other sources of feed in the paddock, such as pasture, help to overcome this shortfall.

Following Norman et al. (2010) and Fancote et al. (2009) it was assumed that from November through to the end of April, saltbush comprised half the pasture intake and senesced annual inter-row the other half. The calculated DMD and CP content during this period was 55% and 16% in the first month, and 52% and 15%, respectively for the remainder of the grazing period. This fall reflected the decline in the quality of the annual pasture inter-row. Though lower in quality and productivity to kikuyu at Bremer Bay, saltbush pasture provided green feed in the non-growing season at a higher quantity and quality than the declining dry annual pasture and crop stubble.

4.3.4 Crop information

Lupins were grown for the sole purpose of providing supplementary feed to livestock during the dry season. In early- to mid-April, prior to sowing, pre-emergent herbicides were applied to the paddocks using farm machinery. The lupin crop was then sown late April using seed retained from previous years' harvests. At seeding and soon after, fertilisers containing N, P, K, Ca and S were applied. The crop was harvested in December, with an average grain yield of 1,200 kg/ha (Table 4.8).

Wickepin also produced income crops for sale. Though the exact area dedicated to each crop altered with crop rotations, on average; 1400 ha of wheat, 600 ha of canola, 200 ha of barley and 150 ha oats were grown annually. GHG emissions resulting from the production of income crops was excluded from this study, so it was not necessary to collect input information on inputs. However, as livestock grazed the stubble, crop characteristics such as grain yield and feed quality attributes were required to calculate animal emissions (Table 4.10).

4.3.4.1 Crop attributes

Crop stubble was grazed by livestock between harvest in December and sowing in April (lupins) and May (income crops). The starting DMA, or stubble yield, ranged from 2,580 kg DM/ha for the lupin stubble to 9,520 kg DM/ha for oat stubble (Table 4.10). The harvest indices applied to each crop grain yield to obtain these DMA values were; lupins 0.3, oats 0.21, wheat 0.37, barley 0.38 and canola 0.28 (Unkovich et al., 2010a). The DMD and CP content of lupin stubble immediately post-harvest was 56%

and 8%, respectively. These values then averaged 55% and 7%, respectively over the months grazed (Figure 4.3a). The income crop stubble was of lower quality, with average DMDs ranging from 46% to 47% and average CP content from 3% to 5% over the period of grazing (DAFWA, 2006a, 2006b).

Table 4.10 - Characteristics and feed quality attributes of the four income crop stubbles grazed at the Wickepin sheep enterprise

Input^	Unit	Oat stubble		Wheat stubble		Barley stubble		Canola stubble		Income crop average	
		Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
Area	ha	150		1400		200		600			
Years since establishment	yr	1		1		1		1			
Month sown		May		May		May		May			
Month harvested		Dec		Dec		Dec		Dec			
Grain yield	kg/ha	2750		2800		2900		1500			
Available DMA[a]											
Non-growing season	t DM/ha	8.46	*0.79*	3.90	*0.33*	3.65	*0.49*	3.30	*0.26*	4.83	*0.22*
DMD[b]											
Non-growing season	%	47	*1.07*	46	*1.12*	47	*1.21*	46	*1.12*	46	*0.01*
CP content[b]											
Non-growing season	%	4	*0.83*	3	*0.25*	4	*0.83*	5	*0.83*	4	*0.22*

^ All averages and standard deviations calculated only from months stubble is grazed within that period.

[a] Income crop stubble DMA calculated as per lupin stubble.

[b] DMD and CP content sourced from (DAFWA, 2006a).

4.3.5 Grazing management

Annual pasture

Annual pasture at Wickepin was grazed from April to December. Over the growing season, annual pasture was able to support the entire flock. However, following senescence and the subsequent decline in pasture quality by the end of November, livestock required supplementary feeding in December. Lupins and hay were provided in December and April , totalling 197,216 kg, with both sourced on-farm (Table 4.11).

Perennial pasture

Following the decline in annual pasture, a portion of the mature ewes were moved to the saltbush pasture in November, at a stocking rate of 4 sheep/ha. The saltbush was grazed until the end of April when the ewes were moved back to annual pasture, coinciding with the beginning of the annual growing season. This also rested the saltbush over the colder months when growth was lower. All stock grazing saltbush

pasture were provided supplementary feed from December to April (14,819 kg).

Table 4.11 - Feed quality attributes and annual inputs of the supplementary feed supplied at the Wickepin sheep enterprise

Attribute or input^	Unit	Annual pasture	Saltbush pasture	Lupin stubble	Income crop stubble	Total
			Supplementary feed attributes[a]			
DMD						
Lupins	%		94			
Hay	%		59			
CP content						
Lupins	%		38			
Hay	%		8			
			Supplementary feed provided[b]			
Oats	kg	157075	11803	39160	120360	318647
Hay	kg	40141	3016	10008	30759	85403

^ Inputs are presented on an annual basis.

[a] DMD and CP content of supplied feed sourced from (DAFWA, 2006a; NSW DPI, 2016).

[b] As fed, considering the DM content of feed and assumed wastage (20%).

Crop stubble

At the beginning of January, all remaining stock on annual pasture were moved to crop stubble. The recently weaned Merino wethers and ewe lambs were moved to the higher quality lupin stubble. Whilst wethers were sold directly off lupin stubble at the end of January, weaner ewes destined to become replacement ewes remained on stubble until they were moved onto annual pasture in April, prior to lupin crop seeding.

The remainder of the flock were moved to income crop stubble. It was assumed that Merino ewes pregnant with SAMM lambs were kept separate from the remaining sheep classes in January and moved back onto annual pasture at the end of March for lambing in April. The remaining ewes and rams grazed income crop stubble until the end of April before being returned to annual pasture in May. All livestock grazing stubble were provided with lupins and hay for the duration of grazing. The total supplementary feed provided on lupin stubble and income crop stubble was 49,168 kg and 151,119 kg, respectively. Unlike Bremer Bay, Wickepin did not have a feedlot and lambs were finished on either pasture (SAMM) or stubble (Merino).

4.3.6 Calculated on-farm inventory outputs

As the sheep flock at Wickepin produced both wool and meat, it was necessary to allocate emissions between products. Using the methods described in Chapter Three; a protein mass allocation factor was calculated for each. The allocation factor for liveweight produced was 62% and to wool was 38%. These values are in line with those obtained in other studies which applied protein mass allocation (Cottle & Cowie,

2016; Wiedemann et al., 2015a). Chapter Five allocates emissions according to carbon dioxide equivalents (CO_2-e) however Table 4.12 presents the unallocated emissions for each of the calculated on-farm outputs at Wickepin.

Enteric methane emissions

The total enteric methane produced at Wickepin was 49,755 kg CH_4/year, four times the annual amount produced at Bremer Bay. This was expected given livestock and feedbase differences between the two sheep enterprises. Comparing the feedbases in terms of total liveweight grazed, lupin stubble which only supported immature stock, produced the most emissions (1.76×10^{-2} kg CH_4/kg LW grazed). This was followed by annual pasture, which was of higher quality and supported all stock classes within the flock for most of the year (1.50×10^{-2} kg CH_4/kg LW grazed).

In terms of saleable liveweight production, enteric methane production prior to allocation, was 4.61×10^{-1} kg CH_4/kg LW produced for sale, more than double Bremer Bay. This indicates that though Wickepin ran a larger enterprise and turned off more liveweight annually; in terms of enteric methane emissions, it was unable to convert liveweight into saleable product as efficiently as Bremer Bay. Of the feedbases, annual pasture produced saleable liveweight most efficiently (4.03×10^{-1} kg CH_4/kg LW), while livestock grazing saltbush pasture produced 7.22×10^{-1} kg CH_4/kg LW, 44% more than annual pasture. This reflected the large amount of liveweight produced on annual pasture. Section 4.3.6.1 investigates these differences further.

Even once wool production had been accounted for through allocation, to produce a revised value of 2.85×10^{-1} kg CH_4/kg LW, Wickepin remained less efficient from an emissions perspective at producing saleable liveweight. This revised value was still 24% higher than Bremer Bay.

Manure methane emissions

Total unallocated manure methane emissions were 12.13 kg CH_4/year. As for Bremer Bay, in terms of liveweight grazed, livestock on both pastures produced less emissions than crop stubble. This was even though the saltbush pasture had lower monthly DMDs than lupin stubble per month grazed. The higher quantities of enteric methane produced on lupin stubble was instead a response to the higher intake by growing stock per kilogram of liveweight than mature stock on perennial pasture (Figure 4.3). In terms of saleable liveweight, annual pasture was the most efficient, producing 9.13×10^{-5} kg CH_4/kg LW produced for sale, at least 30% less than the other feedbases. This is unsurprising considering that annual pasture supported the majority of the flock throughout the production year, including the two lambings.

Overall, the farm produced 1.12×10^{-4} kg CH_4/kg LW, 57% more than Bremer Bay. Even with allocation, Wickepin produced more manure methane per unit of saleable liveweight than Bremer Bay, indicative of the lower feed quality across the farm.

Nitrous oxide emissions from excreta

Excreta nitrous oxide emissions totalled 468.14 kg N_2O/yr, or 4.33×10^{-3} kg N_2O/kg LW produced for sale. In terms of saleable liveweight, this meant that Wickepin operated at less than half the efficiency of Bremer Bay. Even on an allocated basis, Wickepin produced 29% more nitrous oxide per kilogram of liveweight. Across the feedbases, saltbush was the least efficient, producing 6.19×10^{-3} kg N_2O/kg LW, with annual pasture the most efficient, producing 4.22×10^{-3} kg N_2O/kg LW. The high efficiency of annual pasture reflects its saleable liveweight output, as it supported the production of 84% liveweight sold. This is more evident when examined in terms of liveweight grazed; where annual pasture produced the highest emissions. This is a combined result of the higher CP content of this pasture and that it supported the breeding herd through two lambing cycles. Lactating and growing stock excrete more nitrous oxide per kilogram of liveweight than dry stock. By comparison, saltbush was only grazed by mature dry ewes and so produced a smaller proportion of liveweight sold.

Other soil emissions

The unadjusted nitrous oxide emissions from stubble decomposition at Wickepin produced 156.30 kg N_2O/yr. Unlike Bremer Bay, this was less than a third of excreta emissions, attributed to the larger sheep flock at Wickepin and lower stubble yields. On a saleable liveweight basis, this totalled 4.88×10^{-2} kg N_2O/kg LW produced for sale on lupin stubble, 48% higher than lupin stubble at Bremer Bay. However, once allocation for wool production was considered, this fell to 13% less than Bremer Bay (3.01×10^{-2} kg N_2O/kg LW). Presenting the emissions instead as a measure of efficiency across area cropped, lupin stubble produced 5.21×10^{-1} kg N_2O/ha prior to allocation. This was less than Bremer Bay and reflected the lower post-grazing stubble yield at Wickepin. These calculated metrics demonstrated that stubble load was managed more efficiently at Wickepin, but that Bremer Bay, through grazing management, was able to produce saleable liveweight on the stubble more efficiently.

Pasture residue at Wickepin contributed 7.55×10^{-4} kg N_2O/kg LW (70.73 kg N_2O/yr), more than triple that of Bremer Bay. Examination of annual and saltbush pasture separately revealed that on a per hectare basis, both produced less nitrous oxide than Bremer Bay (annual, 2.95×10^{-2} kg; saltbush, 2.13×10^{-2} kg N_2O/ha). This was because the Wickepin farm, with a lower annual NPP of annual pasture, had a higher pasture

Table 4.12 - Calculated annual animal, soil and associated on-farm emissions produced on the Wickepin sheep enterprise

Output	Unit	Annual pasture	Saltbush pasture	Lupins	Income crop stubble	Total
			Animal emissions			
Enteric CH_4	kg CH_4/yr	36532.84	2152.43	1914.27	9156.40	49755.93
	kg CH_4/kg LW grazed[a]	1.50E-02	1.16E-02	1.76E-02	1.26E-02	1.44E-02
	kg CH_4/kg LW produced for sale[b]	4.03E-01	7.22E-01	5.98E-01	8.20E-01	4.61E-01
Manure CH_4	kg CH_4/yr	8.28	0.66	0.42	2.77	12.13
	kg CH_4/kg LW grazed	3.40E-06	3.57E-06	3.87E-06	3.80E-06	3.51E-06
	kg CH_4/kg LW produced for sale	9.13E-05	2.22E-04	1.31E-04	2.48E-04	1.12E-04
			Direct soil emissions			
N_2O Excreta	kg N_2O/yr	382.54	18.46	15.10	52.04	468.14
	kg N_2O/kg LW grazed	1.57E-04	9.99E-05	1.39E-04	7.14E-05	1.35E-04
	kg N_2O/kg LW produced for sale	4.22E-03	6.19E-03	4.72E-03	4.66E-03	4.33E-03
N fertilisers N_2O	kg N_2O/yr	-	-	3.86	-	3.86
	kg N_2O/ha	-	-	1.29E-02	-	1.29E-02
	kg N_2O/kg LW produced for sale	-	-	-	-	3.58E-05
Crop residue N_2O	kg N_2O/yr	-	-	156.30	-	156.30
	kg N_2O/ha	-	-	5.21E-01	-	5.21E-01
	kg N_2O/kg LW produced for sale	-	-	4.88E-02	-	1.45E-03
Pasture residue N_2O	kg N_2O/yr	68.40	2.34	-	-	70.73
	kg N_2O/ha	2.95E-02	2.13E-02	-	-	2.91E-02
	kg N_2O/kg LW produced for sale	7.55E-04	7.84E-04	-	-	7.55E-04
			Indirect soil emissions			
Atmospheric deposition N_2O	kg N_2O/yr	72.92	3.48	3.24	9.65	89.29
	kg N_2O/kg LW produced for sale	8.04E-04	1.17E-03	1.01E-03	8.64E-04	8.27E-04
			Other calculated on-farm emissions			
Liming CO_2	kg CO_2/yr	91872.00	4356.00	11880.00	-	108108.00
	kg CO_2/ha	3.96E+01	3.96E+01	3.96E+01	-	3.96E+01
	kg CO_2/kg LW produced for sale	1.01E+00	1.33E-02	1.24E-021	-	1.00E+00
Urea hydrolysis CO_2	kg CO_2/yr	-	-	-	-	-
	kg CO_2/ha	-	-	-	-	-
	kg CO_2/kg LW produced for sale	-	-	-	-	-

LW= liveweight

[a] To enable comparison across feedbases, emissions were adjusted according to output per kg of liveweight grazed on each.

[b] Using the functional unit, overall emissions were adjusted according to output per kg of liveweight produced for sale on each pasture.

utilisation and thus lower quantity of pasture residue remaining after grazing. These values were even lower once allocation between products had been considered. Despite this, in terms of saleable liveweight produced, annual pasture at Wickepin produced higher emissions than both annual and kikuyu pasture at Bremer Bay (20% and 68%, respectively). This highlights that the Bremer Bay farm was more efficient at producing saleable liveweight.

Other sources of nitrous oxide emission at Wickepin included those from the application of N fertiliser (0.08% of total N_2O emissions) and from atmospheric deposition (2% of total N_2O emissions). As the farm employed a periodic liming treatment, annually 108,108 kg CO_2, or 1.00 kg CO_2/kg LW, was produced because of this practice. Similar to Bremer Bay, the enterprise did not experience leaching or runoff, or emissions associated with urea hydrolysis.

4.3.6.1 Analysis of the interactions between feedbase, animal intake and enteric methane production at the Wickepin sheep enterprise

The previous Section found that annual pasture at Wickepin had the lowest enteric methane EI, followed by lupin stubble, saltbush pasture and finally income crop stubble. This Section examines the factors influencing these calculated values, from pasture attributes, to livestock intake, to the influence of grazing management.

As expected, feed intake and ensuing enteric methane emissions were high for all stock classes on annual pasture over the growing season (Figure 4.3). Annual pasture supported the production of SAMM lambs to sale and production of Merino lambs until the final month prior to sale. Of all liveweight sold from Wickepin, 84% (Table 4.13) was produced on annual pasture, yielding the high emissions efficiency of the pasture. In December, as the quality of annual pasture declined, so too did pasture intake. To meet this deficit, supplementary feed comprised 20% of total intake, increasing intake and ensuing enteric emissions (Table 4.13). Mature stock and older replacement ewes moved to income crop stubble required high levels of supplementary feed, averaging 25% of the ewes' total intake. The Merino wethers destined for sale and maiden ewes destined to join the breeding herd grazing lupin stubble also required supplementary feeding, despite the lower daily feed requirements of these two stock classes. During this period, supplementary feed comprised 36 to 43% of total intake.

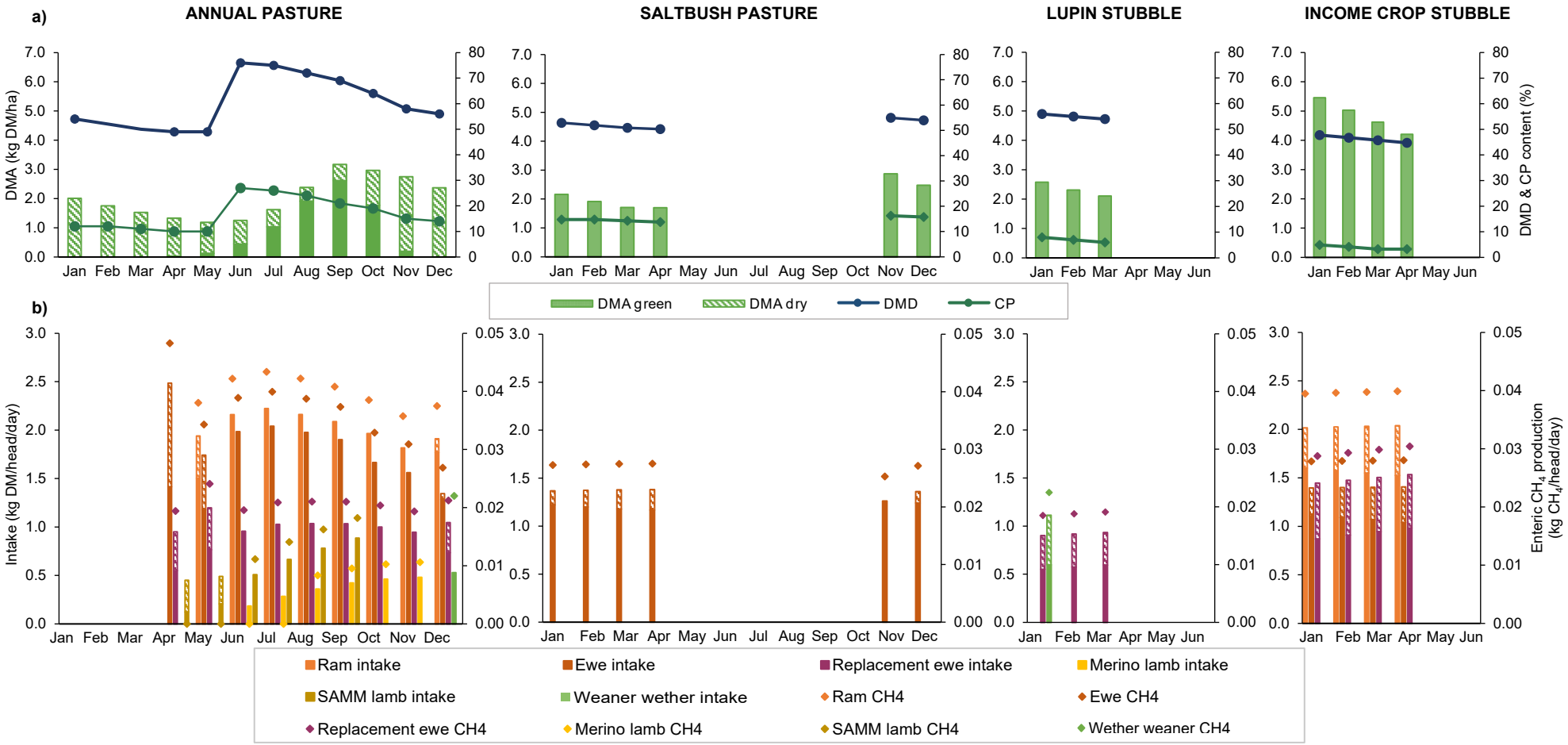


Figure 4.3 - Relationship between monthly feedbase attributes, feed intake and enteric methane production of the breeding flock at the Wickepin sheep enterprise.

(Where; a) the feedbase attributes; dry matter availability (DMA, kg DM/ha/yr), dry matter digestibility (DMD, %) and crude protein content (CP, %); b) the average daily feed intake (kg DM/hd; solid colour is feedbase intake, patterned colour is supplementary feed intake) and average daily enteric methane production (kg/head, scatter plot) of each stock class. Note: Only annual pasture data modelled in GrassGro could distinguish between green and dry DMA, perennial pasture monthly values reflect total DMA only).

The inability of annual pasture to support livestock during the non-growing season is observed in April, where 50% of the intake of ewes moved back onto pasture for the SAMM lambing was supplementary feed. By contrast, saltbush supported the maintenance requirements of dry ewes with minimal supplementary feed; ranging from 9% in December to a maximum of 15% at the end of the non-growing season. This was 55 to 71% less than that required on crop stubble and annual pasture, respectively. As observed in the previous section, saltbush had a higher enteric methane EI than that of annual pasture and lupin stubble as it only supported breeding ewes, of which a small proportion were converted into sold liveweight (3%). However, it also played a role in providing green feed of sufficient quality during a period where otherwise high levels of supplementary feed would have been required.

Across stock classes, the greatest average daily feed intake and ensuing enteric methane production was attributed to lactating mature ewes (2.13 kg DM/head/day, 0.042 kg CH_4/head/day; Table 4.13), followed by rams, dry ewes, replacement ewes and finally lambs. Of the two lamb breeds, SAMM lambs, with faster growth rates and turn-off periods, had average intake enteric methane production levels of 13% and 20% higher than Merino lambs.

Across the farm, overall feed availability and quality was lower than that at Bremer Bay and this was reflected in calculated intakes and enteric methane outputs. Dry and lactating Merino ewes consumed on average 6% and 8% less feed and produced 7% and 5% less enteric methane, respectively, compared to Dorper ewes at Bremer Bay. Considering that the ewes at both farms were the same liveweight, these differences could be attributed to differences in the feedbase consumed. Despite the lower quality of the feedbases at Wickepin compared to Bremer Bay, both Merino and SAMM rams consumed 16% more feed and produced 15% more enteric methane than Dorper rams at Bremer Bay. This can be attributed to differences in ram liveweights. The Merino lambs yielded similar output values to Dorper lambs, while the SAMM lambs consumed more and produced more enteric methane. These differences highlight the importance of both breed and feedbase choice from an emissions perspective.

As for Bremer Bay, intake and enteric methane output of each stock class varied between feedbases grazed at Wickepin. For example, mature dry ewes, for months grazed, had average daily intakes ranging from 1.47 kg DM/head/day (0.029 kg CH_4/head/day) on annual pasture, to 1.40 kg DM/head/day (0.028 kg CH_4/head/day) on income crop stubble, to only 1.35 kg DM/head/day (0.027 kg CH_4/head/day) on saltbush pasture. The higher intake on annual pasture is reflective of higher feed quality during the growing season, when intake was less restricted. By contrast, the

higher intake values on crop stubble reflect high levels of required supplementary feed. Though intake on saltbush is between 4 to 8% less than the other feedbases, the animal is still able to meet its requirements. Compared to kikuyu at Bremer Bay, growing or lactating livestock cannot be supported on saltbush pasture during the dry season. However, saltbush can support the breeding herd at maintenance, yielding less enteric methane and consuming less supplementary feed than the same stock class grazing crop stubble or senesced annual pasture. This highlights its potential as an out-of-season green feed, explored further in Chapter Six.

Table 4.13 - Daily feed intake and calculated enteric methane production across stock classes for the annual pasture, perennial pasture and crop stubbles at the Wickepin sheep enterprise

Input	Unit	Annual pasture		Saltbush pasture		Lupin stubble		Income crop stubble		Total	
		Mean	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**
Months grazed		Apr-Dec		Dec-Apr		Jan-Mar		Jan-Mar			
Stocking rate[a]	Hd/ha	3		4		3		3			
Proportion of LW produced for sale[b]	%	84		3		3		10			
						Daily intake[c]					
Rams	kg DM/hd	2.17	*0.36*	-	-	-	-	2.03	*0.01*	2.11	*0.33*
Ewes (dry)	kg DM/hd	1.47	*0.12*	1.35	*0.04*	-	-	1.40	*0.00*	1.42	*0.09*
Ewes (lact.)	kg DM/hd	2.13	*0.28*	-	-	-	-	-	-	2.13	*0.28*
Ewes (rep.)	kg DM/hd	1.05	*0.17*	-	-	0.92	*0.01*	1.49	*0.01*	1.12	*0.24*
SAMM lambs	kg DM/hd	0.63	*0.16*	-	-	-	-	-	-	0.63	*0.16*
Merino lambs	kg DM/hd	0.47	*0.27*	-	-	1.11	*0.0*	-	-	0.55	*0.33*
						Daily enteric methane production					
Rams	kg CH_4/hd	0.042	*0.007*	-	-	-	-	0.040	*0.005*	0.041	*0.006*
Ewes (dry)	kg CH_4/hd	0.029	*0.002*	0.027	*0.001*	-	-	0.028	*0.000*	0.028	*0.002*
Ewes (lact.)	kg CH_4/hd	0.042	*0.005*	-	-	-	-	-	-	0.042	*0.007*
Ewes (rep.)	kg CH_4/hd	0.021	*0.003*	-	-	0.019	*0.000*	0.030	*0.001*	0.023	*0.003*
SAMM lambs	kg CH_4/hd	0.015	*0.002*	-	-	-	-	-	-	0.015	*0.005*
Merino lambs	kg CH_4/hd	0.012	*0.004*	-	-	0.023	*0.00*	-	-	0.012	*0.005*
						Supplementary feed					
Months fed		Dec, Apr-May		Dec-Apr		Jan-Mar		Jan-Apr			
Proportion of total intake											
December	%	20		9		-		-		16	
January	%	-		11		43		24		26	
February	%	-		13		37		26		25	
March	%	-		15		36		27		26	
April	%	52		15		-		29		32	
May	%	36		-		-		-		36	

Hd = head (livestock unit)

[a] Average stocking rate as per farm information.

[b] Proportion of total liveweight sold produced on each feedbase.

[c] Calculated average of combined feedbase and supplementary feed intake.

Influence of management practices on feed intake and enteric methane production

The results presented in Figure 4.3 and Table 4.13 allow for a comparison of SAMM and Merino lamb production at Wickepin. Of the two, in terms of average daily

emissions over each lambing cycle, the production of SAMM lambs had a higher impact than Merino lambs. The average combined daily intake of both lactating ewes and SAMM lambs was 2.87 kg DM/day (0.056 kg CH_4/day), 12% higher than Merino lambs with intakes of 2.54 kg DM/day (0.049 kg CH_4/day). As discussed previously, SAMM lambs consumed more and produced more enteric methane than Merino lambs, despite weaned Merino wethers being finished on a high intake of lupin stubble and supplementary feed. In addition, the Merino ewes which produced SAMM lambs yielded higher daily average intakes (11% higher) and enteric methane emissions (11% higher), than the Merino ewes which produced Merino lambs. These differences can be attributed to the higher quantity of supplementary feed required by ewes during the SAMM lambing in April and May, prior to the start of the growing season. By contrast the winter lambing of Merinos meant that no supplementary feed was required by lactating ewes.

Despite the differences in the combined daily average emission output of the production of SAMM versus Merino lambs, the total impact over the duration of each lambing cycle presents a different finding. The higher growth rates of SAMM lambs meant that they were able to be sold at weaning at six months and an average liveweight of 45 kg. By contrast, Merino lambs were sold at eight months and an average liveweight of 37 kg. Using the daily enteric methane production averages calculated in previously, the enteric emissions impact of producing Merino lambs was actually 14% higher than SAMM lambs. Comparing these lambs on a per kilogram of liveweight sold increases this gap to 30%. The implications of the two lamb breeds is explored further in Chapter Five. The example presented here demonstrates the influence of practices; such as time of lambing, breed choice and turn-off period.

4.4 Inventory of the Dongara beef production enterprise

4.4.1 Farm overview

The Dongara case study farm was a cattle breeding and backgrounding enterprise located east of the town of Dongara, in WA's Northern Agricultural Region (115°E, 29°S). The farm covered 3,600 ha, with 3,000 ha dedicated to pasture production and the remaining 600 ha to native vegetation and farm infrastructure. The region is characterised by long, hot summers and mild winters, reflected by the site-specific historical average minimum temperature of 9°C in August and maximum of 35°C in February (Figure 4.4). The farm received average annual rainfall of 442 mm (Jeffrey et al., 2001), with minimum and maximum average monthly rainfall of 7 mm in January and 92 mm in July, respectively. Though the farm did not receive the lowest rainfall of

the farms considered in this study (Wickepin enterprise), it had the shortest growing season, receiving 70% of annual rainfall between June and October. It also received the lowest out-of-season rainfall, with only 15% rainfall received between December and April.

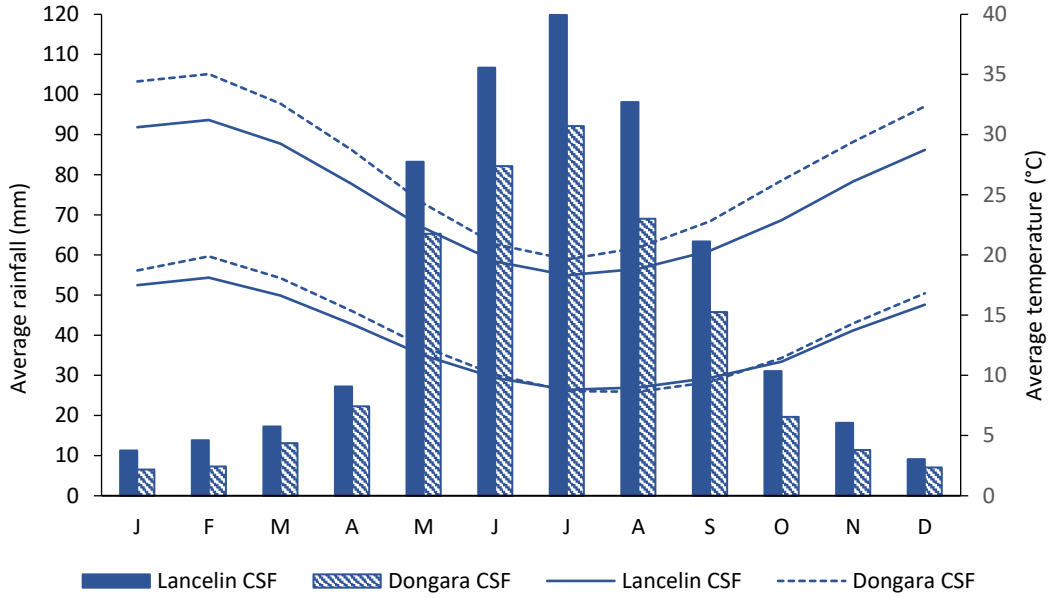


Figure 4.4 - The 30-year (1985-2014) average monthly rainfall (mm), minimum and maximum temperatures (°C) at the two beef production enterprises.

Where: average monthly rainfall (mm, bars), minimum and maximum temperatures (°C, lines) at Lancelin (blue, solid) and Dongara (blue, striped). Site-specific historical weather data was sourced from SILO (Jeffrey et al., 2001).

The primary focus of the enterprise was the backgrounding of cattle received from pastoralists in northern WA. The farm was a member of a profit share alliance with pastoralists, incentivised by a share of sales received per liveweight gain on-farm. The farm turned off around 3,000 pastoral cattle annually, predominantly *Bos indicus* breeds such as Brahman, Droughtmaster and Santa Gurtrudis. Cattle received onto the farm were backgrounded to target market specifications; including heifers and steers for domestic feedlots as well as steers and bull calves destined for live export. Backgrounding primarily occurred during the growing season, with cattle received between June and September and most sold off-farm by the end of March. This allowed the farmer to match feed demand to pasture supply, with cattle entirely grass-fed and no requirement for supplementary feed.

In addition to backgrounding pastoral cattle, Dongara also ran a small herd of 200 breeders of the British *Bos taurus* breed, Red Angus. Calving occurred at the start of the growing season in June, with calves weaned through December and January in

yards. Bull calves were sold directly onto boats for live export while heifer calves destined for sale remained on pasture for six to eight months, or until they met domestic feedlot specifications. The remaining heifers joined the breeding herd as replacements, calving for the first time at two years. Cull cows and bulls were sold annually into the domestic trade.

Dongara had only in recent years converted to cattle production. Until the 1980s and 1990s they had run a mixed cropping and sheep production enterprise. Located on poor quality, deep sands, the farm struggled to maintain annual crops and pasture productivity, encountering issues such as herbicide resistance, erosion, water logging and diseases. To alleviate these issues, the perennial fodder shrub tagasaste (*Chamaecytisus palmensis*) was planted in the late 1980s-early 1990s, followed by subtropical perennial grasses in 2001. Several subtropical perennials had been trialled since initial establishment, with Rhodes grass (*Chloris gayana*) and Gatton panic (*Panicum maximum*) found to persist best in the local conditions. The introduction of perennials and a rotational grazing strategy doubled the carrying capacity of the farm, enabling a transition from crop and sheep to cattle production.

The arable component of the farm (3,000 ha) was comprised of three pasture types;

- subtropical perennial grasses (1,600 ha) containing Rhodes grass, Gatton panic, sub clover, blue lupin (*Lupinus cosentinii*), annual ryegrass, capeweed and wild radish (*Raphanus raphanistrum*);
- perennial fodder shrubs (450 ha) comprised of double rows of tagasaste with annuals (as below) in the inter-row; and
- annuals (approx. 950 ha) containing serradella, sub clover, blue lupin, annual ryegrass, wild radish and capeweed.

Together, these three pasture types ensured that the farm had green feed year-round and extended the primary growing season beyond the regional average by two months, to occur from April to November.

4.4.2 Livestock information

4.4.2.1 Breeding herd

The Red Angus breeding herd at Dongara comprised of 200 cows and 5 bulls with average mature weights of 600 kg and 875 kg, respectively (Table 4.14). The primary products from the breeding herd included weaner bulls destined for export and heifers destined for domestic trade, however the farm also sold cull cows and bulls. Joining occurred from September to October, with calving the following June to July. In

February, all cows were pregnancy tested and empty cows sold. On average 30 cull cows were sold at a liveweight of 600 kg. It was assumed that the one cull bull was also sold at a liveweight of 875 kg. On average, 30 replacement heifers were sourced annually from the breeding herd. One replacement bull aged two years, with a liveweight of 650 kg, was purchased prior to joining.

Table 4.14 - Characteristics of the Red Angus breeding herd and annual veterinary inputs at the Dongara beef enterprise

Input	Unit	Bull[a]	Cow	Replace-ment heifer	Bull calf	Heifer calf[b]	Total
Breed				Red Angus			
Joining dates				1 Sep – 31 Oct			
Calving date				1 Jun			
Weaning date		-	-	-	1 Dec	1 Jan	
Stock count	hd	5	200	30	80	80	
Age	months	24 ->36	>36	7-36	0-6.75	0-14	
LW	kg	650-875	600	200-600	34-215[c]	34-350[c]	
Growth rate[d]	kg/hd/day	0.00-0.67	0.00	0.18-0.71	0.84-1.81	0.68-1.81	
Sale information							
Sale date		1 Feb	1 Feb	-	21 Dec	7 Jan	
Sale count	hd	1	30	-	80	50	
Sale LW	kg	875	600	-	215	350	
Total LW sold	kg	875	18000	-	17200	17500	53575
Veterinary product application							
Vaccination	ml	25	1800	1159	-	950	3934
Drench	ml	440	12000	2543	-	1500	16483
Veterinary product transportation[e]							
Vaccination	tkm						51
Drench	tkm						328

Hd = head LW = liveweight tkm = tonne-kilometres

[a] Includes the one replacement bull purchased at two years and a liveweight of 650 kg.

[b] Includes both heifer calves sold and calves (prior to weaning) destined to become replacement heifers.

[c] Where the birthweight of 34 kg was sourced from (Winder et al., 1990).

[d] As detailed in Chapter Three, mature cattle (>36months) were assumed to have a zero net liveweight change unless otherwise specified by the farm. Required growth rates of growing animals to meet farm specifications were calculated from farm-specific information.

[e] Total transportation of veterinary products includes transportation of products used by both the breeding herd and the backgrounding herd detailed in Table 4.15.

It was assumed that calving occurred at the beginning of June. On average 160 calves, 80 bulls and 80 heifers, were weaned annually, yielding an 80% weaning rate. Following Winder, Brinks, Bourdon and Golden (1990) the average birthweight of the calves was assumed to be 34 kg. In the absence of differentiated growth rate information for bull calves versus heifer calves, it was assumed that the average growth rate to weaning of all unweaned calves was 0.84 kg/head/day. Depending on the supply to boats, the bull calves could remain on-farm through to January.

However, for the purposes of this study it was assumed that bull calves were weaned in yards over a week at the beginning of December, at an average liveweight of 175 kg. They then spent an additional two weeks in the yards, before being sold directly for export at an average weight of 215 kg. Heifer calves were weaned in January, a month after the bull calves, at a liveweight of 200 kg. They spent a week in the yards before being moved back onto pasture. The 50 weaned heifers destined for sale remained on-farm until they reached a target liveweight of 350 kg, at approximately 14 months, before being sold to a domestic feedlot.

The 30 heifers designated for the breeding herd were joined over that year's joining period, at 15 months of age. Following industry recommendations for joining weights, the target liveweight at joining was assumed to be 360 kg, 60% of mature liveweight (DairyNZ, 2019; MLA, 2017). Calving occurred the following June, when the heifers were two years of age and an average liveweight of 540 kg. In May, all breeding stock were drenched and vaccinated. Breeding cows and replacement heifers were also provided with additional vaccinations prior to joining and calving, respectively.

4.4.2.2 Backgrounding cattle

Dongara backgrounded on average 3,000 cattle during the growing season, receiving pastoral heifers, steers and weaner bulls from May through to September. Pastoral cattle were then sold from October through to April to targeted markets, with the farm receiving a portion of the sale price as a share of the liveweight gained on the farm. The pastoralists received an amount equivalent to the initial weight onto the Dongara farm and a smaller portion of the liveweight gained on the farm.

Though cattle arrived for backgrounding throughout the growing season and were sold gradually depending on liveweight attained, for the purposes of this study it was assumed stock were received and sold on set dates. These dates were determined using farm information; including average duration each stock class remained on-farm, stock movements over the backgrounding period and target sale weights (Table 4.15). In total, pastoral cattle gained approximately 388,500 kg on the Dongara enterprise, representing over 40% of the total pastoral liveweight sold off-farm.

Pastoral heifers arrived on-farm at an average liveweight of 200 kg and, over an average of nine months, attained a target liveweight of 350 kg (liveweight gain of 0.60 kg/head/day) before being sold to domestic feedlots. Pastoral steers arrived at an average 250 kg and were sold after four to seven months to both the domestic and live export market at liveweights ranging from 343 kg to 400 kg (liveweight gains of 0.83 and 0.68 kg/head/day, respectively). Pastoral weaner bulls were retained on-

farm for four months prior to sale at an average 225 kg to the live export market after entering the farm at 125 kg (liveweight gain of 0.89 kg/head/day).

Table 4.15 - Characteristics of the backgrounding herd at the Dongara beef enterprise

Input	Unit	Pastoral heifer	Pastoral steer	Pastoral weaner bull	Total
Breed		Brahman, Droughtmaster, Santa Gurtrudis			
Stock count	hd	1200	900	900	
Date at entry		1 June	1 July	1 August	
LW at entry	kg	200	250	125	
Growth rate[d]	kg/hd/day	0.60	0.68-0.83	0.89	
		Sale information			
Sale date(s)		1 Mar, 1 Apr	1 Nov, 1 Feb	1 Dec	
Sale count	hd	1200	900	900	
Sale LW	kg	350, 365	343, 400	225	
Total LW sold	kg	432800	331500	202500	966800
Total sold LW gained on farm	kg	192000	106500	90000	388500
		Veterinary product application[a]			
Vaccination	ml	3000	2250	2250	7500
Drench	ml	25200	23400	8610	57210

Hd = head LW = liveweight tkm = tonne-kilometres

[a] Transportation of veterinary products applied to backgrounding cattle is included in the total provided in Table 4.14.

The first consignment of pastoral cattle, 1,200 heifers, was assumed to arrive at the beginning of June, followed by 900 steers at the beginning of July and 900 weaner bulls at the beginning of August. This meant at the peak period of green feed availability in September, on average 3,000 pastoral cattle were on-farm. At the end of October, 500 steers were sold at 343 kg. The 900 weaner bulls were sold over November and December, assuming an average sale date at the beginning of December. The remaining 400 steers were sold at the beginning of February, at 400 kg. This ensured that by February the farm met its target count of 1,200 pastoral cattle on-farm. To match stock demand with feed availability, the farm targeted 800 pastoral cattle on-farm by March. As such, it was assumed that 400 heifers were sold at the beginning of March, weighing an average 350 kg. The remaining 800 heifers were assumed to be sold at the beginning of April at 366 kg. All pastoral cattle were drenched and vaccinated upon arrival onto the farm.

4.4.3 Pasture information

At the time of this study, 2,050 ha of arable land had been converted to perennial grasses and shrubs. The 950 ha of annual pasture was sown on average every five years with French serradella cultivars, Cadiz and Margurita. Serradella was sown directly into the paddock without any pre-sowing treatment. Using farm machinery

with purchased seed, it was applied at a rate of 7.5 kg/ha in May. Following seeding and annually ongoing, 70 kg/ha of super phosphate was applied (Table 4.16).

Table 4.16 - Characteristics and annual inputs of the three pasture types at the Dongara beef enterprise

Input^	Unit	Annual pasture	Perennial grass pasture	Tagasaste pasture	Total
Predominant soil type		Sand plains; deep sands and sand over gravel			
Area	ha	950	1600	450	
Years since establishment	yr	5[a]	2-13	12-30	
Month sown		May	August	September	
Sowing rate	kg/ha	7.5	3	0.45[b]	
Chemical application					
Herbicide					
Glyphosate	l	-	457	-	457
2, 4-D	l	-	103	-	103
Pesticide					
Alpha-cypermethrin	l	-	23	-	23
Fipronil	l	-	1.4	0.15	1.55
Fertiliser					
Super phosphate	kg	79800	112000	31500	223300
Phosphate manganese	kg	-	-	6429	6429
DAP	kg	-	12800	2250	15050
N fertiliser	kg	-	26667	-	26667
Dolomite	kg	63333	106667	30000	200000
Chemical transportation					
Herbicide					
Glyphosate	tkm	-	15164	-	15164
2, 4-D	tkm	-	3526	-	3526
Pesticide					
Alpha-cypermethrin	tkm	-	2704	-	2704
Fipronil	tkm	-	262	4	266
Fertiliser					
Super phosphate	tkm	46527	65302	18366	130195
Phosphate manganese	tkm	-	-	3422	3422
DAP	tkm	-	6036	1061	7097
N fertiliser	tkm	-	17576	-	17576
Dolomite	tkm	13300	22400	6300	42000
Seed transportation					
Annual	tkm	76	-	36	112
Perennial grasses	tkm	-	147	-	147
Tagasaste	tkm	-	-	3	3
Fuel consumption & transportation					
Tractors	l	1963	4591	1238	7792
Fuel transportation	tkm	150	351	95	596
Farm machinery production[c]					
Tractors	USD	2021	-	2075	4096
Seeder	USD	2779	-	1317	4096

^ Inputs are presented on an annual basis. Inputs applied at establishment x years ago are also considered in this table by adjusting the input value for the number of years since establishment.

[a] Initially established over 40 years ago. Serradella sown on average every five years.

[b] Tagasaste was seeded at a rate of 0.5 kg/km. At 20 m inter-row and 2 m double row of tagasaste, this equated to the tractor travelling 1 km/ha at seeding.

[c] Contracted machinery was excluded from machinery production calculations.

Following initial establishment of subtropical grasses in 2001, more land had been progressively converted to perennial grasses. Prior to sowing in September, paddocks were prepared by spading and mould-boarding soil, followed by applications of single knockdown herbicides and a broad-spectrum insecticide. Using contract machinery, Gatton panic and Rhodes grass seed was applied at rates of 2 kg/ha and 1 kg/ha, respectively. Post-emergence, a broadleaf herbicide and an insecticide targeting grasshoppers was applied. Fertiliser was not applied during the year of establishment. Instead, pasture was fertilised with super phosphate in the following July, and each July thereafter. In addition to this, two types of N fertilisers were applied when pasture became N deficient following a drop in the proportion of legumes in the pasture or following significant spring growth. These events occurred on average every four years.

Following the introduction of tagasaste in the late 1980s and early 1990s, the perennial fodder shrub had been planted as recently as the mid-2000s. Tagasaste was sown into existing annual pasture in a 2 m wide double row configuration with 20 m inter-rows. As farm information was unavailable for planting density, following Thomas et al. (2015) tagasaste plants were allocated a spacing of 0.7 m with an establishment density of approximately 1,299 trees/ha. The tagasaste component was thus calculated to comprise almost 9% of the paddock area, whilst annual inter-rows the remaining 91%.

Dongara established tagasaste from seed and assumed to occur in winter (Wiley, 2005b). Seeding was conducted by contractors using specialist machines which scalp away the topsoil and rip below the planted seed, removing the need for chemical weed control prior to seeding. Di-ammonium phosphate (DAP) fertiliser was applied as the soil was ripped. Super phosphate was applied annually to both annual and perennial grass pastures. Every seven to eight years a manganese super phosphate mix was applied to address manganese deficiency in the tagasaste. In the summer of the establishment year, insecticide targeting grasshoppers was applied. The tagasaste was trimmed every three years by contractors with specialist machinery.

The farm had also planted smaller areas to native shrubs such as rhagodia (*Rhagodia* spp.) and wattles (*Acacia* spp.), however due to lack of inventory data, these were omitted from the analysis.

All pasture was on a liming program with dolomite applied to 200 ha annually on average at a given rate of 1 t/ha. The liming interval for each paddock was calculated to be 15 years.

4.4.3.1 Pasture attributes

The attributes of the three pasture systems were obtained using different methods.

Annual pasture data sources

Monthly annual pasture attributes were obtained from GrassGro by modelling the Dongara annual pasture system over a 30-year period (1985-2014). Per the approach at Bremer Bay, in lieu of a parameter set for serradella, it was assumed sub clover was the dominant legume in the sward. Sub clover typically struggles to persist in the deep sands of the NAR in WA (Nichols et al., 2007). However, as there was no suitable legume alternative in GrassGro and given that sub clover already comprised a portion of the annual sward (10-15%, farm information), it was decided that the modelled annual pasture would include sub clover, annual ryegrass and capeweed.

Subtropical grass pasture data sources

At the time of the study, aside from the non-performing Kikuyu parameter set, no subtropical grasses were available in GrassGro. Instead, subtropical grass growth rates for the Mingenew-Irwin region, in which Dongara was located, were obtained from DPIRD (R. Verbrugge, unpublished data, 2016). These were used alongside feed intake calculations and wastage factors, to calculate monthly DMA values. The DMD and CP content of Rhodes grass, Gatton panic and annual components of the perennial grass pasture were obtained from a local study (Moore et al., 2009b).

Tagasaste pasture data sources

The tagasaste pasture attributes were calculated from both the tagasaste and annual inter-row components. Annual inter-row data was obtained from GrassGro output. Monthly DMA values of the tagasaste double rows were calculated using regional EDM growth rates sourced from DPIRD (R. Verbrugge, unpublished data, 2016) adjusted for local planting density, feed intake calculations and wastage. The regional annual NPP of tagasaste was calculated to be 3,776 kg EDM/ha/yr. To apply this information to Dongara, annual EDM production per plant was obtained by adjusting for planting density per hectare. Regionally recommended planting specifications are 2 m double row with 10 m inter-rows (B. Wilson, pers comm), or 2,381 plants/ha (assuming 0.7 m spacing). This equates to an average annual production of 1.59 kg EDM/plant/yr and compares well to the measured tagasaste production of 1.50 kg EDM/plant/yr by a long-term study conducted in the NAR under similar conditions (Oldham et al 1991/1988). DMD and CP contents were obtained from industry publications and southern Australia studies (Moore et al., 2006; Wiley, 2005a).

Calculated pasture attributes

Figure 4.5 compares the monthly growth rates of the three pasture types at Dongara and demonstrates the value of perennial pastures to supply out-of-season feed. From December through to April when the annual pasture was senescent, both the subtropical grasses and tagasaste continued to provide green feed. In the autumn months, when the quality of the annual pasture had declined to below that required to meet animal maintenance requirements (Table 4.17), the growth rate of tagasaste, in particular, increased, providing a valuable source of feed. The growth rates of the perennial species declined over the cooler months, allowing the annual pasture components to dominate and maintain high levels of productivity through the growing season. As the annual pasture senesced at the end of the growing season, the growth rate of the perennials remained high, and in the case of the subtropical grasses, increased into the summer months. Figure 4.5 shows that with the introduction of perennials into the Dongara pasture system, the farm was able to extend their growing season by at least a month on either side of the historical June to October season.

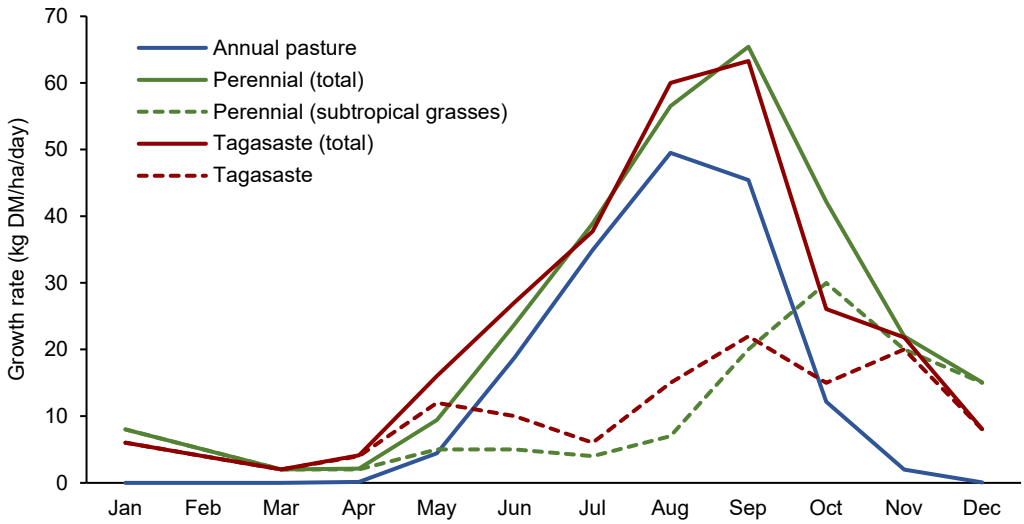


Figure 4.5 - Comparison of the average monthly growth rates (kg DM/ha/day) of the three pasture types; annual, subtropical grasses and tagasaste, at the Dongara beef enterprise.

Where: Annual pasture growth rates (blue solid line) sourced from site-specific GrassGro modelled outputs. Perennial grass pasture (green solid line) rates sourced from DPIRD (R. Verbrugge, unpublished data, 2016) with the subtropical grass component denoted by the green dashed line. The tagasaste pasture growth rates (red solid line) calculated from regional data (DPIRD) with the tagasaste component denoted by the red dotted line.

Using these growth rates, the annual NPP of annual pasture was calculated to be 5.12 t DM/ha, perennial grass pasture 9.92 t DM/ha and tagasaste 6.72 t DM/ha (Table 4.17). The subtropical grasses contributed 3.75 t DM/ha, or 38% of the total annual biomass of perennial grass pasture. The annual NPP of the tagasaste plants

was calculated to be 2.06 t EDM/ha or 31% of total biomass. Of total biomass produced by annual pasture, over 95% was produced during the annual growing season, from June to October. While overall, the perennial grass and tagasaste pasture produced 77% and 83% of annual biomass within these months, respectively, individually the subtropical grasses produced only 54% and tagasaste plants 55%.

Table 4.17 - Productivity and feed quality attributes of the three pasture types grazed at the Dongara beef enterprise

Input^	Unit	Annual pasture		Perennial grass pasture		Tagasaste pasture	
		Mean	**SD**	**Mean**	**SD**	**Mean**	**SD**
Length of growing season		Mid-Jun - Oct		Jan - Dec		Jan - Dec	
Annual NPP[a]	t DM/ha	5.12	-	9.92	-	6.72	-
Growing season NPP[b]	t DM/ha	4.92	-	7.67	-	5.61	-
Non-growing season NPP[b]	t DM/ha	0.20	-	2.26	-	1.11	-
Available DMA (% green)[c]							
Start of growing season	t DM/ha	0.76 (19%)	*0.00*	2.05	*0.00*	1.49	*0.00*
Peak growing season	t DM/ha	2.29 (79%)	*0.81*	4.57	*1.39*	3.27	*0.99*
End of growing season	t DM/ha	3.25 (37%)	*0.00*	5.51	*0.00*	4.61	*0.00*
Non-growing season	t DM/ha	1.59 (1%)	*0.61*	2.39	*0.95*	2.20	*0.69*
DMD[d]							
Start of growing season	%	80	*0.00*	71	*0.00*	73	*0.00*
Peak growing season	%	74	*3.00*	67	*0.99*	72	*0.82*
End of growing season	%	66	*0.00*	65	*0.00*	69	*0.00*
Non-growing season	%	51	*4.00*	63	*0.90*	67	*1.81*
CP content[d]							
Start of growing season	%	29	*0.00*	23	*0.00*	25	*0.00*
Peak growing season	%	24	*2.05*	22	*1.26*	24	*0.47*
End of growing season	%	19	*0.00*	18	*0.00*	21	*0.00*
Non-growing season	%	11	*2.00*	13	*1.25*	17	*0.88*

^ With the exception of annual net primary productivity (NPP), all averages and standard deviations calculated only from months feedbase was grazed within that period.

[a] GrassGro modelled output was obtained for annual pasture and tagasaste pasture annual inter-row. Subtropical grass and tagasaste values calculated using regionally specific information from DPIRD (R. Verbrugge, unpublished data, 2016) adjusted for the Dongara specifics.

[b] Where growing season refers to the annual species growing season at the case study farm location.

[c] GrassGro provided values for both total available DMA and green DMA, the regional perennial data only provided overall values.

[d] DMD and CP content values for annual pasture and tagasaste pasture annual inter-row obtained from GrassGro. Subtropical perennial grass attributes sourced from Moore et al. (2009b) , while tagasaste attributes were obtained from (Moore et al., 2006; Wiley, 2005a).

Figure 4.6a (Section 4.4.5.1) displays the calculated monthly DMA, DMD and CP content values for the three pasture types. Reflecting the growth rates outlined above, the DMA values for the annual pasture were highest during the growing season (averaging 2.29-3.25 t DM/ha) with average DMD and CP contents ranging from 80%

and 29% at the beginning of the growing season to 66% and 19% by October. The quantity and quality of the senesced pasture declined considerably over the non-growing season; DMA fell below 1.00 t DM/ha, DMD to 45% and CP content to 8% by May. As surmised by the farmer, this sharp decline likely reflects the poor soil quality of the region and poor persistence of traditional annual pasture species.

Subtropical grass and tagasaste pasture followed a similar trend to annual pasture, with higher DMAs, DMDs and CP contents during the growing season and lower values in the non-growing season. This reflects the contribution of annual species to each pasture type. However, during the non-growing season, the DMAs of the subtropical grasses and tagasaste averaged 2.39 and 2.20 t DM/ha, reflecting the contribution of green feed by the perennial species within the sward. This supply of green feed throughout the year resulted in higher average pasture quality, with the subtropical grass pasture grazed by cattle providing between 62%-71% DMD and 12%-23% CP content during the production year and between 63%-73% and 15%-25% for tagasaste pasture (Figure 4.6a). In particular, the average non-growing season quality was able to satisfy animal requirements; 63% DMD and 13% CP content for subtropical grasses and 67% and 17% for tagasaste, respectively.

4.4.4 Grazing management

Pasture was planted in a “wagon wheel” structure, whereby each wheel involved inter-dispersed paddocks of annuals, perennials grasses and tagasaste through which cattle were rotated throughout the production year. The breeding herd and pastoral cattle were managed to ensure that grazing demand matched pasture supply. For example, during the growing season the farm supported lactating cows along with pastoral cattle. As annual pasture senesced and carrying capacity declined, calves were weaned and sold, and pastoral cattle sold.

All livestock were provided mineral licks from October to May. Mineral licks were produced on-farm and contained lupins, salt, molasses, urea, dolomite and biochar. All ingredients were purchased and approximately five tonnes produced annually (Table 4.18). The mineral licks provided a source of protein and scarce minerals when pasture was of low quality. In addition to the licks, mineral supplements were dissolved in water and provided to livestock at each watering hub throughout the production year. No other supplementary feed was provided.

Calves from the breeding herd were weaned in the yards over a week in December and January. Weaner bulls were retained for a further two weeks prior to sale to live export, while weaner heifers were moved back onto pasture following weaning. While

in the yards, weaner cattle were supplied with high-quality pellets and hay (Table 4.18). Assuming a combined ration comprised of 80% pellets and 20% hay, the weighted DMD and CP content of the ration was 82% and 18%, respectively. In addition to weaner cattle, all cattle sold, from the breeding and backgrounding herds, were moved to the yards, for a few days prior to sale, where they were supplied hay.

Table 4.18 - Feed quality attributes and annual inputs of the supplementary feed and feedlot ration supplied at the Dongara beef enterprise

Attribute or input^	Unit	Annual pasture	Perennial grass pasture	Tagasaste pasture	Total
			Supplementary feed attributes[a]		
Months provided			Oct-May		
DMD					
Mineral lick	%		38		
CP content					
Mineral lick	%		48		
			Supplementary feed input[b]		
Mineral lick	kg	1585	2663	752	5000
Mineral supplement in water	l	63	107	30	200
			Supplementary feed transportation		
Mineral lick	tkm	522	877	248	1647
Mineral supplement in water	tkm	282	474	133	889
			Feedlot ration attributes[c]		
DMD					
Pellets	%		88		
Hay	%		59		
CP content					
Pellets	%		21		
Hay	%		8		
			Feedlot ration input[b]		
Pellets	kg				16901
Hay	kg				100553
			Feedlot ration transportation		
Pellets	tkm				7420
Hay	tkm				2703

^ Inputs are presented on an annual basis.

[a] The mineral lick was produced on-farm and contains lupins, urea, molasses, salt, dolomite and biochar. All ingredients were sourced off-farm and the weighted DMD of all components is presented in this Table.

[b] Presented on an "as-fed" DM basis with an assumed wastage of 20%.

[c] DMD and CP content of pellets and hay sourced from product specifications and from (DAFWA, 2006a).

4.4.5 Calculated on-farm inventory outputs

The information collected for Dongara was used to calculate the on-farm inventory outputs presented in Table 4.19.

<u>Enteric methane emissions</u>

Enteric methane contributed a total of 112,507.10 kg CH_4/year, with 2.54×10^{-1} kg CH_4/kg LW produced for sale. As expected, the cattle enterprise produced substantially more enteric methane than the sheep enterprises; more than double that

of Wickepin and almost nine-fold more than Bremer Bay. In terms of saleable liveweight production, Dongara was more efficient at producing saleable product than Wickepin but less than Bremer Bay. Such comparisons must be made with caution as different methods were applied to calculate enteric methane for sheep and cattle.

It was possible, however, to compare the emissions efficiency of the three pasture systems present at Dongara. Each production year, annual pasture was the least efficient (3.38×10^{-1} kg CH_4/kg LW), followed by the subtropical grass pasture (2.58×10^{-1} kg CH_4/kg LW), with the tagasaste pasture most efficient (1.77×10^{-1} kg CH_4/kg LW). Unsurprisingly, the feedlot produced saleable product most efficiently, emitting only 1.59×10^{-1} kg CH_4/kg LW. Unlike the sheep enterprises which did not graze annual pasture during the dry season, annual pasture at Dongara was grazed year-round. This meant that during the dry season when annual pasture was of declining quality (Figure 4.6a), it was unable to support the daily requirements of cattle. Instead, perennial grasses and tagasaste supported saleable liveweight production during this period. Despite this, annual pasture was still grazed during this non-growing period by all cattle, resulting in the production of methane not offset by saleable liveweight.

In contrast to annual pasture, tagasaste, which covered half the area and thus grazed less stock through the year, was able to support the growth of livestock throughout the production year with 21% of whole-farm saleable liveweight produced on this pasture (Table 4.20). Considering that annual pasture also produced 21% of saleable liveweight, this highlights the productivity of the tagasaste. Finally, subtropical grasses, which supported the majority of the herds' grazing requirements throughout the year and thus produced the highest emissions in terms of total liveweight grazed (1.60×10^{-2} kg CH_4/kg LW grazed), was still 24% more efficient at producing saleable product than the annual pasture. Section 4.4.5.1 explores this further.

<u>Manure methane emissions</u>

Total manure methane output from Dongara represented a small portion of animal emissions, contributing 26.14 kg CH_4/year. Compared to the sheep enterprises, the farm produced 5.91×10^{-5} kg CH_4/kg LW produced for sale, less than Wickepin but more than Bremer Bay. Following the same trend as for enteric methane, annual pasture was the least emissions efficient and tagasaste the most efficient, with tagasaste producing 52% less manure methane per kilogram of saleable liveweight. Examining emissions in terms of liveweight grazed, annual pasture produced the least emissions, consistent with the lower average annual quality of annual pasture compared to the two perennial pastures (Figure 4.6a).

Table 4.19 - Calculated annual animal, soil and associated on-farm emissions produced on the Dongara beef enterprise

Output	Unit	Annual pasture	Perennial grass pasture	Tagasaste pasture	Feedlot	Total
			Animal emissions			
Enteric CH_4	kg CH_4/yr	30774.85	63403.10	16496.38	1832.80	112507.14
	kg CH_4/kg LW grazed[a]	1.30E-02	1.60E-02	1.46E-02	1.31E-02	1.48E-02
	kg CH_4/kg LW produced for sale[b]	3.38E-01	2.58E-01	1.77E-01	1.59E-01	2.54E-01
Manure CH_4	kg CH_4/yr	7.09	15.12	3.47	0.46	26.14
	kg CH_4/kg LW grazed	2.99E-06	3.81E-06	3.08E-06	3.25E-06	3.43E-06
	kg CH_4/kg LW produced for sale	7.78E-05	6.14E-05	3.72E-05	3.97E-05	5.91E-05
			Direct soil emissions			
N_2O excreta	kg N_2O/yr	272.25	492.62	154.18	32.55	951.60
	kg N_2O/kg LW grazed	1.15E-04	1.24E-04	1.37E-04	2.32E-04	1.25E-04
	kg N_2O/kg LW produced for sale	2.99E-03	2.00E-03	1.65E-03	2.83E-03	2.15E-03
N fertilisers N_2O	kg N_2O/yr	-	34.06	1.27	-	35.33
	kg N_2O/ha	-	2.13E-02	2.83E-03	-	1.72E-02
	kg N_2O/kg LW produced for sale	-	1.38E-04	1.36E-05	-	7.99E-05
Crop residue N_2O	kg N_2O/yr	-	-	-	-	-
	kg N_2O/ha	-	-	-	-	-
	kg N_2O/kg LW produced for sale	-	-	-	-	-
Pasture residue N_2O	kg N_2O/yr	23.26	53.09	10.02	-	86.38
	kg N_2O/ha	2.45E-02	3.32E-02	2.23E-02	-	2.88E-02
	kg N_2O/kg LW produced for sale	2.55E-04	2.16E-04	1.07E-04	-	1.95E-04
			Indirect soil emissions			
Atmospheric deposition N_2O	kg N_2O/yr	54.45	101.94	30.97	6.51	193.85
	kg N_2O/kg LW produced for sale	5.98E-04	4.14E-04	3.32E-04	5.66E-04	4.39E-04
			Other calculated on-farm emissions			
Liming CO_2	kg CO_2/yr	28679.44	48302.22	13585.00	-	90566.67
	kg CO_2/ha	3.02E+01	3.02E+01	3.02E+01	-	3.02E+01
	kg CO_2/kg LW produced for sale	3.15E-01	1.96E01	1.46E-01	-	2.05E-01
Urea hydrolysis CO_2	kg CO_2/yr	-	7040.00	-	-	7040.00
	kg CO_2/ha	-	4.40E+00	-	-	4.40E+00
	kg CO_2/kg LW produced for sale	-	2.86E-02	-	-	1.59E-02

LW = liveweight

[a] To enable comparison across feedbases, animal emissions were adjusted according to output per kg of liveweight grazed on each.

[b] Using the functional unit, overall emissions were adjusted according to output per kg of liveweight produced for sale on each pasture.

Nitrous oxide emissions from excreta

Excreta nitrous oxide emissions at Dongara totalled 951.60 kg N_2O/yr, or 2.15×10^{-3} kg N_2O/kg LW produced for sale. Per the other animal emission sources, this meant Dongara produced saleable liveweight at a greater efficiency than Wickepin, but less than Bremer Bay. Nitrous oxide emissions from stock grazing tagasaste were 44% less than annual pasture and 18% less than subtropical grasses. Unlike for enteric methane and manure methane, in terms of total livestock grazed, subtropical grasses (1.24×10^{-4} kg N_2O/kg LW grazed) did not produce the highest emissions, rather tagasaste did, producing 1.37×10^{-4} kg N_2O/kg LW grazed. This was because though subtropical grasses had high DMA and DMD levels through the year, proportionally CP content was lower (Figure 4.6a), reducing CP intake relative to tagasaste.

Other soil emissions

Dongara did not grow crops so only nitrous oxide emissions from pasture residue was considered. In total, 86.38 kg N_2O was produced by the three pasture types. On a per hectare basis, subtropical grasses produced the highest emissions (3.32×10^{-2} kg N_2O/ha), followed by annual pasture (2.45×10^{-2} kg N_2O/ha) and then tagasaste (2.23×10^{-2} kg N_2O/ha). A direct comparison between pastures across enterprises revealed that the annual and perennial pastures at Dongara produced lower emissions on a per hectare basis than both sheep enterprises, a result of both the lower annual NPP at Dongara and a closely managed grazing system. This also meant that in terms of liveweight sold, pastures at Dongara produced only 1.95×10^{-4} kg N_2O/kg LW produced for sale, lower than both Bremer Bay and Wickepin. Comparing the respective efficiencies of the three pastures at Dongara, annual pasture was the least efficient, producing 2.55×10^{-4} kg N_2O/kg LW, attributed to the lower ability to produce saleable liveweight compared to the perennial pastures.

Nitrous oxide emissions from the application of N fertiliser totalled 35.33 kg N_2O/yr (0.8% of total on-farm N_2O emissions) while emissions from atmospheric deposition totalled 193.85 kg N_2O/yr (4.4% of total N_2O emissions). Similar to Wickepin, the farm employed a periodic liming treatment, annually resulting in 90,566.67 kg CO_2, or 2.05×10^{-1} kg CO_2/kg LW. Dongara also produced carbon dioxide emissions from urea hydrolysis of fertiliser applied to subtropical grasses, comprising 7% of total on-farm carbon dioxide emissions. The farm did not experience leaching or runoff.

4.4.5.1 Analysis of the interactions between feedbase, animal intake and enteric methane production at the Dongara beef enterprise

Of the three pastures at Dongara, tagasaste pasture had the lowest enteric methane EI, followed by subtropical grass pasture and finally annual pasture. This Section

examines the factors that influenced these calculated emission outputs.

Annual pasture at Dongara was grazed year-round, unlike annual pasture on the sheep enterprises. Figure 4.6 demonstrates how the variation in annual pasture availability and quality across the production year influenced the intake of grazing cattle. During the non-growing season, feed intake of stock grazing annual pasture was increasingly restricted. Though this also meant that the production of enteric methane was lower, it came at a cost in terms of reduced productivity as annual pasture could not meet animal feed requirements. Without perennial pasture, all livestock would have to be supplementary fed. Due to the short growing season at Dongara, intake was restricted for over half the year, indicative of the burden that supplementary feeding would place upon the farm.

The rotational grazing strategy ensured that the surplus intake consumed on the subtropical grasses and tagasaste offset the deficit encountered on annual pasture during the non-growing season. The intake of subtropical grass pasture remained relatively constant across the production year, which can be attributed to its consistent DMD and high DMA, both of which do not restrict the relative intake of livestock. The variability of intake on tagasaste pasture was also lower than annual pasture, however there was a noticeable increase over the growing season. This is reflective of the influence of the annual component of tagasaste pasture during these months. Later Chapters explore the role of the perennials and annuals at the enterprise.

Across the stock classes at Lancelin, daily feed intake and ensuing enteric methane production was highest for mature bulls (11.68 kg DM/head/day, 0.242 kg CH_4/head/day) followed by lactating mature cows (10.28 kg DM/head/day, 0.213 kg CH_4/head/day; Table 4.20). The lowest values were attributed to bull and heifer calves produced by the farm's breeder herd. Across the three feedbases, the average daily values of all stock classes were highest on perennial grasses, followed by tagasaste and then annual pasture. Interestingly, the intake and enteric methane values of the bulls and lactating cows on annual pasture were almost the same; both restricted during the growing season by low DMA. The calculated values for bulls were further restricted by the poor quality of the senesced pasture during the non-growing season. This did not influence the values for lactating cows as calving occurred at the beginning of the growing season and weaning occurred prior to its completion.

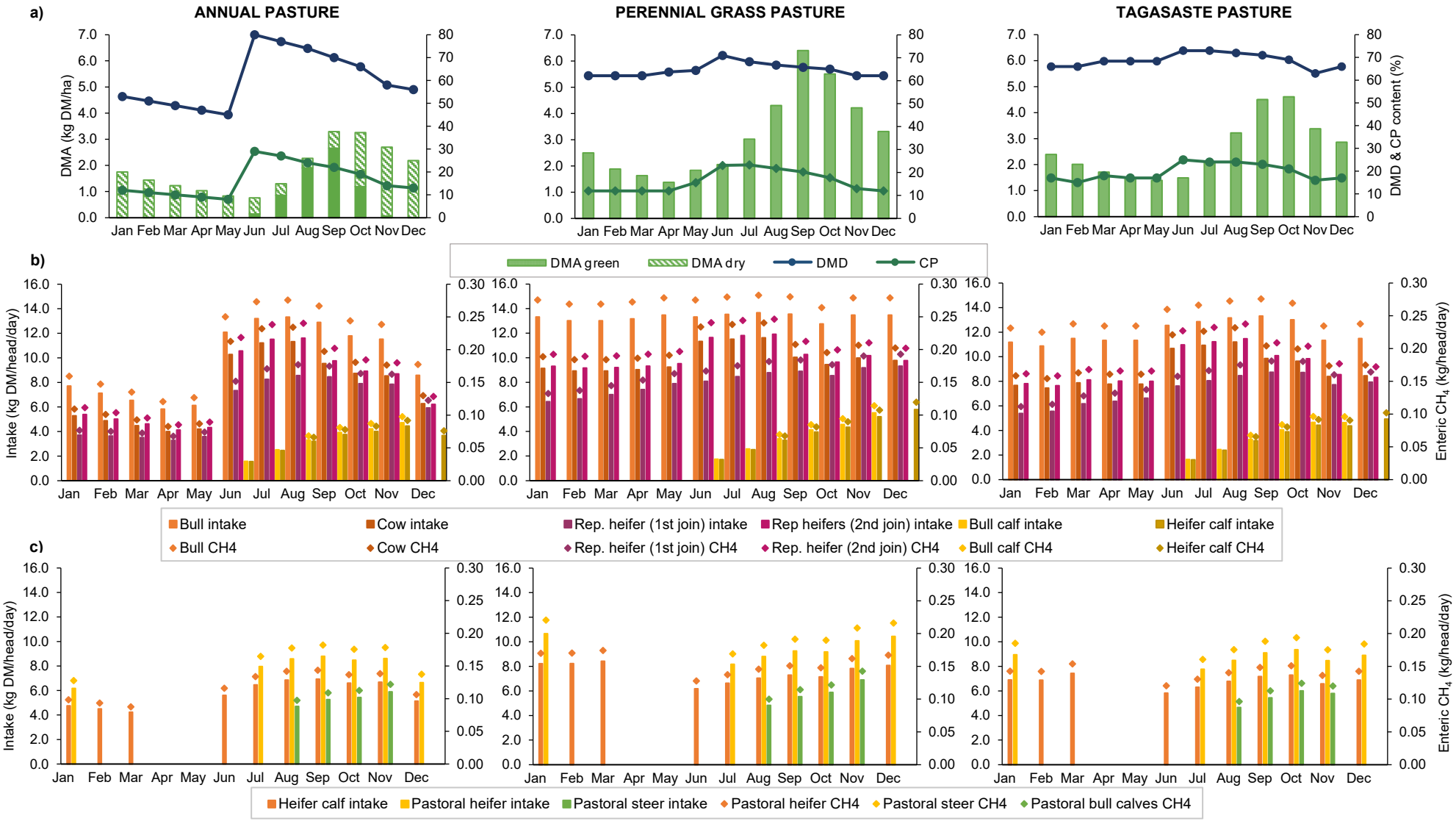

a)
ANNUAL PASTURE
PERENNIAL GRASS PASTURE
TAGASASTE PASTURE
DMA (kg DM/ha)
DMD & CP content (%)
Jan Feb Mar Apr May Jun Jul Aug Sep Oct Nov Dec
DMA green
DMA dry
DMD
CP
b)
Intake (kg DM/head/day)
Enteric CH4 (kg/head/day)
Bull intake
Cow intake
Rep. heifer (1st join) intake
Rep heifers (2nd join) intake
Bull calf intake
Heifer calf intake
Bull CH4
Cow CH4
Rep. heifer (1st join) CH4
Rep. heifer (2nd join) CH4
Bull calf CH4
Heifer calf CH4
c)
Intake (kg DM/head/day)
Enteric CH4 (kg/head/day)
Heifer calf intake
Pastoral heifer intake
Pastoral steer intake
Pastoral heifer CH4
Pastoral steer CH4
Pastoral bull calves CH4

Figure 4.6 (previous page) - Relationship between monthly feedbase attributes, feed intake and enteric methane production of cattle at the Dongara beef enterprise.

(Where; a) the feedbase attributes; dry matter availability (DMA, kg DM/ha/yr), dry matter digestibility (DMD, %) and crude protein content (CP, %); b) & c) the average daily feed intake (kg DM/head; solid colour is feedbase intake, patterned colour is supplementary feed intake) and average daily enteric methane production (kg/head, scatter plot) of each stock class within the breeding and pastoral herds, respectively. Note: Only annual pasture data modelled in GrassGro could distinguish between green and dry DMA, perennial pasture monthly values reflect total DMA only).

Dry mature cows consumed on average less feed and produced less enteric methane than breeding heifers on their second joining (11% higher feed intake, 10% higher enteric methane) and pastoral steers (8% higher for both intake and enteric methane), despite the increased liveweight of mature cows. Though a portion of the increased values of heifers will be due to lactation, a portion can also be attributed to the growth of the animals. This is also true for pastoral steers. This demonstrates why, in the context of emissions produced per kilogram of liveweight grazed, not all kilograms are created equal. Growing stock will produce more emissions per kilogram of liveweight than mature stock due to the increased intake requirements to meet daily requirements. If the growing cattle are destined for sale at Dongara, these increased emissions may be offset through a lower EI. However, if they are destined for the breeding herd, there is less chance that these emissions will be offset by liveweight produced elsewhere on the farm. This is explored in Chapter Five which compares the relative contributions of the pastoral and breeding herds to overall emissions produced at Dongara.

As expected, the intake and enteric methane production of each stock class varied between the three pasture types, with intake typically highest on the subtropical grasses and lowest on annual pasture. For example, for the days pastoral steers were grazed on subtropical grasses, average feed intake was 9.51 kg DM/head (0.197 kg CH_4/head). This was 8% higher than tagasaste pasture, averaging 8.72 kg DM/head (0.180 kg CH_4/head/day) and 17% higher than annual pasture, which averaged 7.89kg DM/head (0.163kg CH_4/head/day). The differences between pastures were less pronounced for calves produced by the breeding herd, with average daily intake by bull calves on subtropical grasses only 5% higher than tagasaste pasture and 7% higher than annual pasture. This was because calves only grazed annual pasture during the growing season when it was of higher quality and availability.

The calculated outputs presented in this Section help to explain the differences in the enteric methane EI between the three pasture types at Dongara. They demonstrate the trade-off between emissions and productivity. For example, annual pasture, while

presenting lower average enteric methane emissions, was compromised by its inability to support livestock production during the non-growing season, reducing its overall EI. Instead, liveweight gains during this period occurred on subtropical grasses and tagasaste, improving the EIs of these pastures. Chapter Five explores this further.

Table 4.20 - Daily feed intake and calculated enteric methane production across stock classes for the annual and perennial pastures at the Dongara beef enterprise

Input	Unit	Annual pasture		Perennial grass pasture		Tagasaste pasture		Total	
		Mean	SD	Mean	SD	Mean	SD	Mean	SD
Months grazed		Jan-Dec		Jan-Dec		Jan-Dec			
Proportion of LW produced for sale[a]	%	20		56		21			
				Daily intake					
Breeder herd									
Bulls	kg DM/hd	9.73	*2.86*	13.32	*0.26*	12.00	*0.87*	11.68	*2.28*
Cows (dry)	kg DM/hd	6.67	*1.96*	9.13	*0.18*	8.23	*0.59*	8.01	*1.56*
Cows (lact.)	kg DM/hd	9.74	*1.78*	10.90	*1.01*	10.21	*1.21*	10.28	*1.45*
Rep. heifers (1st join)	kg DM/hd	6.00	*2.17*	8.06	*0.94*	7.28	*1.15*	7.11	*1.74*
Rep. heifers (2nd join)	kg DM/hd	7.56	*2.78*	10.14	*1.01*	9.17	*1.39*	8.96	*2.17*
Bull calves	kg DM/hd	2.90	*1.54*	3.13	*1.73*	2.96	*1.59*	3.50	*1.16*
Heifer calves	kg DM/hd	3.30	*0.92*	3.83	*1.33*	3.54	*1.11*	3.56	*1.16*
Pastoral herd									
Heifers	kg DM/hd	5.78	*1.00*	7.51	*0.72*	6.80	*0.45*	6.70	*1.04*
Steers	kg DM/hd	7.89	*0.98*	9.51	*0.84*	8.72	*0.49*	8.71	*1.04*
Bull calves	kg DM/hd	5.33	*0.43*	4.63	*0.74*	5.48	*0.52*	5.53	*0.61*
				Daily enteric methane production					
Breeder herd									
Bulls	kg CH_4/hd	0.201	*0.058*	0.276	*0.005*	0.248	*0.018*	0.242	*0.047*
Cows (dry)	kg CH_4/hd	0.138	*0.040*	0.189	*0.004*	0.170	*0.012*	0.166	*0.032*
Cows (lact.)	kg CH_4/hd	0.202	*0.036*	0.226	*0.019*	0.211	*0.025*	0.213	*0.030*
Rep. heifers (1st join)	kg CH_4/hd	0.124	*0.043*	0.167	*0.017*	0.151	*0.024*	0.147	*0.036*
Rep. heifers (2nd join)	kg CH_4/hd	0.157	*0.056*	0.210	*0.020*	0.190	*0.029*	0.185	*0.045*
Bull calves	kg CH_4/hd	0.084	*0.039*	0.091	*0.044*	0.086	*0.012*	0.087	*0.013*
Heifer calves	kg CH_4/hd	0.079	*0.006*	0.093	*0.013*	0.086	*0.012*	0.086	*0.015*
Pastoral herd									
Heifers	kg CH_4/hd	0.120	*0.042*	0.155	*0.048*	0.141	*0.009*	0.139	*0.021*
Steers	kg CH_4/hd	0.163	*0.061*	0.197	*0.070*	0.180	*0.010*	0.180	*0.021*
Bull calves	kg CH_4/hd	0.110	*0.050*	0.120	*0.056*	0.113	*0.011*	0.114	*0.013*

Hd = head (livestock unit)

[a] Proportion of total liveweight produced for sale on each feedbase. In addition to pasture, 3% of total sold liveweight was produced in the yards.

Influence of management practices on intake and enteric methane production

The results presented in this Section allow for a preliminary comparison of the emissions impact of producing saleable liveweight through the breeding herd versus backgrounding herd at Dongara. Using the daily averages presented in Table 4.20 and the liveweights sold and gained as detailed in Tables 4.14 and 4.15, the average total emissions for the six months that a mature breeding cow is lactating and the six months and three weeks that the bull calf is on-farm prior to sale, is 51.99 kg CH_4 per bull calf sold. Considering this in terms of kilograms of bull calf liveweight sold this

totals 0.242 kg CH_4/kg LW. If enteric methane production during pregnancy is also included, this increases to 78.81 kg CH_4 or 0.367 kg CH_4/kg LW. The impact of a bull calf produced by a heifer was more, producing 52.71 kg CH_4 per calf or 0.245 kg CH_4/kg LW if only lactation was considered. If pregnancy was also included this increased to 79.89 kg CH4, or 0.372 kg CH_4/kg LW.

In contrast to the breeding herd, the total enteric emissions of backgrounding a pastoral steer was 37.8 kg CH_4, or 0.252 kg CH_4/kg LW. Similarly, the impact of backgrounding a pastoral heifer for ten months was 41.7 kg CH_4, or 0.252 kg CH_4/kg LW. Backgrounding pastoral bull calves was the most efficient, producing only 13.68 kg CH_4 per calf, or 0.137 kg CH_4/kg LW. In terms of enteric methane emissions produced on-farm, backgrounding cattle was more efficient in all instances. It must be noted that this ignores the impact of producing pastoral cattle prior to arrival on the farm, as this is considered outside the boundaries of this study.

4.5 Inventory of the Lancelin beef production enterprise

4.5.1 Farm overview

The Lancelin enterprise was located around 130km north of Perth (115°E, 31°S). One of the first farms to plant tagasaste in the NAR during the 1980s and 90s, at the time of this study, almost 50% of the farm's 2000 ha was established tagasaste pasture. Approximately 400 ha was dedicated to perennial grasses, 450 ha to annual pasture and 150 ha to laneways, tree belts and farm infrastructure. Lancelin had the highest annual rainfall of all the regions considered, with a 30-year average of 599 mm (Jeffrey et al., 2001). Like Dongara, the region was characterised by long and dry summers, with 87% of annual rainfall falling between May and November (Figure 4.4). The historical minimum and maximum average monthly temperatures were 9°C in July and 31°C in January. This resulted in a six-month growing season from late May to late November.

The enterprise had been historically cropped; however poor-quality deep sands resulted in low grain yields and challenges with soil erosion and fertility. To combat this, the farm had converted to sheep production but following the wool market crash in 1990, transitioned to cattle production. Previously focussed on agistment, the unreliability of cattle supply meant that the farm had altered focus to building their breeding herd.

The farm ran around 600 breeders of mixed *Bos taurus* and *Bos indicus* origins. Reflective of the expanding breeding herd, around half of these were either first-time

heifers or undergoing their first joining. Calving occurred between May and August, with calves weaned from the end of December through to February on pasture. Bull calves were sold directly to live export while heifer calves remained on-farm to join the breeding herd. Cull cows, heifers and bulls were sold into the domestic market. The enterprise also agisted cattle, predominantly Brahman breed. Pregnant heifers were received onto the farm at the end of the growing season, just prior to calving. The agistment herd grazed the out-of-season green feed supplied by perennials, reducing supplementary feed requirements.

Annual pasture was first established in the early 1980s with annual ryegrass and sub clover. The pasture had not been reseeded since and at the time of data collection for this study, consisted primarily of brome grass, annual ryegrass, capeweed and sub clover. The system was grass-dominant, with the legume component contributing less than 5% to the total sward. The perennial grass pasture system comprised of Gatton panic, Rhodes grass and annual grasses. The perennial grasses were first sown in 2003 into existing annual pasture. Following the initial success of the subtropicals, more annual pasture had been converted in the years following. The third pasture system was tagasaste, comprising almost 1000 ha. The tagasaste was sown into existing annual pasture on marginal land and since establishment, had more than doubled the farm's carrying capacity.

The introduction of both the tagasaste and subtropical grasses had extended the farm's growing season, providing a guaranteed source of green feed to meet livestock growth requirements from October until the end of December. In particular, the tagasaste was a reliable source of feed during the dry season feed gap, meeting cattle maintenance energy requirements until the following growing season.

4.5.2 Livestock information

4.5.2.1 Breeding herd

Lancelin's breeding herd comprised 15 bulls, 283 mature cows, 108 pregnant heifers and 177 unmated heifers (Table 4.21). The cows had average mature liveweight of 550 kg while the mature bulls weighed 850 kg. The primary product from the breeding herd was weaner bull calves destined for live export, however the farm also sold cull cows, heifers and bulls.

Joining occurred from August to October, with the ensuing calving period commencing the following May. For the purposes of this study it was assumed that calving occurred at the beginning of July. Prior to joining, five replacement bulls, aged two years, were purchased at 650 kg. In December, all mated mature cows, second-

joining heifers and first-joining heifers underwent pregnancy testing. Following this, at the beginning of February, 114 empty or non-performing cows and heifers were sold to local saleyards at an average combined sale weight of 464 kg. It was assumed that the five cull bulls, weighing on average 649 kg, were also sold in February.

Table 4.21 - Characteristics of the breeding herd and annual veterinary inputs at the Lancelin beef enterprise

Input	Unit	Bull[a]	Cow	Replace-ment heifer[b]	Bull calf[b]	Heifer calf[bc]	Total
Breed(s)			Red Angus, Gelbvieh, Murray grey, Brahman				
Joining date				1 Sep			
Calving date				1 Jul			
Weaning date					1 Jan (31 Jul)	1 Feb (31 Jul)	
Stock count	hd	15	283	285 (60)	165 (27)	174 (26)	
Age	months	24 ->36	>36	7-36	0-7 (1-10)	0-8 (1-12)	
LW	kg	650-850	550	215-550 (450+)	32.5-230 (49-273)	32.5-230 (48-300)	
Growth rate[d]	kg/hd/day	0.00-0.55	0.00	0.20-0.56	0.82-1.01 (0.82-0.92)	0.71 (0.43-0.87)	
Sale information							
Sale date		1 Dec	1 Feb	1 Feb	1 Feb (1 Feb)	-	
Sale count	hd	5	38	76	165 (27)	-	
Sale LW	kg	649	550	339-464	230 (273)	-	
Total LW sold	kg	3245	20900	25460	37950 (7371)	-	97110
Veterinary product application							
Vaccination	ml	350	3538	2535 (1038)	1815 (365)	1740 (325)	11706
Drench	ml	1275	15565	7764 (2760)	- (189)	- (182)	27735
Veterinary product transportation[e]							
Vaccination	tkm						55
Drench	tkm						114

Hd = head LW = liveweight tkm = tonne-kilometres

[a] Includes the five replacement bulls purchased at two years and a liveweight of 650 kg.

[b] Values within () refer to purchased stock brought in at the beginning of the calculation period.

[c] Heifer calves, prior to weaning, destined to become replacement heifers.

[d] As detailed in Chapter Three, mature cattle (>36 months) are assumed to have a zero net liveweight change unless otherwise specified by the farm. Required growth rates of growing animals to meet farm specifications are calculated from farm-specific information.

[e] Total transportation of veterinary products includes transportation of products used by both the breeding herd and the backgrounding cattle detailed in Table 4.22.

At calving, mature cows and heifers produced a combined 165 bull calves and 174 heifer calves with an average birthweight of 32.5 kg. This yielded an annual weaning rate of 92% for mature cows and 88% for heifers. At beginning January, at six months and liveweights of just over 200 kg, bull calves were fitted with nose rings to initiate weaning. They were then placed back on pasture with cows for three weeks, after

which they were separated and finished on hay for a week. Though sales can be staggered through December to March, for the purposes of this study all weaner bull calves were assumed to be sold at the beginning of February at 230 kg. Heifer calves were fitted with nose rings for weaning at the beginning of February and a month later, at 230 kg, joined the breeding herd. Following weaning in February, weaner heifers were first joined in at 13 months of age and an average liveweight of 300 kg. Empty heifers were sold the following February, with remaining heifers calving the following July at an average liveweight of 450 kg.

Lancelin also purchased 60 heifers to improve the genetic diversity of the herd. These heifers arrived in May with 27 bull and 26 heifer calves on foot. Due to lack of available information, it was assumed that the heifers and calves followed the same growth pattern as the farm-produced stock. All calves were weaned at the end of July at four months of age and liveweights of 135 kg. This enabled purchased heifers to achieve condition prior to the September joining. Purchased weaner bull calves remained on-farm until the following January when they were sold for export at an average sale weight of 275 kg. Weaned heifer calves remained on-farm to join the breeding herd.

All stock were drenched and vaccinated in April and May prior to calving. Bulls were provided with further vaccinations in August prior to joining, while cows and heifers were provided further vaccinations in October following joining. Bull calves were vaccinated in October and at weaning in December. Heifer calves were vaccinated in October and at weaning in January. Purchased cattle were provided with the same vaccines but at purchase in May, July at weaning and October.

4.5.2.2 Agistment cattle

In October, 265 pregnant Brahman heifers arrived for agistment at an average liveweight of 425 kg (Table 4.22). Calving was assumed to occur at the beginning of December. As information regarding distinguishing bull and heifer calves was unavailable, average growth rates were assumed at 0.86 kg/head/day. The agistment stock remained on-farm until the calves were weaned. The total calculated liveweight gained at Lancelin by the agistment cattle was 52,999 kg. The agistment cattle were drenched in April and vaccinated in October and April.

Table 4.22 - Characteristics of the agistment herd and annual veterinary inputs at the Dongara beef enterprise

Input	Unit	Agistment heifer	Agistment calf	Total
Breed		Brahman		
Stock count	hd	265	265	
Date at entry		1 Oct	1 Dec[a]	
LW at entry	kg	425	32.5[b]	
Growth rate	kg/hd/day	0.20-0.56	0.86[c]	
		Sale information		
Sale date(s)		-	-	
Total LW gained on-farm[d]	kg	15311	37688	52,999
		Veterinary product application[e]		
Vaccination	ml	663		663
Drench	ml	1325		1325

Hd = head LW = liveweight

[a] Calving date of pregnant agistment heifers.

[b] Assumed birthweight of calves following farm information.

[c] Stock breakdown between heifer and bull calves was unavailable. Average growth rates were adopted.

[d] Total liveweight gained on-farm during the calculation period.

[e] Transportation of the veterinary products applied to agistment cattle is included in the total provided in Table 4.21.

4.5.3 Pasture information

The characteristics and inputs for each pasture type at Lancelin are summarised in Table 4.23. The 450 ha of annual pasture had not been reseeded since initial establishment thirty years prior and there was no available chemical, seed or machinery information pertaining to this. It was expected that the contribution to the overall carbon footprint would be minor once annualised over thirty years and hence was excluded from the calculations. Annually following establishment, 100 kg of super potash was applied using farm machinery. No other chemicals were applied.

Subtropical grasses were first established on 62 ha in 2003. On average, every two years following this further land was sown to subtropical grasses, totalling 400 ha at the time of the study. Prior to each sowing in September, paddocks were prepared by hard grazing livestock followed by applications of single knockdown herbicide and broad-spectrum insecticide. Gatton panic and Rhodes grass seed were sown at a combined rate of 3 kg/ha. A single application of super potash was applied at establishment and thereafter annually. Two insecticides were applied post-emergence. All seeding and chemical applications involved farm machinery.

Most of the 1000 ha of tagasaste was sown between 1986 and 1991. Tagasaste plants were established into existing annual pasture in a double row 2 m apart with a 6 m inter-row. Assuming 0.7 m spacing between plants (Thomas et al., 2015), this

equated to 3,571 plants/ha. The tagasaste plants thus comprised 25% of the total paddock area and annual inter-rows the remaining 75%. Tagasaste seed was sown in winter at a rate of 0.5 kg/km, or 1.25 kg/ha, using specialist farm machinery. Super potash and manganese sulphate fertiliser were applied in the year of establishment.

Table 4.23 - Characteristics and annual inputs of the three pasture types present at the Lancelin beef enterprise

Input^	Unit	Annual pasture	Perennial grass pasture	Tagasaste pasture	Total
Predominant soil type			Deep, grey over yellow sands		
Area	ha	450	400	1000	
Years since establishment	yr	30	7[a]	25	
Month sown		-	Sep	Sep	
Sowing rate	kg/ha	-	3.00	1.25[b]	
			Chemical application		
Herbicide					
Glyphosate	l	-	114	-	114
Pesticide					
Cypermethrin	l	-	9	-	9
Fipronil	l	-	0.6	0.4	1
Dimethoate	l	-	14	-	14
Fertiliser					
Super potash	kg	45000	42857	100000	187857
Manganese sulphate	kg	-	-	5600	5600
			Chemical transportation		
Herbicide					
Glyphosate	tkm	-	3697	-	3697
Pesticide					
Cypermethrin	tkm	-	763	-	763
Fipronil	tkm	-	107	11	118
Dimethoate	tkm	-	2815	-	2815
Fertiliser					
Super potash	tkm	9884	13179	21965	45028
Manganese sulphate	tkm	-	-	1070	4586
			Seed transportation		
Perennial grasses	tkm	-	622	-	622
			Fuel consumption & transportation		
Tractor	l	519	1086	2973	4771
Fuel transportation	tkm	65	136	371	572
			Farm machinery production		
Tractor	USD	605	1380	2111	4096
Seeder	USD	183	696	487	1366
Boomspray	USD	-	1661	387	2048
Trimmer	USD	-	-	819	819
Spreader	USD	351	356	1341	2048

tkm = tonne-kilometres

^ Inputs are presented on an annual basis. Inputs applied at establishment x years ago are also considered in this table by adjusted the input value for the number of years since establishment.

[a] Subtropical grasses had been progressively sown over multiple years. This value represents the weighted average years since establishment.

[b] Tagasaste was seeded at a rate of 0.5 kg/km. With a 6 m inter-row and 2 m double row of tagasaste, this equated to the tractor travelling 1.25 km/ha at seeding.

In the summer following establishment, insecticide to target grasshoppers was applied. Once established, tagasaste was trimmed every three years using farm specialist machinery. A total of 100 kg/ha of super potash was applied annually and manganese sulphate (40 kg/ha) applied every five years.

4.5.3.1 Pasture attributes

The pasture attributes for each pasture type at Lancelin were obtained using the same approaches as for Dongara.

Annual pasture data sources

A grass-dominant annual pasture was modelled in GrassGro, containing annual ryegrass, capeweed and sub clover.

Subtropical grass pasture data sources

The perennial pasture, containing the mix of subtropical grasses and annual species, was calculated using regional growth rates and quality attributes obtained from the same sources as the subtropical grass pasture at Dongara.

Tagasaste pasture data sources

The tagasaste pasture attributes were calculated from both the tagasaste and annual inter-row components, with annual inter-row data sourced from GrassGro output and tagasaste data obtained from the same sources as Dongara, adjusted for the specifics of the Lancelin planting.

Calculated pasture attributes

The growth of each pasture followed a similar pattern to Dongara. Annual species dominated each pasture type during the growing season, while perennials supplied green feed outside of the growing season, from November to May. The longer growing season and higher annual rainfall received by Lancelin resulted in higher annual NPP to Dongara, with annual pasture supplying an average 7.53 t DM/ha/yr (Table 4.24). Of this annual biomass production, 93% was produced during the growing season, from May to December. Figure 4.7a shows that by December, annual pasture had senesced. By contrast, the calculated annual NPP of tagasaste pasture at Lancelin was 11.33 t kg DM/ha. Of this, almost 20% was produced outside of the growing season. The tagasaste plants contributed over 33% of total tagasaste pasture annual NPP. The calculated annual NPP of the subtropical grass pasture was 9.92 t DM/ha, with 16% of this produced outside of the growing season by the subtropical grasses.

Figure 4.7a displays the calculated monthly DMA, DMD and CP content values for the three pasture types, whilst averages are presented in Table 4.24. As expected,

the DMA of annual pasture was highest during the growing season, increasing from an average of 1.29 t DM/ha at the start to 4.69 t DM/ha at the end. In fact, Lancelin had the highest average DMA during the peak growing season of all farms examined, at 3.52 t DM/ha. Despite this, the low proportion of legumes in the pasture sward meant that while the average DMD during the peak months of the growing season, 71%, was comparable to the other modelled regions, the CP content was the lowest at 22%. Annual pasture also declined in quality more rapidly than the other regions, with DMD and CP content falling to 45% and 8%, respectively, by the end of March.

Table 4.24 - Productivity and feed quality attributes of the three pasture types grazed at the Lancelin beef enterprise

Input^	Unit	Annual pasture		Perennial grass pasture		Tagasaste pasture	
		Mean	**SD**	**Mean**	**SD**	**Mean**	**SD**
Length of growing season		End-May - Mid-Nov		Jan - Dec		Jan - Dec	
Annual NPP[a]	t DM/ha	7.53	-	9.92	-	11.33	-
Growing season NPP[b]	t DM/ha	7.34	-	8.27	-	9.21	-
Non-growing season NPP[b]	t DM/ha	0.19	-	1.66	-	2.11	-
Available DMA (% green)[c]							
Start of growing season	t DM/ha	1.29 (45%)	*0.00*	2.52	*0.00*	2.50	*0.00*
Peak growing season	t DM/ha	3.52 (77%)	*1.26*	5.74	*1.47*	4.61	*0.95*
End of growing season	t DM/ha	4.69 (6%)	*0.00*	5.12	*0.00*	5.85	*0.00*
Non-growing season	t DM/ha	2.18 (2%)	*0.89*	2.30	*0.87*	3.30	*0.98*
DMD[d]							
Start of growing season	%	79	*0.00*	71	*0.00*	72	*0.00*
Peak growing season	%	71	*4.00*	66	*1.19*	71	*0.82*
End of growing season	%	66	*0.00*	62	*0.00*	65	*0.00*
Non-growing season	%	50	*4.00*	63	*0.94*	67	*1.81*
CP content[d]							
Start of growing season	%	28	*0.00*	22	*0.00*	24	*0.00*
Peak growing season	%	22	*2.59*	20	*1.73*	23	*1.22*
End of growing season	%	19	*0.00*	13	*0.00*	17	*0.00*
Non-growing season	%	10	*1.77*	13	*1.34*	17	*0.90*

^ With the exception of annual net primary productivity (NPP), all averages and standard deviations were calculated only from months that the pasture was grazed within that period.

[a] GrassGro modelled output for the annual pasture and tagasaste pasture annual inter-row. Subtropical grass and tagasaste values calculated using regionally specific information from DPIRD (R. Verbrugge, 2016, pers comm) adjusted for planting density and stock intake specifics.

[b] Where growing season refers to the annual species growing season at the enterprise location.

[c] GrassGro provided values for both total available DMA and green DMA, whereas the regional perennial data only provided overall values.

[d] DMD and CP values for the annual pasture and inter-row component of tagasaste content obtained from GrassGro. Subtropical perennial grass attributes sourced from Moore et al. (2009b), while tagasaste attributes obtained from Wiley (2005a).

Per Dongara, the high proportion of annuals within the tagasaste and perennial grass pastures meant that the average DMA of both pastures mirrored the overall trend of annual pasture. Both the tagasaste and subtropical grass pastures provided higher pasture DMAs than the senesced annual pasture from November to May. Again, this is indicative of the green feed provided by the perennials during these months. Compared to Dongara, perennial grass pasture had a higher average DMA of 5.74 t DM/ha during the peak growing season, which was attributed to a lower grazing intensity at Lancelin during this period. By contrast, pasture at Lancelin was grazed at a higher intensity during the dry season, resulting in an average DMA of 2.30 t DM/ha, lower than Dongara. Tagasaste at Lancelin yielded higher DMAs throughout the year; resulting from the increased biomass production by the annual species and higher planting density of tagasaste, despite higher stocking intensities at Lancelin.

The DMD of both perennial pastures remained relatively constant through the production year. Both were boosted by green biomass produced by the annual species in the growing season, averaging 71% and 66% for tagasaste and subtropical grass pastures, respectively. Green biomass provided by perennial components during the non-growing season offered quality feed with tagasaste and perennial grass pastures averaging 67% and 63% during this period. The CP content fluctuated more; tagasaste averaged 23% during the growing season and 17% in the non-growing season, while perennial grasses averaged 20% and 13%, respectively. By contrast, during the same period the average DMD and CP content of the annual pasture was 50% and 10%, respectively.

4.5.4 Grazing management

Lancelin employed a rotational grazing strategy between the three pasture types throughout the production year, with the farmer working to match feed demand with supply by altering grazing intensity and duration.

<u>Annual pasture</u>

Annual pasture was grazed all year, with grazing intensity and duration the greatest in the opening months of the growing season, from June to August. During this period pasture quality was at its highest and coincided with the calving of the breeding herd. As such pregnant and lactating stock, in particular first-time heifers, were grazed on annual pasture during these months. Through the dry season, when annual pasture provided below maintenance energy to grazing livestock, the pasture was grazed for shorter durations. This allowed the perennial pasture to be frequently rested, with the livestock gains on perennials offsetting the losses on senesced annual pasture.

Tagasaste pasture

The tagasaste pasture was a primary grazing source throughout the production year, particularly from September to the end of December when the overall energy requirements of the herd were high. During these months, the quantity and quality of annual pasture steadily declined (Figure 4.7a) and could no longer meet these energy requirements. By contrast, tagasaste provided ample green feed of higher quality than annual pasture from October to June. Tagasaste was also important at the end of the dry season, where supplementary feed would otherwise be essential.

Table 4.25 - Feed quality attributes and annual inputs of the supplementary feed and yards ration supplied at the Lancelin beef enterprise

Attribute or input^	Unit	Annual pasture	Perennial grass pasture	Tagasaste pasture	Total
		Supplementary feed attributes[ab]			
Months provided			Jan-May		
DMD					
Pellets	%		80		
Hay	%		59		
CP content					
Pellets	%		21		
Hay	%		8		
Mineral lick	%		30		
		Supplementary feed input[c]			
Pellets	kg	45395	29844	20593	95832
Hay	kg	32043	21067	14536	67646
Mineral lick	kg	1100	1906	993	3999
		Supplementary feed transportation			
Pellets	tkm	6446	4238	2924	13608
Hay	tkm	4326	2844	1962	9132
Mineral lick	tkm	235	408	213	856
		Yards ration attributes[b]			
DMD					
Hay	%		59		
CP content					
Hay	%		8		
		Yards ration input[c]			
Hay	kg				9771
		Yards ration transportation			
Hay	tkm				1319

tkm = tonne-kilometres

^ Inputs are presented on an annual basis.

[a] Provided to agistment heifers and calves only.

[b] DMD and CP content of pellets and hay were sourced from product specifications and DAFWA (2006a).

[c] Presented on an "as-fed" DM basis with an assumed wastage of 20%.

Subtropical grass pasture

Subtropical grass pasture was the primary feed source for agisted cattle. The pasture was grazed by stock through the production year though was rested for longer periods during the cooler months when subtropical grasses were largely dormant. The arrival

of the agistment stock coincided with the peak growth period of the subtropical grasses which, as a result, were able to support cattle requirements.

The rotational grazing of the breeding herd at Lancelin removed the requirement for supplementary feed. Whilst senesced annual pasture was grazed, the energy deficiencies encountered by cattle grazing this pasture were offset by gains obtained on perennial pastures. The high energy requirements of lactating agistment heifers and growing calves meant that supplementary feed was provided to the agistment herd from January through to the end of the non-growing season. During these months, agistment cattle were provided almost 96 tonnes of pellets, 68 tonnes of hay, and four tonnes of mineral licks (Table 4.25). The pellets were high quality; 80% DMD and 21% CP content, while the hay provided roughage. As the lactating stock had higher protein requirements, mineral licks with CP contents of 30% were provided.

In January, farm-bred and purchased weaner bull calves remained in the yards for one week prior to sale. While in the yards, they were supplied approximately 10 t hay. All other stock were sold directly off pasture.

4.5.5 Calculated on-farm inventory outputs

The calculated on-farm emission outputs for Lancelin, the final enterprise considered in this study, are presented in Table 4.26.

<u>Enteric methane emissions</u>

Total enteric methane emissions at Lancelin were 60,216.93 kg CH_4/year, less than half produced by the other beef enterprise, Dongara. Despite this, the EI of enteric methane was 63% higher than Dongara, totalling 4.01×10^{-1} kg CH_4/kg LW produced for sale. This lower efficiency can be attributed to the higher proportion of breeding stock to agistment cattle at Lancelin, which meant that a greater proportion of total enteric methane was produced by the breeding herd as opposed to livestock destined for sale. Of all the enterprises considered in this study, Lancelin had the highest enteric methane EI.

Of the three pasture types, subtropical grasses produced the lowest enteric methane emissions in terms of saleable liveweight (2.98×10^{-1} kg CH_4/kg LW), followed by annual pasture (4.20×10^{-1} kg CH_4/kg LW) and then tagasaste (4.50×10^{-1} kg CH_4/kg LW). This contrasts to Dongara, where tagasaste was the most emissions efficient and annual pasture the least. These results did not, however, mean that tagasaste was the least productive of the pastures at Lancelin. In fact, tagasaste was the dominant grazing source of the breeding herd through the production year, particularly

through the second half of the calving cycle and the non-growing season. By contrast, annual pasture supported only a small proportion of the breeding stock and was grazed through the non-growing season when intake, and thus enteric methane production, was lower. This is reflected in the enteric emissions produced on a liveweight grazed basis, where annual pasture produced the lowest emissions output (1.06×10^{-2} kg CH_4/kg LW grazed). The high productivity of subtropical grasses was likely due to it being predominantly grazed by the agistment herd. In fact, subtropical grasses supported the production of 27% of total annual saleable liveweight as compared to 21% on annual pasture, despite supporting less livestock through the production year (Table 4.27). Section 4.5.5.1 explores the factors influencing enteric methane production at Lancelin further.

Manure methane emissions

Manure methane contributed a small fraction to overall on-farm methane emissions, totalling 13.96 kg CH_4/year, or 9.30×10^{-5} kg CH_4/kg LW produced for sale. This was higher than the three other enterprises. For the same reasons as outlined in the previous paragraph, the manure methane EI on tagasaste pasture was higher than subtropical grasses. By contrast, annual pasture was the most efficient (1.05×10^{-4} kg CH_4/kg LW), reflective of the lower average DMD across the production year.

Nitrous oxide emissions from excreta

Excreta nitrous oxide emissions totalled 511.31 kg N_2O/yr, or 3.41×10^{-3} kg N_2O/kg LW produced for sale. This was the highest of all the considered enterprises and is reflective of the high proportion of growing, lactating and suckling stock at Lancelin. This is evident when tagasaste pasture at each cattle enterprise were compared. In terms of saleable liveweight production, the output at Lancelin was 4.13×10^{-3} kg N_2O/kg LW, 60% higher than Dongara. By contrast, in terms of liveweight grazed, the output was 1.07 kg N_2O/kg LW grazed, 22% less than Dongara. As discussed earlier, tagasaste pasture at Lancelin was the primary grazing source of the breeding herd and was grazed the most intensively throughout the production year. This meant that, in terms of liveweight grazed, the tagasaste pasture was more efficient at Lancelin, but that only a small proportion of that liveweight was sold. A comparison of the three pasture types at Lancelin reinforces this finding. The tagasaste was the least emissions efficient, reflecting that the other two pasture types at Lancelin were more focussed on the production of saleable liveweight than tagasaste.

Table 4.26 - Calculated annual animal, soil and associated on-farm emissions produced on the Lancelin beef enterprise

Output	Unit	Annual pasture	Perennial grass pasture	Tagasaste pasture	Yards	Total
				Animal emissions		
Enteric CH_4	kg CH_4/yr	11069.40	12271.10	36719.11	157.32	60216.93
	kg CH_4/LW grazed[a]	1.06E-02	1.42E-02	1.16E-02	1.49E-02	1.19E-02
	kg CH_4/kg LW produced for sale[b]	4.20E-01	2.98E-01	4.50E-01	1.43E-01	4.01E-01
Manure CH_4	kg CH_4/yr	2.76	3.13	8.03	0.04	13.96
	kg CH_4/LW grazed	2.64E-06	3.62E-06	2.54E-06	4.13E-06	2.75E-06
	kg CH_4/kg LW produced for sale	1.05E-04	7.60E-05	9.84E-05	3.95E-05	9.30E-05
				Direct soil emissions		
N_2O excreta	kg N_2O/yr	95.68	78.15	337.08	0.40	511.31
	kg N_2O/LW grazed	9.15E-05	9.05E-05	1.07E-04	3.87E-05	1.01E-04
	kg N_2O/kg LW produced for sale	3.63E-03	1.90E-03	4.13E-03	3.71E-04	3.41E-03
N fertilisers N_2O	kg N_2O/yr	-	-	-	-	-
	kg N_2O/ha	-	-	-	-	-
	kg N_2O/kg LW produced for sale	-	-	-	-	-
Crop residue N_2O	kg N_2O/yr	-	-	-	-	-
	kg N_2O/ha	-	-	-	-	-
	kg N_2O/kg LW produced for sale	-	-	-	-	-
Pasture residue N_2O	kg N_2O/yr	16.31	33.92	27.22	-	77.44
	kg N_2O/ha	3.62E-02	8.48E-02	2.72E-02	-	4.19E-02
	kg N_2O/kg LW produced for sale	6.18E-04	1.74E-03	3.34E-04	-	5.16E-04
				Indirect soil emissions		
Atmospheric deposition N_2O	kg N_2O/yr	19.14	15.63	67.42	0.08	102.26
	kg N_2O/kg LW produced for sale	7.25E-04	3.80E-04	8.27E-04	7.42E-05	6.81E-04
				Other calculated on-farm emissions		
Liming CO_2	kg CO_2/yr	-	-	-	-	-
	kg CO_2/ha	-	-	-	-	-
	kg CO_2/kg LW produced for sale	-	-	-	-	-
Urea hydrolysis CO_2	kg CO_2/yr	-	-	-	-	-
	kg CO_2/ha	-	-	-	-	-
	kg CO_2/kg LW produced for sale	-	-	-	-	-

LW= liveweight

[a] To enable comparison across feedbases, emissions were adjusted according to output per kg of liveweight grazed on each.

[b] Using the functional unit, overall emissions were adjusted according to output per kg of liveweight produced for sale.

Other soil emissions

Total nitrous oxide emissions from pasture residue were 77.44 kg N_2O/yr. On a per hectare basis, pasture residue emissions followed the same trend across the pastures as for Dongara, with perennial pastures producing less than annual pasture. Compared to Dongara, emissions from both annual pasture (3.62×10^{-2} kg N_2O/ha) and tagasaste (2.72×10^{-2} kg N_2O/ha) were higher, a result of the higher annual NPP of annual pasture and lower grazing intensity at Lancelin. Despite the results on a per hectare basis, in terms saleable liveweight production, annual pasture, subtropical grass pasture and tagasaste pasture at Dongara produced 59%, 88% and 68% less nitrous oxide, respectively than Lancelin. This reflected the differences in the pasture management of each farm, annual NPPs and liveweight sold, with Dongara selling 66% more liveweight than Lancelin.

Emissions from atmospheric deposition totalled 102.23 kg N_2O/year, or 3% of whole-farm nitrous oxide outputs. No leaching or runoff occurred. As no N fertiliser was applied or liming program employed, no emissions were produced from these sources.

4.5.5.1 Analysis of the interactions between feedbase, animal intake and enteric methane production at the Lancelin beef enterprise

Of the three pasture types at Lancelin, subtropical grasses had the lowest enteric methane EI, followed by annual pasture and tagasaste. This Section further examines the factors which influenced these emissions.

Each pasture at Lancelin was grazed year-round. Unlike Dongara, not all stock classes grazed all three pasture types and of those which did, few grazed each through the production year. This is evident in Figure 4.7 which, though doesn't provide information on the number of livestock grazing each pasture, gives an indicator of grazing intensity through the presentation of various stock classes grazed. At the beginning of the growing season, when pasture quality was high, annual pasture supported mature cows and heifers (on-farm and purchased) through the calving period. As pasture quality and availability declined towards the end of the growing season, lactating stock and calves were removed from annual pasture, rotating instead between the two perennial pastures. During the dry season intake across all stock classes grazing annual pasture was noticeably restricted. The perennial pastures were critical during this period by providing feed surplus to requirements, which offset the deficit encountered on annual pasture.

The feed intake across all stock classes on perennial and tagasaste pasture remained relatively constant over the production year, with variations attributed to changes in

the physiological status of stock (i.e. lactation, growth) rather than large changes in pasture attributes. Though the farm employed a rotational grazing strategy, tagasaste pasture supported the greatest share of the breeding herd during the production year, while subtropical grasses were the primary grazing source for the agistment herd. Though supplementary feed was required to support lactating agistment heifers on perennials, this was not required until the end of the non-growing season (26 to 32% of total intake, Table 4.27) when the daily requirements were greatest. By contrast, on annual pasture in the opening months of the non-growing season, when heifer daily requirements were lower, supplementary feed comprised 39 to 45% of total intake. This highlights the important role of the perennials in supporting the production of cattle at Lancelin during periods where annual pasture was insufficient.

Of all the stock classes at Lancelin, mature bulls had the highest average daily feed intake and enteric methane production, totalling 12.47 kg DM/head/day and 0.258 kg CH_4/head/day, respectively (Table 4.27). This was followed by agistment heifers (10.61 kg DM/head/day, 0.220 kg CH_4/head/day) and lactating mature cows from the breeding herd (9.22 kg DM/head/day, 0.191 kg CH_4/head/day). The higher calculated outputs from the agistment heifers than the heavier, mature cows, is likely due to a combination of continued liveweight gain throughout the agistment period, lactation for five of the seven months that agistment heifers were on-farm and also the provision of supplementary feed.

As for Dongara, intake and enteric methane production for each stock class was highest on subtropical grasses, followed by tagasaste and finally annual pasture. Whilst intake by a stock class on the perennials did not vary considerably throughout the production year, fluctuations on annual pasture were substantial (Figure 4.7). However, interpretation of the calculated annual pasture averages must also consider management decisions. For example, dry cows from the breeding herd grazed all three pastures most months and consumed on average 30 to 33% less each day on annual pasture as opposed to perennials (26 to 30% less enteric methane). By contrast, mature bulls from the breeding herd only grazed annual pasture during the growing season and so differences between annual and perennial pastures were less pronounced (1% to 6% less pasture intake and enteric methane production). This demonstrates the importance of grazing management from productivity and emissions perspective.

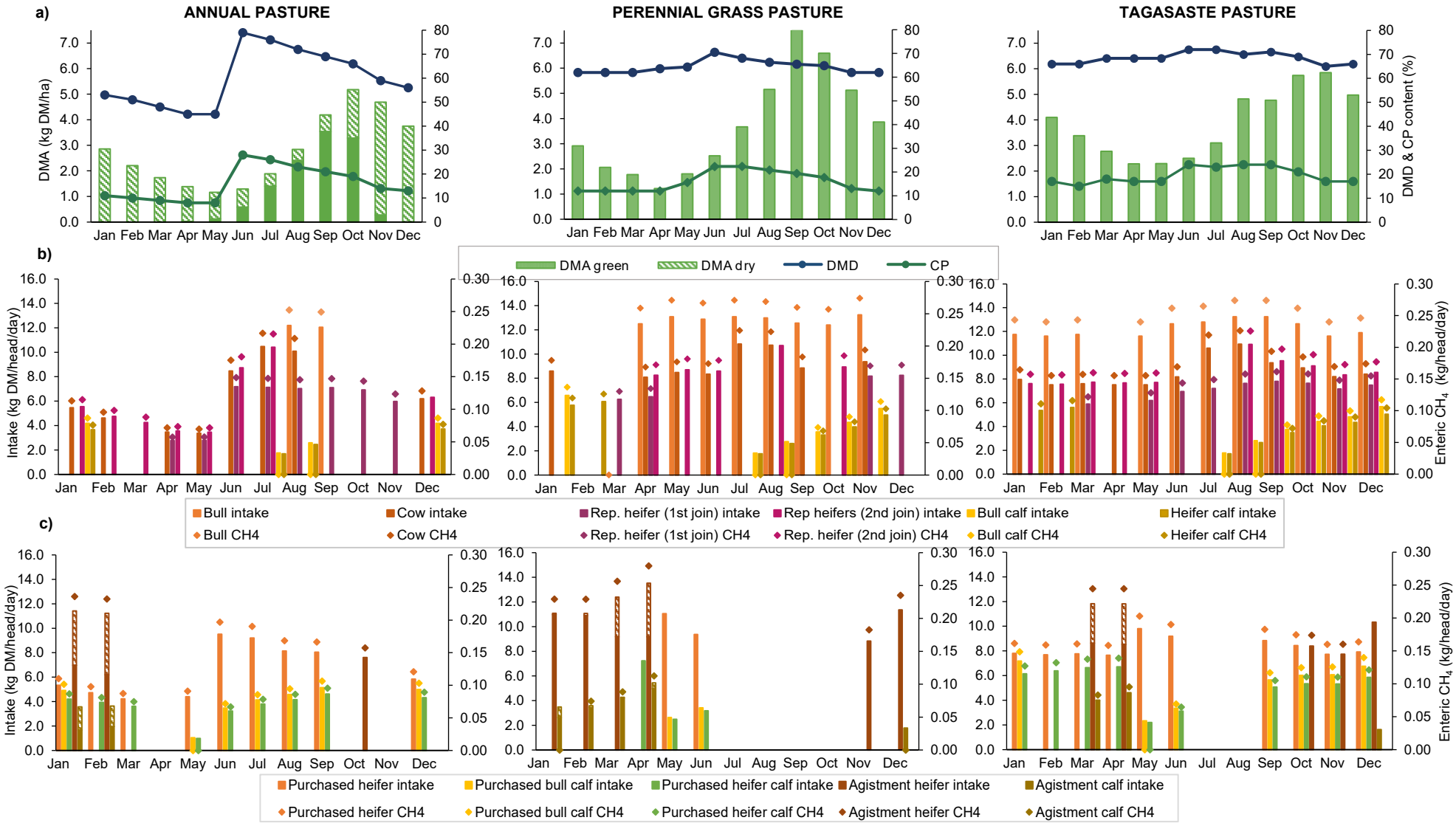

a)
ANNUAL PASTURE
PERENNIAL GRASS PASTURE
TAGASASTE PASTURE
DMA (kg DM/ha)
DMD & CP content (%)
Jan Feb Mar Apr May Jun Jul Aug Sep Oct Nov Dec
DMA green
DMA dry
DMD
CP
b)
Intake (kg DM/head/day)
Enteric CH4 (kg/head/day)
Bull intake
Cow intake
Rep. heifer (1st join) intake
Rep heifers (2nd join) intake
Bull calf intake
Heifer calf intake
Bull CH4
Cow CH4
Rep. heifer (1st join) CH4
Rep. heifer (2nd join) CH4
Bull calf CH4
Heifer calf CH4
c)
Purchased heifer intake
Purchased bull calf intake
Purchased heifer calf intake
Agistment heifer intake
Agistment calf intake
Purchased heifer CH4
Purchased bull calf CH4
Purchased heifer calf CH4
Agistment heifer CH4
Agistment calf CH4

Figure 4.7 (previous page) - Relationship between monthly feedbase attributes, feed intake and enteric methane production of cattle at the Lancelin beef enterprise

Where; a) the feedbase attributes; dry matter availability (DMA, kg DM/ha/yr), dry matter digestibility (DMD, %) and crude protein content (CP, %); b) & c) the average daily feed intake (kg DM/head; solid colour is feedbase intake, patterned colour is supplementary feed intake) and average daily enteric methane production (kg/head, scatter plot) of each stock class within the breeding and agistment herds, respectively Note: Only annual pasture data modelled in GrassGro could distinguish between green and dry DMA, perennial pasture monthly values reflect total DMA only.

Influence of management practices on feed intake and enteric methane production

Using the results presented in this Section, it was possible to compare the impacts, in terms of enteric methane, between saleable liveweight produced by the breeding herd and by the agistment herd. Total emissions associated with the production of weaner bull calves by mature cows in the breeding herd was 49.01 kg CH_4 per calf or 0.213 kg CH_4/kg LW produced for sale. These values only considered the bull calf to sale and the mature cow during the period of lactation. If emissions during pregnancy were included, these increase to 76.46 kg CH_4 or 0.332 kg CH_4/kg LW. If first-calf heifers were instead considered, the total enteric emissions were 49.49 kg CH_4 or 0.215 kg CH_4/kg LW. With the inclusion of pregnancy emissions, the revised value was 77.80 kg CH_4 or 0.338 kg CH_4/kg LW. By contrast, emissions associated with the production of agistment cattle totalled 51.81 kg CH_4. As these emissions include both the agistment of heifers and calves, the liveweight gained by both stock classes is considered, resulting in 0.259 kg CH_4/kg LW. These results align the findings of Dongara, that the production of agistment cattle is more emissions efficient than stock produced by the breeding herd. It should be noted that these values only consider the final two months of pregnancy for agistment heifers as emissions prior to the farm gate fell outside the study system.

Compared to Dongara, the impact of producing bull calves using the breeding herd was lower at Lancelin. Enteric methane production by bull calves produced by mature cows and first-calf heifers, from pregnancy through to sale, was 10% and 9% lower in terms of liveweight sold, respectively. By contrast, all pastoral stock classes at Dongara produced less enteric methane per kilogram of saleable liveweight compared to agistment stock at Lancelin, ranging from 3% (pastoral steers and heifers) to 47% (pastoral bull calves). Considering 88% of total saleable liveweight produced at Dongara was from pastoral cattle, while 64% was from the breeding herd at Lancelin, these differences are likely to play a role in the overall carbon footprint differences between the two farms. This is explored further in Chapter Five.

Table 4.27 - Daily feed intake and calculated enteric methane production across stock classes for the annual and perennial pastures at the Lancelin beef enterprise

Input	Unit	Annual pasture		Perennial grass pasture		Tagasaste pasture		Total	
		Mean	SD	Mean	SD	Mean	SD	Mean	SD
Months grazed[a]		Jan-Dec		Jan-Dec		Jan-Dec			
Proportion of LW produced for sale[b]	%	18		27		54			
				Daily intake					
Breeder herd									
Bulls	kg DM/hd	12.12	*0.07*	12.84	*0.30*	12.25	*0.64*	12.47	*0.58*
Cows (dry)	kg DM/hd	5.56	*1.95*	8.33	*0.17*	7.89	*0.41*	7.44	*1.48*
Cows (lact.)	kg DM/hd	8.22	*2.25*	9.86	*0.92*	9.35	*1.09*	9.22	*1.56*
Heifers (rep. 1st join)	kg DM/hd	5.86	*1.82*	7.29	*0.92*	7.11	*0.62*	6.67	*1.41*
Heifers (rep. 2nd join)	kg DM/hd	5.88	*2.35*	9.03	*0.86*	8.28	*0.94*	7.71	*1.93*
		(6.60)	*(2.00)*	(10.21)	*(0.85)*	(8.28)	*(0.72)*	(7.75)	*(1.82)*
Bull calves	kg DM/hd	3.18	*1.05*	4.09	*1.61*	3.88	*1.29*	3.78	*1.41*
		(3.14)	*(2.05)*	(3.02)	*(0.39)*	(5.34)	*(1.67)*	(4.48)	*(1.64)*
Heifer calves	kg DM/hd	2.88	*0.85*	4.06	*1.50*	4.03	*1.27*	3.80	*1.38*
		(3.65)	*(1.02)*	(4.30)	*(2.09)*	(5.28)	*(1.43)*	(4.48)	*(1.59)*
Agistment herd[c]									
Heifers	kg DM/hd	10.08	*1.75*	11.38	*1.43*	10.02	*1.70*	10.61	*1.73*
Calves	kg DM/hd	3.60	*0.04*	3.72	*1.19*	3.41	*1.30*	3.61	*1.11*
				Daily enteric methane production					
Breeder herd									
Bulls	kg CH_4/hd	0.251	*0.002*	0.266	*0.006*	0.254	*0.013*	0.258	*0.012*
Cows (dry)	kg CH_4/hd	0.121	*0.040*	0.172	*0.003*	0.163	*0.008*	0.154	*0.031*
Cows (lact.)	kg CH_4/hd	0.170	*0.046*	0.204	*0.019*	0.193	*0.022*	0.191	*0.32*
Heifers (rep. 1st join)	kg CH_4/hd	0.121	*0.038*	0.151	*0.019*	0.147	*0.013*	0.138	*0.029*
Heifers (rep. 2nd join)	kg CH_4/hd	0.122	*0.049*	0.187	*0.094*	0.171	*0.019*	0.160	*0.040*
		(0.137)	*(0.041)*	(0.211)	*(0.018)*	(0.171)	*(0.015)*	(0.160)	*(0.038)*
Bull calves	kg CH_4/hd	0.087	*0.000*	0.103	*0.020*	0.097	*0.014*	0.097	*0.019*
		(0.094)	*(0.012)*	(0.071)	*(0.000)*	(0.121)	*(0.025)*	(0.105)	*(0.025)*
Heifer calves	kg CH_4/hd	0.076	*0.000*	0.100	*0.022*	0.096	*0.016*	0.095	*0.019*
		(0.082)	*(0.008)*	(0.108)	*(0.042)*	(0.102)	*(0.022)*	(0.101)	*(0.026)*
Agistment herd									
Heifers	kg CH_4/hd	0.209	*0.036*	0.235	*0.030*	0.207	*0.035*	0.220	*0.036*
Calves	kg CH_4/hd	0.075	*0.000*	0.092	*0.016*	0.089	*0.006*	0.088	*0.013*
				Yards					
Daily intake									
Bull calves	kg DM/hd							5.54	-
								(6.38)	-
Daily enteric CH_4 production									
Bull calves	kg CH_4/hd							0.115	-
								(0.131)	-
				Supplementary feed[d]					
Months fed				Jan-Apr					
Proportion of total intake									
January	%	39		1		-		26	
February	%	45		2		-		21	
March	%	-		26		27		26	
April	%	-		32		28		30	

Hd = head (livestock unit)
[a] Rotationally grazed through the production year.
[b] Proportion of total liveweight sold produced on each feedbase.
[c] Calculated average of combined pasture and supplementary feed intake.
[d] Applicable to the agistment herd only.

4.6 Summary of inventory results

The previous Sections presented the inventories and preliminary analyses of the four

livestock production enterprises. The scope of the inventory information collected for the carbon footprint analyses was considerable. A summary of the key characteristics of Bremer Bay, Wickepin, Dongara and Lancelin is presented in Table 4.28 as a quick reference for following Chapters.

Each farm was selected based on their adoption of perennials within their enterprise. The inventories have highlighted that each enterprise integrated perennials into their system differently. For example, more than half the arable land dedicated to pasture at Bremer Bay was allocated to kikuyu and was continually grazed through the year and supported the second lambing cycle. The analyses of feedbase attributes, animal requirements and intake demonstrated that this would not have been possible on annual pasture without significant quantities of supplementary feed. The motivation behind the introduction of perennials at Wickepin was quite different, with saltbush initially planted to remediate salt-affected farmland. However, the saltbush could also support mature ewes during the dry season, providing a source of green feed and enabling the preservation of best quality stubble for growing stock. Dongara and Lancelin introduced tagasaste and subtropical grasses to improve the productivity of their poor soils, increasing the carrying capacity of their enterprises. Each managed the perennials differently; Dongara rotationally grazed cattle using a wagon wheel approach, enabling the backgrounding of large numbers of pastoral cattle. Lancelin also rotationally grazed cattle but managed the herds so that the breeding herd predominantly grazed annual pasture and tagasaste in alignment with breeding events, while the agistment herd predominantly grazed subtropical grasses.

These initial emissions calculations highlight the variation that exists between feedbases, between stock classes and through the production year. For example, annual pasture DMD, CP content and DMA varied by up to 40%, 70% and 43%, respectively across the four farms at any particular month, largely a result of different growing season lengths across the farms and pasture species. This variation would not be captured if regional averages or seasonal values had been adopted instead of GrassGro output and farm-specific information. Similarly, large variations are exhibited between feedbases on each farm (Figures 4.2, 4.3, 4.6, 4.7), with subsequent flow-on effects to calculated animal emissions. For example, at Dongara, enteric methane production by dry cows varied by up to 23% depending on the pasture grazed. By contrast, at Wickepin this difference by dry ewes was only 7%, not a reflection of the similarities between feedbases at Wickepin, but rather the high rates of supplementary feed required on certain feedbases.

Table 4.28 - Summary of the key characteristics of the four livestock production enterprises

Farm characteristic	Bremer Bay	Wickepin	Dongara	Lancelin
Region	Great Southern	Wheatbelt	NAR	NAR
Rainfall	514 mm	360 mm	442 mm	599 mm
Enterprise type	Prime lamb	Prime lamb, wool, crop	Beef cattle (breeding & backgrounding)	Beef cattle (breeding & agistment)
Enterprise area	610 ha	6,000 ha	3,600 ha	2,000 ha
Breed	Dorper	Merino; SAMM	Red Angus (breeding herd); Brahman, Droughtmaster, etc (pastoral herd)	Red Angus, Gelbvieh, etc (breeding herd); Brahman (agistment herd)
Annual growing season	May to Nov	Mid-May to Mid-Nov	Jun to Oct	Late May to late Nov
Pasture, crops & feedlot				
Pasture type (area)	Annual (170 ha); subtropical grasses (240 ha)	Annual (2,320 ha) Saltbush (110 ha)	Annual (950 ha); subtropical grasses (1,600 ha); tagasaste (450 ha)	Annual (450 ha); subtropical grasses (400 ha); tagasaste (1,000 ha)
Months grazed	Annual (Apr-Dec); Subtropical grasses (Jan-Feb)	Annual (Apr-Dec) Saltbush (Nov-Apr)	All pasture types (Jan to Dec)	All pasture types (Jan to Dec)
Months supplementary fed	Annual (Apr; Dec) Subtropical grasses (Jan-Apr)	Annual (Dec-May) Saltbush (Dec-Apr)		All pasture types (Jan-Apr; agistment cattle only)
Income crop (area)		Oats, Barley, Wheat, Canola (2,350 ha)		
Supplementary feed crop (ha)	Lupins (40 ha) Oats (90 ha)	Lupins (300 ha)		
Months grazed)	Lupins (Jan-Mar) Oats (Jan-Mar)	Lupins (Jan-Mar) Income crop (Jan-Apr)		
Months supplementary fed	Lupins (Feb-Mar) Oats (Feb-Mar)	Lupins (Jan-Mar) Income crop (Jan-Apr)		
Feedlot Y/N	Y - weaner lamb (finishing ration)	N	Y – weaner cattle (finishing ration); other sold stock (maintenance ration)	N
Breeding flock/herd				
Breeding flock/herd count(ewes/cow)	800	3,100	200	355
Month of lambing/calving	Mar; Sep	Apr (SAMM) Jun (Merino)	Jun	Jul (breeding herd) Dec (agistment herd)
Weaning percentage	150% (75% per lambing)	92.5% (SAMM) 85% (Merino)	80%	92% (mature cow) 72% (on-farm heifer)
Month of weaning (lamb/calf)	Jan; Jul	Oct (SAMM) Dec (Merino)	Dec (bull calves) Jan (heifer calves)	Jan (bull calves) Feb (heifer calves) Jul (bought calves)
Month of sale (lamb/calf)	Mar; Sep	Oct (SAMM) Feb (Merino)	Dec (bull calves) Aug (heifers)	Feb (bull calves)
Sale weight (lamb/calf)	45 kg	45 kg (SAMM) 37 kg (Merino)	215 kg (bull calves) 350kg (heifers)	230 kg, 273 kg (bull calves)
Sale count (lamb/calf)	1,050	800 (SAMM) 800 (Merino)	80 (bull calves) 50 (heifers)	192 (bull calves)
Brought-in flock/herd				
Brought in flock/ herd count			1,200 (heifers) 900 (steers) 900 (bull calves)	265 (heifers) 265 (calves)
Month onto farm			Jun (heifers) Jul (steers) Aug (bull calves)	October (heifers) December (calves)
Month off-farm			Mar, Apr (heifers) Nov, Feb (steers) Dec (bull calves)	
Sale weight			350 & 365 kg (heifers) 343 & 400 kg (steers) 225 kg (bull calves)	
Total saleable LW produced on-farm	57,975 kg	108,000 kg	53,575 kg (breeding) 388,500 kg (pastoral)	97,110 kg (breeding) 52,999 kg (agistment)

Given the influence of liveweight, growth rate and physiological status on animal emissions, the ability of the Frameworks to consider more stock classes than standard NIR recommendations, such as differentiated growing and lactating stock classes, enhanced the analyses. The emission variations, both across stock classes and within a stock class across a production year, are evident. The inclusion of such detail enabled examination of emissions efficiency not just across the farms, but also across the feedbases present at each.

The inventory results also enabled a preliminary examination of the influence of farm practices on emissions. For example, at Bremer Bay finishing lambs in the feedlot reduced enteric methane EI, a result of the higher growth rates and shorter turnoff of lambs in the feedlot. This offset the higher daily enteric emissions resulting from increased intake in the feedlot. At Wickepin, monthly calculations enabled a comparison of the enteric methane EI of SAMM and Merino lamb production. With the consideration of lambing month, pasture attributes and breed characteristics, it was determined that the production of SAMM lambs yielded the lower EI. This was because the SAMMs achieved higher growth rates, sale weights and faster turnoff than Merino lambs, with a lower overall requirement for supplementary feed. At Dongara and Lancelin, cattle produced by the breeding herd yielded higher EIs to backgrounded or agistment cattle, a result of the requirement to allocate the emissions burden of the breeding herd to calf production. The influence of practices at a feedbase, rather than whole-farm, scale was also observed. For example, at Lancelin the enteric EI of tagasaste pasture was higher than the other pastures. This was not a reflection of the inefficiency of tagasaste pasture, rather a result of grazing management decisions, with the breeding herd grazing predominantly the tagasaste. Beyond cull animals, breeding herds produce no direct saleable liveweight, which resulted in a higher pasture EI than the other pastures, on which the majority of saleable liveweight was produced. These preliminary examples highlight the depth of analysis possible using the developed Frameworks.

4.7 Conclusion

This Chapter has presented the inventories and preliminary GHG emission analyses of the four examined livestock production systems. The inventories provided detailed information on each enterprise, highlighting the diversity which exists between farm systems, both in terms of inherent farm characteristics and also adopted management practices, across south-western Australia. The capture of such information was made possible by the Frameworks, which considered the effect of intra-annual fluctuations,

stock class, farm location and feedbase on emissions output.

The information presented in this Chapter lays the foundation for the carbon footprint analysis undertaken in Chapter Five, where the preliminary analyses conducted here can also be examined in the context of a whole-farm carbon footprint. The influence of the farm characteristics and management strategies identified in this Chapter on the carbon footprint of the four case study farms, along with the potential implications for livestock production enterprises in south-western Australia, is then presented in Chapter Six.

5 CARBON FOOTPRINT ANALYSIS OF LIVESTOCK PRODUCTION

5.1 Introduction

The previous Chapter introduced the livestock production enterprises considered in this study. Together, they are representative of a range of broadacre systems found through south-western Australia from biophysical attributes, to enterprise type, to implemented management practices. Building on the preliminary findings of Chapter Four, this Chapter presents in detail the carbon footprint analyses of each enterprise. By doing so, it addresses the second research objective of this study. That is to;

Quantify the carbon footprint of livestock production systems in south-western Australia, with focus on perennial versus annual pasture systems.

The opening Section of this Chapter presents and compares the whole-farm carbon footprints and ensuing pre- and on-farm emissions hotspots of the examined livestock production systems. The presentation of these hotspots is accompanied by initial interpretations, enabled by the detailed farm inventory and farm management information detailed in Chapter Four. Following this initial carbon footprint presentation and hotspot identification, the following Sections, divided into sheep production and beef cattle production explore the calculated carbon footprints in greater detail, examining the respective roles of feedbase type, stock class distribution, time of year and management practices on emissions output. Detailed interpretations and comparisons between the systems are made, in recognition of the importance of these findings in answering the second objective of this study and laying the foundations of the analyses conducted in Chapters Six and Seven. The final Section of this Chapter discusses these results in the context of other carbon footprint studies, identifying trends and differences.

5.2 Whole-farm carbon footprints of the livestock production systems

The whole-farm carbon footprints of the livestock production systems, expressed in terms of emission intensities (EIs), were 8.18, 9.17, 10.60 and 13.20 kg CO_2-e/kg LW produced for sale, for the Bremer Bay, Dongara, Wickepin and Lancelin case study farms, respectively (Figure 5.1; Table 5.1). Interestingly, the two lowest EIs were not produced by the same enterprise type, rather, Bremer Bay, a sheep production enterprise and Dongara, a beef cattle production enterprise. Bremer Bay was the

smallest of the examined enterprises, in terms of both saleable liveweight produced and arable land dedicated to livestock production, while Dongara was the largest enterprise.

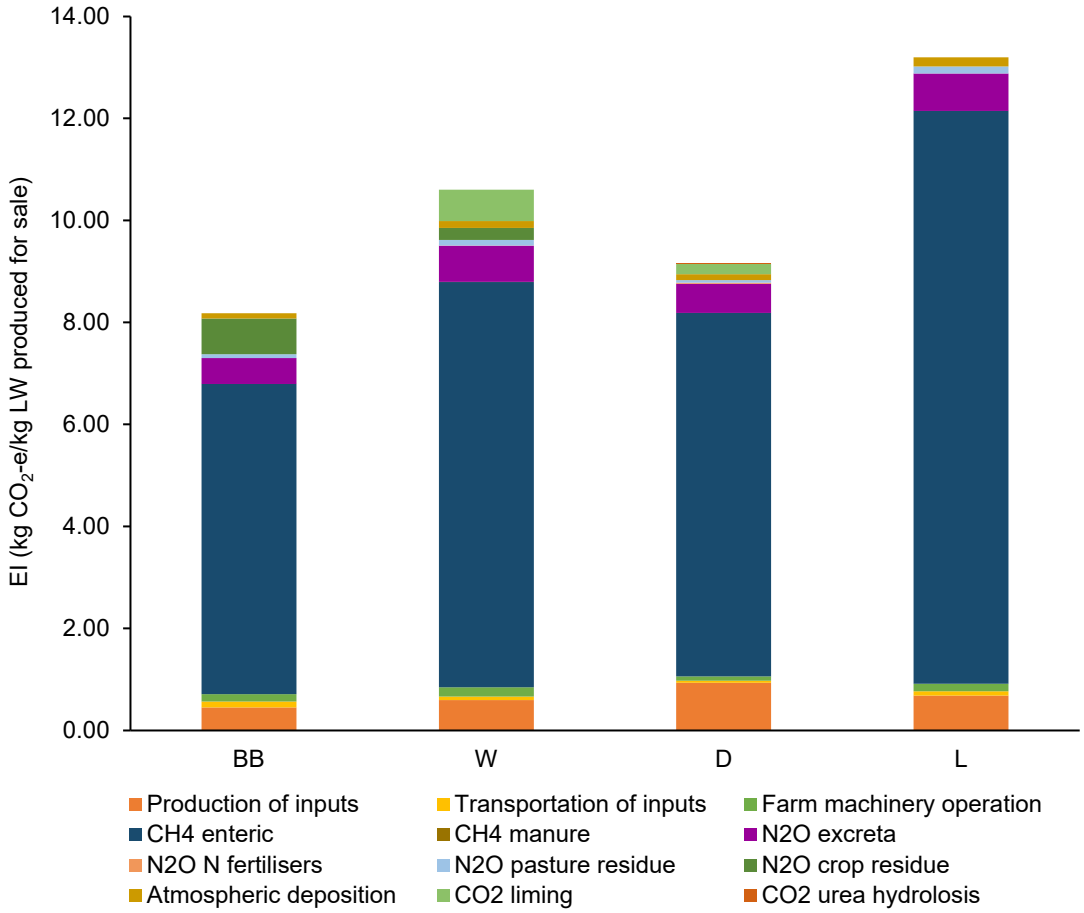


Figure 5.1 - The whole-farm carbon footprints of the Bremer Bay, Wickepin, Dongara and Lancelin livestock production enterprises including a breakdown of the contribution of each emission source. The Wickepin carbon footprint is presented post-allocation of emissions between meat and wool production.

The emissions output of 10.60 kg CO_2-e/kg LW by the Wickepin enterprise represented the EI following the allocation of emissions between meat and wool production. In fact, prior to protein mass allocation, the EI totalled 17.21 kg CO_2-e/kg LW, more than double that of Bremer Bay and almost double that of Dongara. This highlighted that Wickepin was not necessarily efficient from a GHG emissions perspective but benefited from the production of multiple products. There are number of farm characteristics and management practices which provide explanation as to the exhibited differences between the livestock enterprises. These are expanded on in the following Sections.

Table 5.1 - The carbon footprints, presented as GHG emission quantities (EQ) and GHG emissions per unit of liveweight produced for sale (EI) of the four livestock production enterprises

Emission source	Bremer Bay		Wickepin[a]		Dongara		Lancelin	
	EQ (kg CO_2-e/yr)	EI (kg CO_2-e/ kg LW)	EQ (kg CO_2-e/yr)	EI (kg CO_2-e/ kg LW)	EQ (kg CO_2-e/yr)	EI (kg CO_2-e/ kg LW)	EQ (kg CO_2-e/yr)	EI (kg CO_2-e/ kg LW)
Enterprise total	**474,185**	**8.18**	**1,145,073[b]**	**10.60[c]**	**4,051,957**	**9.17**	**1,981,018**	**13.20**
				PRE-FARM				
Pre-farm emissions	**32,740**	**0.56**	**71,510**	**0.66**	**430,090**	**0.97**	**114,851**	**0.77**
Production of inputs	25,977	0.45	63,977	0.59	412,063	0.93	102,257	0.68
Transportation of inputs	6,763	0.12	7,533	0.07	18,027	0.04	12,594	0.08
				ON-FARM				
Animal emissions	**352,677**	**6.08**	**858,601**	**7.95**	**3,150,932**	**7.13**	**1,686,361**	**11.23**
Enteric fermentation (CH_4)	352,600	6.08	858,392	7.95	3,150,200	7.13	1,686,074	11.23
Manure (CH_4)	77	0.001	209	0.002	732	0.002	287	0.002
Direct soil emissions	**74,494**	**1.28**	**114,138**	**1.06**	**284,426**	**0.64**	**130,651**	**0.87**
Excreta (N_2O)	29,535	0.51	76,438	0.71	252,173	0.57	110,130	0.73
N Fertilisers (N_2O)	-	-	630	0.01	9,363	0.02	-	-
Crop residue (N_2O)	40,655	0.70	25,520	0.24	-	-	-	-
Pasture residue (N2O)	4,304	0.07	11,549	0.11	22,890	0.05	20,521	0.14
Indirect soil emissions	**5,907**	**0.10**	**14,579**	**0.13**	**51,371**	**0.12**	**27,100**	**0.18**
Leaching and run-off (N_2O)	-	-	-	-	-	-	-	-
Atmospheric deposition (N_2O)	5,907	0.10	14,579	0.13	52,143	0.12	27,100	0.18
Other on-farm emissions								
Farm machinery operation (CO_2)	8,367	0.14	19,635	0.18	37,532	0.08	22,054	0.15
Liming (CO_2)	-	-	66,610	0.62	90,567	0.20	-	-
Urea hydrolysis (CO_2)	-	-	-	-	7,040	0.02	-	-

[a] Values presented for the Wickepin case study farm are post-allocation of emissions between wool and meat production.

[b] Total annual emissions quantity prior to allocation for wool production was 1,858,448 kg CO_2-e.

[c] EI prior to allocation for wool production was 17.21 kg CO_2-e/kg LW produced for sale.

Along with EI, the carbon footprint of each case study farm could be explored through the total GHG emissions produced. Table 5.1 also provides the emissions quantity (EQ), expressed in kg CO_2-e, for each emission source. Compared to the EIs of the respective enterprises, the sheep production enterprises, Bremer Bay and Wickepin, produced the lowest whole-farm EQs, 474.19 and 1,145.07 t CO_2-e, respectively. By contrast the two beef production enterprises, Dongara and Lancelin, produced higher whole-farm EQs of 4,051.96 and 1,981.02 t CO_2-e, respectively.

It is important to consider both the EI and the EQ of a livestock production system. For example, Bremer Bay, with the lowest EQ over a production year, also produced the least amount of saleable liveweight (57,975 kg; Chapter Four). Dongara by contrast, had the greatest EQ of the four enterprises, but also produced the most saleable liveweight, totalling 442,075 kg/yr (Chapter Four). Yet both of farms produced the lowest EIs of the enterprises considered. This demonstrates the importance of considering both the overall GHG profile of an enterprise and its productivity (in terms of capacity to produce saleable liveweight), as both play roles in determining the emissions efficiency of the system.

5.2.1 Emission hotspot identification

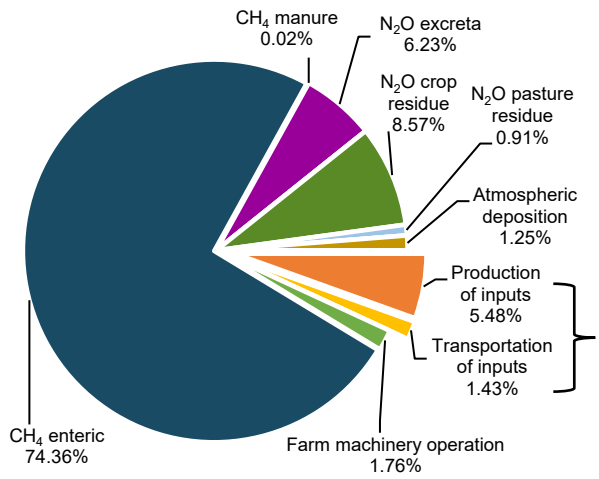
Following the carbon footprint analysis, this Section identifies the emission hotspots of each of the livestock production systems, both on-farm and pre-farm. The respective contributions of feedbases, stock classes and management practices to these hotspots, are explored in later sections of this Chapter.

For each of the four examined livestock enterprises, the primary whole-farm emission hotspot was enteric methane production. In terms of total emissions, the proportion attributed to enteric methane production was lower for the two sheep enterprises (74.36%, Bremer Bay; 74.96%, Wickepin; Figure 5.2) and Dongara (77.75%; Figure 5.3), than for Lancelin (85.11%). This difference was more a reflection of the respective contribution of crop residues and inputs versus enteric methane to the whole-farm GHG profile. For example, at Bremer Bay, which produced supplementary feed crops and maintained a feedlot, total enteric methane output was 352,600 kg CO_2-e whilst total emissions from crop residue and inputs was 73,395 kg CO_2-e, 21% of the value of enteric methane. Similar results were obtained for Wickepin, which produced supplementary feed crops and applied lime. Total emissions from crops and inputs were 163,641 kg CO_2-e, 19% of the total contribution by enteric methane. Whilst Dongara did not produce supplementary feed on-farm, it ran a more intensive system with high annual rates of fertiliser, along with a liming program. This is

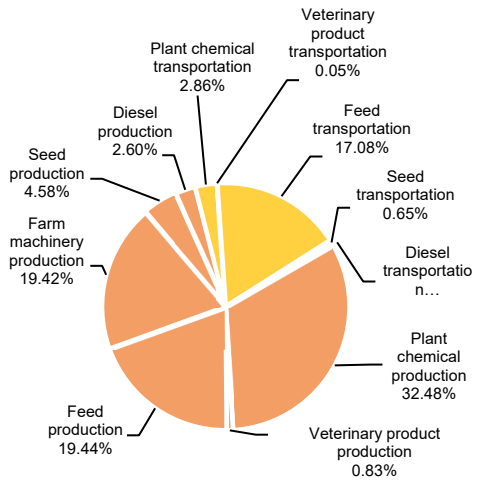
reflected in the whole-farm emissions. Despite producing the most enteric methane out of the four enterprises (3,150,200 kg CO_2-e), emissions pertaining to inputs and their application were 17% (527,697 kg CO_2-e) of the contribution by enteric methane. By contrast, Lancelin was a low input system; it did not produce supplementary feed crops, nor did it conduct regular applications of fertiliser or lime. This is reflected in the whole-farm emissions, with emissions from inputs totalling 114,851 kg CO_2-e, less than 7% of the contribution by enteric methane. This was despite enteric methane production totalling almost half that of Dongara, totalling 1,686,074 kg CO_2-e, reflecting the impact of the low-input system regardless of the size of an enterprise.

BREMER BAY

Total emissions

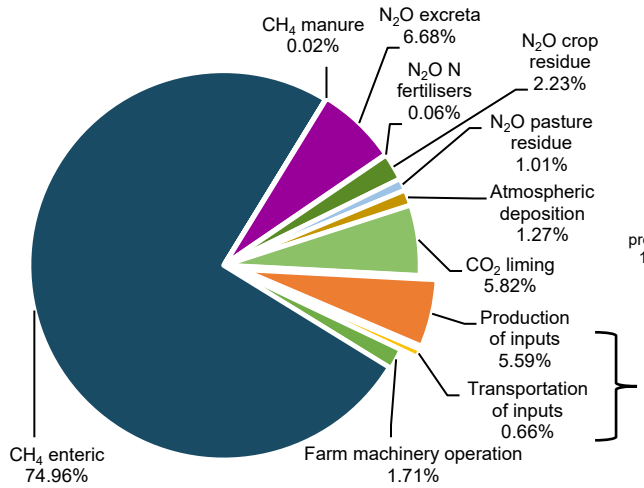


Pre-farm emissions



WICKEPIN

Total emissions



Pre-farm emissions

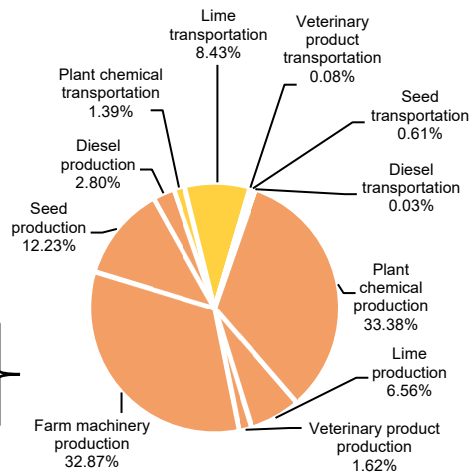


Figure 5.2 - Contribution of each GHG emission source to the total carbon footprint of the two sheep production enterprises, Bremer Bay and Wickepin

Examination of the enteric methane production hotspot in terms of EI presents a different picture. Whilst in terms of total emissions, the cattle enterprises produce at least double the enteric methane of the sheep enterprises, in terms of EI, the results varied. Bremer Bay presented the lowest EI, with the contribution from enteric methane to EI totalling 6.08 kg CO_2-e/kg LW. This was followed by the Dongara cattle enterprise (7.13 kg CO_2-e/kg LW), the Wickepin sheep enterprise (7.95 kg CO_2-e/kg LW) and finally the Lancelin cattle enterprise (11.23 kg CO_2-e/kg LW).

These EI results indicate that Bremer Bay was the most efficient at managing enteric methane production of the enterprises. This is likely a reflection of its multiple joinings each production year and grazing management across stock classes to match pasture supply with demand throughout the production year. Dongara only produced an additional 17% enteric methane per kilogram of saleable liveweight as compared to Bremer Bay, despite surpassing its whole-farm enteric methane EQ by almost nine-fold. The efficiency of Dongara is likely a result of not only the large quantity of saleable liveweight produced annually, but also that 88% of this liveweight was produced by backgrounded cattle (see Section 4.4 Chapter Four), which aren't accompanied by breeding herd emissions. By comparison, Lancelin produced 87% more enteric methane per kilogram of saleable liveweight produced than Bremer Bay and 58% more than Dongara. This likely reflects Lancelin's focus on its expanding breeding herd, with more enteric methane allocated to breeding stock which are not offset by saleable liveweight. Chapter Four commenced the analyses of the interactions between calculated emissions and management practices or enterprise characteristics. The following Chapters continue to explore these relationships in the context of carbon footprint analysis and emissions mitigation potential.

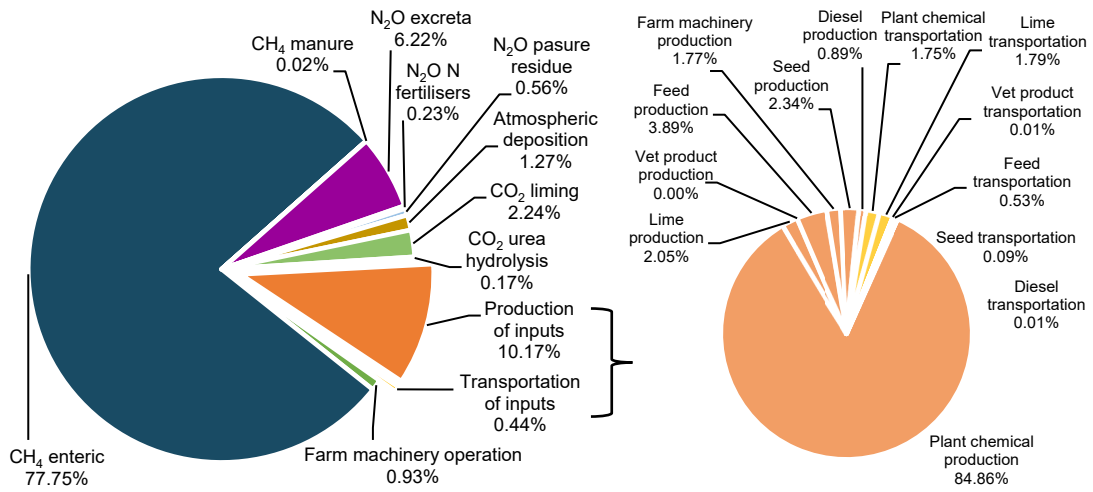
The second and third whole-farm emission hotspots varied across the four enterprises. At Bremer Bay, nitrous oxide emissions from the decomposition of crop residue contributed 8.57% (0.70 kg CO_2-e/kg LW). Of this 8.57%, 6.92% was contributed by the oat stubble and 1.65% by the lupin stubble. The third hotspot of the enterprise was nitrous oxide emissions from excreta, which contributed 6.23% (0.51 kg CO_2-e/kg LW). By contrast, at Wickepin the second hotspot was nitrous oxide emissions from excreta (6.68%; 0.71 kg CO_2-e/kg LW), followed by carbon dioxide emissions following the application of lime (5.82%; 0.62 kg CO_2-e/kg LW).

Though both sheep enterprises produced supplementary feed crops, the contribution of crop residues to total emissions at Wickepin, which also grew lupin crop for the purpose of supplementary feed, was only 2.23%, or 0.24 kg CO_2-e/kg LW, as compared to Bremer Bay. This difference was primarily a result of the different stubble

yields at the two locations, with the oat residue at Bremer Bay producing more than double emissions per hectare to the lupin residue at Wickepin, accompanied by higher lupin residues at Bremer Bay as compared to Wickepin (Sections 4.2 and 4.3, Chapter Four). Thus, despite the fact that Wickepin allocated more than double the arable land to the production of supplementary feed production as compared to Bremer Bay (300 ha versus 130 ha, respectively), total emissions arising from the decomposition of crop residue was almost half of that to Bremer Bay. This highlights the role of different locations, crop types and grazing management of stubble on crop residue and resultant emissions.

DONGARA

Total emissions **Pre-farm emissions**



LANCELIN

Total emissions **Pre-farm emissions**

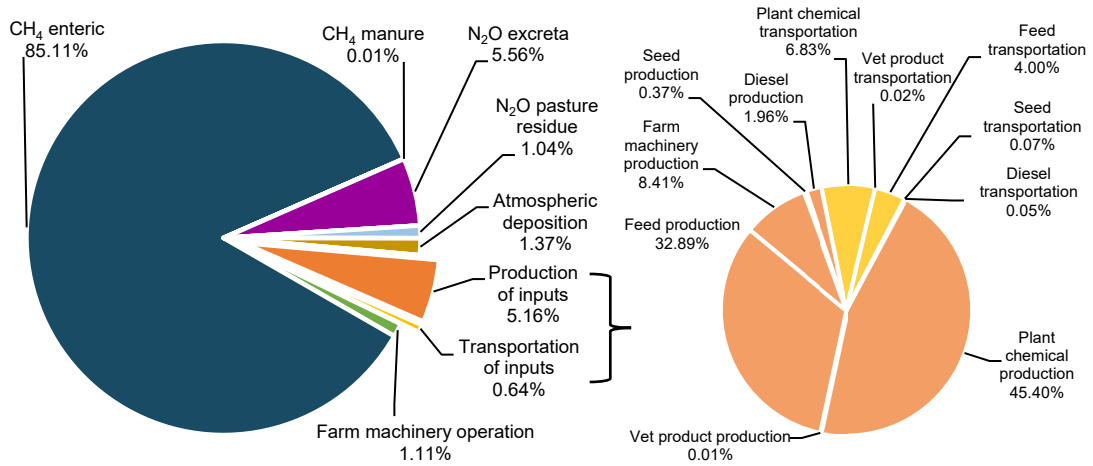


Figure 5.3 - Contribution of each GHG emission source to the total carbon footprint of the two beef cattle production enterprises, Dongara and Lancelin

Unlike for the other enterprises, the second hotspot at Dongara was emissions from the production of inputs, which contributed 10.17%. This value can be attributed to the large quantities of fertilisers applied to the pasture each production year. Despite presenting the second lowest overall EI of all the enterprises examined (9.17 kg CO_2-e/kg LW), the EI of the production of inputs from Dongara was the highest, at 0.93 kg CO_2-e/kg LW. This reflects the greater application of inputs at Dongara compared to the other enterprises. The contribution of such pre-farm inputs, along with transportation, to the carbon footprints of the farms is examined in Section 5.2.1.1. The third hotspot at Dongara was that of nitrous oxide emissions from excreta, totalling 6.22% (0.57 kg CO_2-e/kg LW). At the Lancelin enterprise emissions from excreta was instead the second hotspot, contributing 5.56% (0.73 kg CO_2-e/kg LW), followed by emissions associated with the production of inputs (5.16; 0.68 kg CO_2-e/kg LW).

Aside from enteric methane emissions, nitrous oxide emissions from excreta was the only other emission source calculated to be a hotspot across all four livestock production systems. Interestingly, the EI contribution of this emission source was similar for Wickepin and Lancelin (0.71 and 0.73 kg CO_2-e/kg LW, respectively). This was despite the differences exhibited in whole-farm EIs between the two enterprises, with the EI of the Wickepin sheep enterprise 20% less than that of the Lancelin beef cattle enterprise. Considering the findings of Chapter Four, the explanation for the similarity in nitrous oxide emissions from excreta is a result of the large amounts of supplementary feed, lupins, provided to sheep at Wickepin. Lupins have a high CP content, and this increased the overall CP content of total intake by the breeding flock over the six months supplied. This resulted in an increased excretion of N in dung and urine during this period, which is reflected in emissions. This is an example of the role management practices and feed type can have on the carbon footprint of livestock production. Considered together, enteric methane and nitrous oxide emissions from excreta contribute over 80% of whole-farm emissions at each enterprise, highlighting the importance of livestock emissions in the determination of the carbon footprint of livestock systems. The later Sections of this Chapter explore these emissions in greater detail, across feedbases, stock classes and the production year.

5.2.1.1 Pre-farm emission hotspots

This Section examines the pre-farm emission contributions to the carbon footprints of the four livestock enterprises. Figures 5.2 and 5.3 provide a breakdown of the pre-farm emissions, from both the production and transportation of inputs, produced by each case study farm.

As mentioned earlier, of all the farms examined, the Dongara beef cattle enterprise produced the highest emissions from the production of inputs, totalling 412,063 kg CO_2-e (0.93 kg CO_2-e/kg LW). Unsurprisingly, it also contributed the highest EQ in terms of the transportations of inputs, resulting in total pre-farm emissions of 430,090 kg CO_2-e, or 10.61% of the enterprise's carbon footprint (0.97 kg CO_2-e/kg LW). Of these pre-farm emissions, the production of chemicals, particularly pasture fertiliser, which was applied to all pasture types annually, made up almost 85%. The next highest contributor to pre-farm emissions were the components of the farm-produced mineral licks, producing only 3.89% of pre-farm emissions.

The other beef cattle enterprise, Lancelin, produced the second highest EQ and EI for both the production (102,257 kg CO_2-e; 0.68 kg CO_2-e/kg LW) and transportation (12,594 kg CO_2-e; 0.08 kg CO_2-e/kg LW) of inputs, with total pre-farm emissions contributing 5.80% to the carbon footprint. Like Dongara, Lancelin applied fertiliser to pasture, however the smaller size of the enterprise along with the use of less emissions-intensive chemicals mean that the EQ contribution of chemicals was 86% less than the former cattle enterprise. Unlike Dongara, Lancelin also provided externally sourced supplementary feed to agistment stock and operated on-farm machinery instead of contract machinery. The production of these three inputs contributed 45.40%, 32.89% and 8.41% to total pre-farm emissions, respectively. The application of less emissions-intensive chemicals offset any increased emissions associated with supplementary feed and machinery at Lancelin, resulting in an overall EI contribution of 0.77 kg CO_2-e/kg LW.

Of the two sheep enterprises, Wickepin produced the highest pre-farm emissions, in terms of both EQ and EI (71,510 kg CO_2-e, 0.66 kg CO_2-e/kg LW). Though between 38 and 83% less than the EQ of the cattle enterprises, these pre-farm emissions contributed 6.25%% to the carbon footprint of the enterprise. As was observed in the pre-farm emissions of the cattle enterprises, emissions from plant chemical and machinery production were the largest contributors (33.38% and 32.87%). However, the enterprise infrequently applied chemicals to the pasture and so these emissions were associated with the production of the lupin crop, not pasture. Notably, the emissions from Wickepin's liming program were greater for the transportation of the lime (8.43%) than the production of the lime in its raw form (6.56%) reflecting the large distance travelled to transport the lime to the farm.

The second sheep enterprise, Bremer Bay, produced the lowest pre-farm EQ and EI (32,740 kg CO_2-e, 0.56 kg CO_2-e/kg LW) of all the enterprises examined. This is a reflection of the low-input nature of the enterprise's operation and high productivity in

terms of saleable liveweight production. Despite this, emissions from the transportation of inputs made up the greatest proportion of total pre-farm emissions of all of the enterprises examined, totalling over 20%. The breakdown of these pre-farm emissions as shown in Figure 5.2, reveals that 17% of the emissions from transportation are attributed to the transport of supplementary feed components, namely the hay and feedlot ration components. This is reflective of the large distances required transport these products. The other primary contributors to the pre-farm emissions at Bremer Bay are plant chemical production, feed production and machinery production (32.48%, 19.44% and 19.42%, respectively), in line with the other enterprises included in the study.

5.3 Sheep production carbon footprint analyses

This section explores the whole-farm carbon footprint of the two sheep enterprises, by exploring the contributions of the various feedbases, stock classes and defining management practices to this carbon footprint.

5.3.1 Bremer Bay case study farm

5.3.1.1 Analysis of individual feedbases

The previous Section presented the calculated whole-farm carbon of the Bremer Bay enterprise. Producing total annual emissions of 474,185 kg CO_2-e/yr, or 8.18 kg CO_2-e/kg LW, the three dominant hotspots of the enterprise were; enteric methane production, nitrous oxide emissions from crop residues and nitrous oxide emissions from excreta. To address objective two of this study, this Section explores the contribution of each feedbase; pasture, crop stubble and the feedlot, to the carbon footprint of Bremer Bay.

The perennial pasture, kikuyu, contributed the greatest unadjusted emissions to the carbon footprint of Bremer Bay, totalling 52% (Figure 5.4). This was unsurprising given that kikuyu was the only feedbase to be grazed throughout the production year and also supported the greatest proportion of the Dorper flock. Annual pasture produced 26% of total emissions, oat stubble 13% and lupin stubble 5% and the feedlot 43%. Whilst these values are useful to identify which feedbases are the primary source of the enterprise's emissions, they do not provide an indicator of the productivity of each. That is, they do not distinguish between the area a feedbase covers, the flock numbers it supports, nor its ability to produce saleable product.

While Chapter Four presented and discussed the EI of the individual on-farm GHG outputs of the various feedbases at Bremer Bay, this Section presents the combined

pre- and on-farm contribution in terms of CO_2-e. Figure 5.5 presents the EI of each feedbase in terms of saleable liveweight produced on the respective feedbase. The EI values of each emission source can be found in Appendix F.

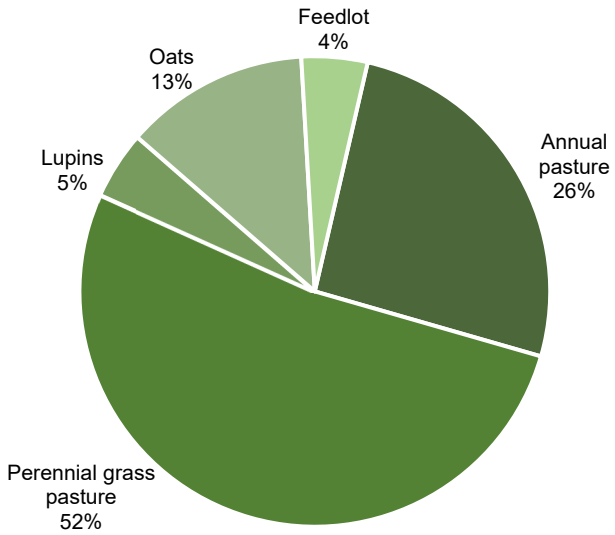


Figure 5.4 - Contribution of each feedbase (CO_2-e) to the overall carbon footprint of the Bremer Bay sheep enterprise

Though kikuyu pasture was the highest contributor to the overall carbon footprint of the enterprise, it only produced 8.28 kg CO_2-e/kg LW on the pasture. The dominant hotspot was enteric methane, followed distantly by excreta nitrous oxide emissions and emissions arising from the production of inputs. Unsurprisingly, the feedlot was the most productive of all the feedbases, with an EI of only 2.05 kg CO_2-e/kg LW. This reflects the intensive production of liveweight by lambs in the feedlot over short time periods. The EI of annual pasture was 8.10 kg CO_2-e/kg LW, 2% lower than that of the perennial pasture. Though this suggests that annual pasture was slightly more productive than kikuyu pasture, it is misleading, as annual pasture was only grazed during the growing months, during which two lambing periods occurred, and thus large quantities of saleable liveweight produced. During the non-growing season, all livestock were moved from annual pasture to the other feedbases. Without this option, livestock would have to graze annual pasture through the non-growing season during which annuals are senesced and of poor quality. This would require the provision of high levels of supplementary feed. This means that in the context of actual months grazed, the emissions efficiency of annual pasture was high, but in the context of a full production year and the requirement to graze livestock on other feedbases, actual efficiency was much lower. This is further explored in Chapter Six.

As expected, the supplementary feed crops were the least efficient feedbases, with lupin and oat crops producing, 25.82 and 38.57 kg CO_2-e/kg LW, respectively. The three hotspots of lupin stubble were enteric fermentation, followed by crop residue and then the production of inputs. Whilst the hotspots were the same for oat stubble, the greatest hotspot was crop residue. This can be attributed to the high stubble yield of the oats, more than double than that of the lupin crop (Chapter Four, Section 4.2.4). If emissions associated with crop residues were excluded, the EIs of the lupin and oat stubble were similar, producing 16.71 and 17.52 kg CO_2-e/kg LW, respectively. Regardless, the crops operated at less than a third of the productivity of both pastures, reflecting that crop stubbles were grazed by livestock only for maintenance purposes. The primary role of crops was the production of supplementary feed.

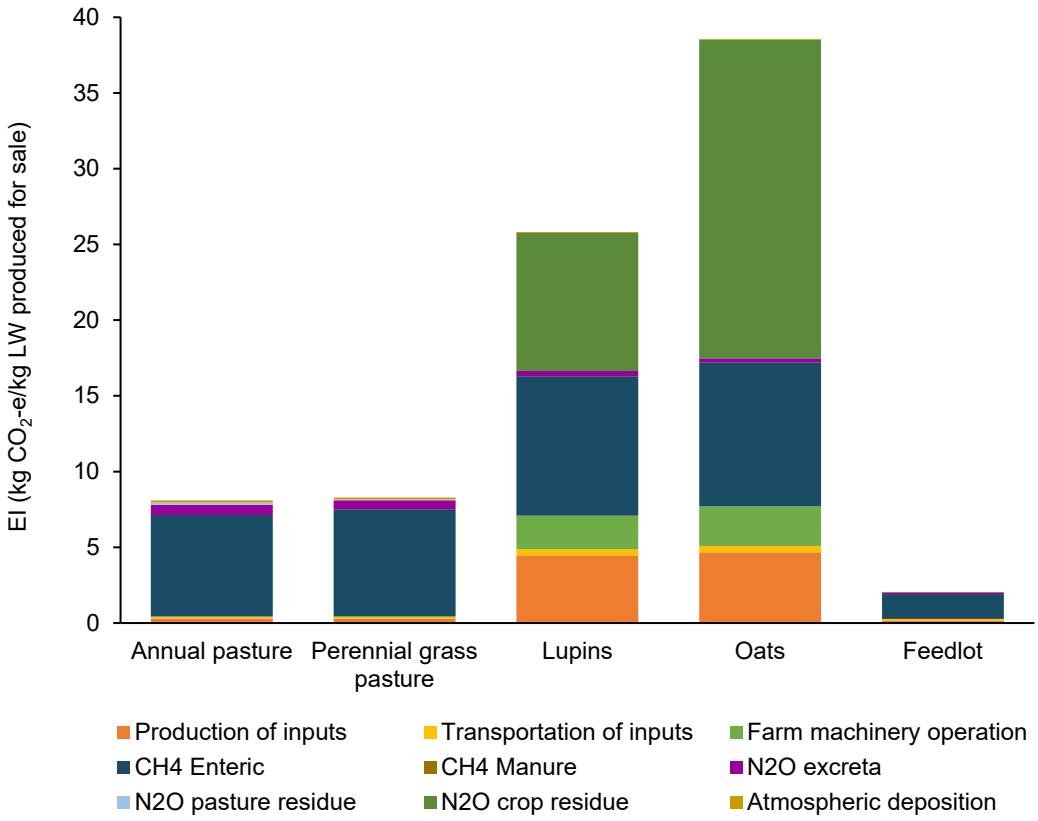


Figure 5.5 - The emissions intensity (EI; kg CO_2-e/kg LW produced for sale) of the pasture, supplementary feed crops and in the feedlot at the Bremer Bay sheep enterprise

The analysis of the respective hotspots of each feedbase at Bremer Bay enables an understanding as to how different management practices can influence the productivity of both individual feedbases and the whole enterprise. Across all the feedbases, aside from oat crop for which it was secondary, enteric methane production was the greatest hotspot. In terms of contribution to the total EI of each feedbase, enteric methane production was higher for stubble than for pasture,

comprising 9.20 and 9.50 kg CO_2-e/kg LW for lupin and oat stubble, respectively, as opposed to 6.69 and 7.06 kg CO_2-e/kg LW for annual and kikuyu pasture, respectively. This reflects the high quantities of supplementary feed required to meet livestock feed requirements whilst grazing poor quality stubble (Chapter Four, Section 4.2.5). Despite an overall feedbase EI only 2% higher than annual pasture, the EI contribution of enteric methane on kikuyu pasture was more than 5% higher than the annual pasture. This can be attributed to the higher overall intake of livestock across the respective grazing months, in conjunction with a higher overall proportion of lactating ewes and growing lamb on kikuyu pasture, both of which have higher intakes on a per kilogram of liveweight basis (see Chapter Four, Section 4.2.6 for detail). The breakdown of the carbon footprint across stock class types and across the months of the production year, as detailed later in this Section, highlight this further.

The second hotspot across the feedbases varied. For the annual and kikuyu pasture, the hotspot was nitrous oxide emissions from excreta, for lupins it was nitrous oxide emissions from crop residue, whilst for oats it was enteric methane emissions. Despite being the second hotspot of the two pasture types, the contribution of nitrous oxide emissions from excreta to each feedbase was significantly lower than the enteric methane, ranging from 7.08% to 8.27% as opposed to between 82.65% and 85.30%, respectively. As discussed in Chapter Four, livestock grazing green pasture with higher CP contents will produce more nitrous oxide than stock grazing dry, protein-deficient stubble. This was reflected in the EI contribution of nitrous oxide emissions to the feedbases, with the contribution on pasture more than double that of crop stubble. For example, the EI contribution on kikuyu pasture was 0.59 kg CO_2-e/kg LW whilst on oats it was 0.27 kg CO_2-e/kg LW. The EI contribution of nitrous oxide emissions from excreta on annual pasture was 13% higher than that on perennial pasture, again reflecting the findings of Chapter Four and indicative of the increased CP content of annual pasture during the actual months grazed.

The third hotspot for all pasture and crop feedbases at Bremer Bay was emissions associated with the production of inputs. For annual and kikuyu pasture, the contribution of this source to the feedbase emissions of each was small, 3.75% and 3.44%, respectively. On lupin and oat crops, the contribution was the higher, totalling 17.30% and 12.10%, respectively, reflecting the primary role of the crops. That is, the production of supplementary feed, an input-intensive operation. Emissions associated with the production of inputs, along with the transportation of inputs, were the second and third hotspots of the feedlot contributing 8.10% and 5.45%, respectively. As for the crop stubble, the feedlot was heavily reliant on inputs, specifically feed, and so

these hotspots are unsurprising.

The variation in hotspots across the feedbases present at Bremer Bay highlights the potential for targeted mitigation strategies and is examined further in Chapters Six and Seven.

5.3.1.2 Analysis of animal emissions at Bremer Bay

Enteric methane production was the primary emission hotspot for Bremer Bay and also four of the five feedbases. Together, enteric methane, manure methane and excreta nitrous oxide contributed 80.61% of the whole-farm carbon footprint (Figure 5.2). Given this contribution, it is useful to breakdown these emissions further into contribution by stock class. Unlike other emission sources, such as crop residue or inputs production, animal emissions can be allocated on a monthly basis rather than a lump figure across the production year. The Frameworks conducted monthly animal emissions calculations, permitting an investigation into which months produced the highest emissions and the monthly distribution of emissions across stock classes.

5.3.1.2.1 Breakdown of enteric methane emissions across stock classes

At Bremer Bay, the breeding Dorper ewes contributed 80.87% to total enteric methane emissions, followed by lambs (17.89%) and then rams (1.24%, Figure 5.6). This is expected, given the size and distribution of the breeding flock. Replacement ewes, which produced similar enteric emissions on a per animal basis to dry mature ewes (Chapter Four, Section 4.2.6), made up 25% of breeding ewe emissions. Lactating ewes, with the highest intake of all stock classes, produced 40% of ewe emissions over the combined seven months of the March and September lambings. The multiple lambings produced more liveweight for sale, but at the cost of higher emissions. The trade-off of this strategy is explored in Chapter Six. Of the two lambing cycles, the March lambing produced more enteric methane than the September lambing. This follows the findings of Chapter Four, which explored this on a per kilogram of saleable liveweight produced basis.

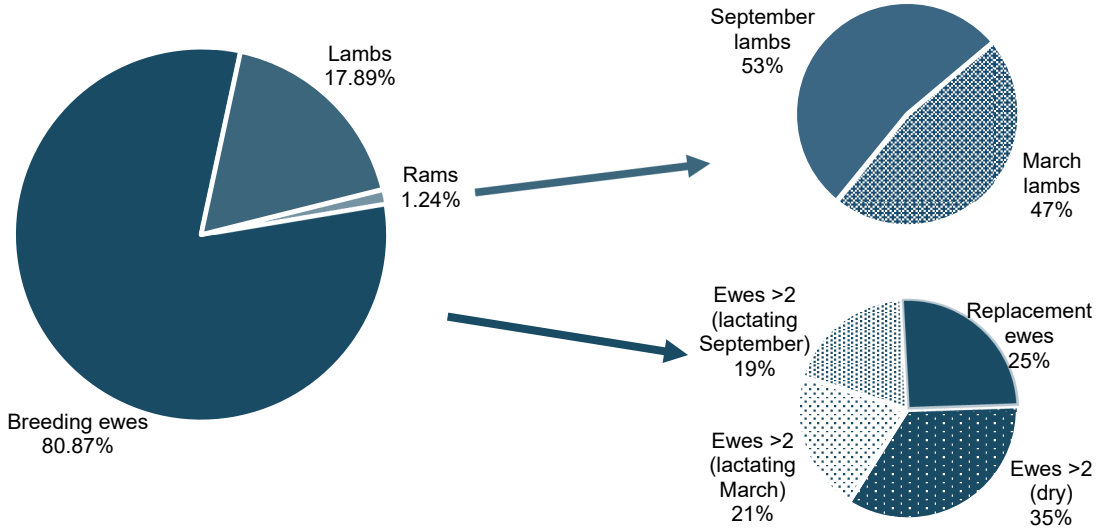


Figure 5.6 - Contribution of each stock class within the Dorper flock to total annual enteric methane emissions (CO_2-e) produced at the Bremer Bay sheep enterprise

5.3.1.2.2 Distribution of animal emissions across the production year

While Section 4.2.6 in Chapter Four investigated the influence of monthly pasture attributes and livestock physiological status on the emissions produced by an animal, this Section explores the influence of stock numbers and distribution over the production year on total animal emissions. Figure 5.7 presents total monthly animal emissions in CO_2-e, for the entire enterprise and also for each feedbase present at Bremer Bay. There is a distinct variability in the monthly whole-farm and also feedbase-specific animal emissions. The influence of the two lambing cycles are obvious, with whole-farm animal emissions increasing in the months following lambing in mid-March and mid-September, followed by a decline after weaning in January and July. The monthly contribution by each stock class as presented in Figure 5.7 confirms that this is, at first a result of the increased emissions produced by the lactating ewes, and then by the increasing emissions produced by the growing lambs.

A comparison of the total monthly emissions produced across the two lambing cycles reveals the higher emissions over the pre-weaning months of the March lambing than the respective months of the September lambing. This is also reflected in the increased monthly enteric methane emissions produced by each stock class across the two cycles, as displayed in Figure 5.8. The higher emissions can be attributed to the higher pasture quality over the March lambing cycle, as described in Chapter Four, Section 4.2.6. Emissions from nitrous oxide emissions support this, with higher emissions during the March cycle when the new season pasture growth had a higher CP content. The exceptions to this trend are April and December, where the increased

emissions were not a result of feedbase quality, rather the increased intake of supplementary feed to support the stock on the annual pasture.

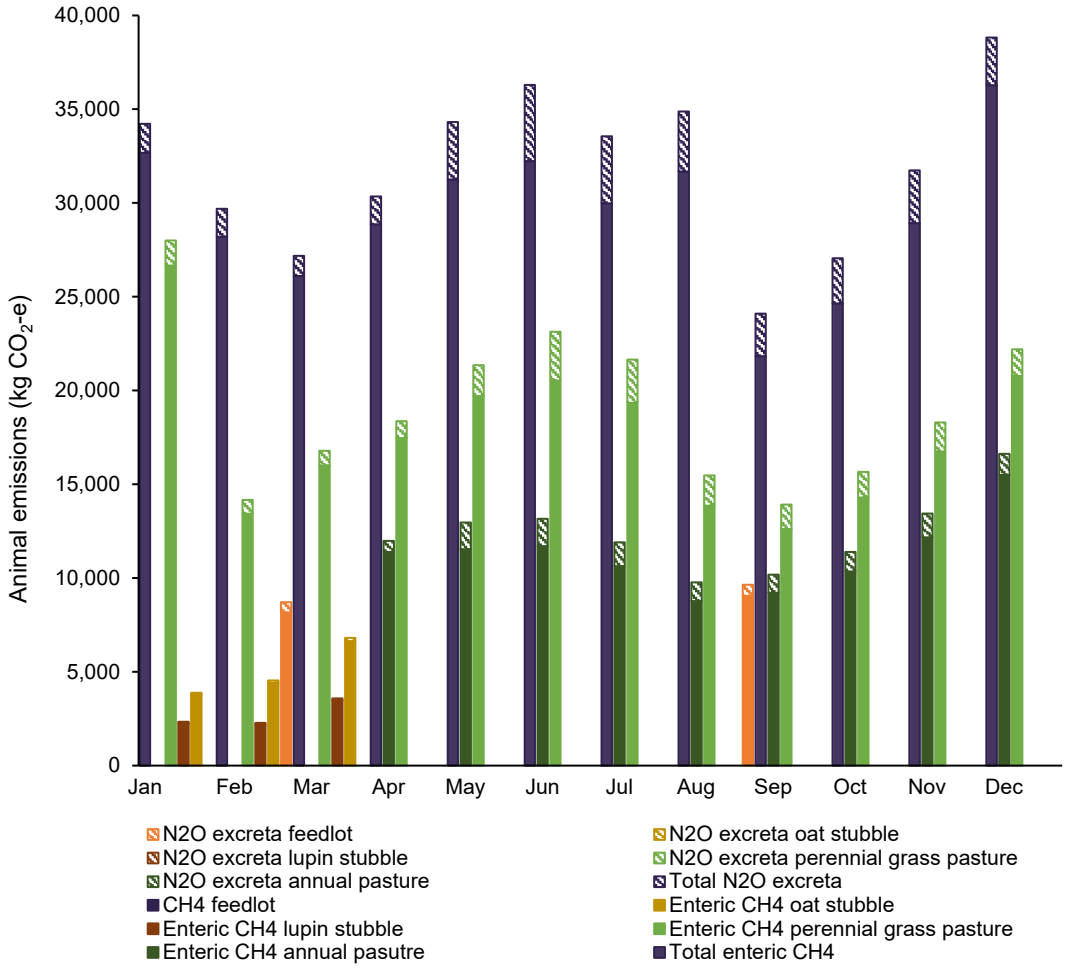


Figure 5.7 - Total monthly whole-farm and individual feedbase animal emissions (CO_2-e) at the Bremer Bay sheep enterprise

Where; solid bar is contribution through enteric methane production and striped bar is contribution through nitrous oxide emissions from dung and urine. Note: though methane from manure is included, its contribution is minor and not visible here.

Examination of the two pasture types reveal that kikuyu pasture produced more animal emissions than annual pasture through the production year. This is unsurprising given that it supported a higher proportion of the Dorper flock and highlights its ability to produce liveweight. As identified previously, kikuyu pasture was the most productive feedbase, with the lowest EI once year-round grazing was considered. Following the relationship identifed previously, from lambing to weaning, both pastures produced higher monthly emissions during the March lambing cycle, than the respective months in the September cycle. This is with exception of the pasture in December, where supplementary feed was provided to support the stock grazing the declining quality annual pasture.

The crop stubbles, while contributing the lowest overall monthly animal emissions, produced the highest EIs of all the feedbases. This highlights the inefficency of stubble to produce saleable liveweight. As shown in Figure 5.8, only dry mature stock were grazed on stubble, with growing stock moved to kikuyu. The supplementary feed requirement of stock grazing stubble, was considerable, comprising around half of total intake by March (Chapter Four, Section 4.2.6). This increased intake also increases enteric methane emissions, with little value in terms of prodouction of saleable liveweight. From both an emissions and a liveweight production perspective, this analysis reveals that there seems to be minimal benefit of grazing crop stubble over annual pasture during the non-growing season. Chapter Seven explores this practice of stubble grazing in the context of the whole-farm carbon footprint.

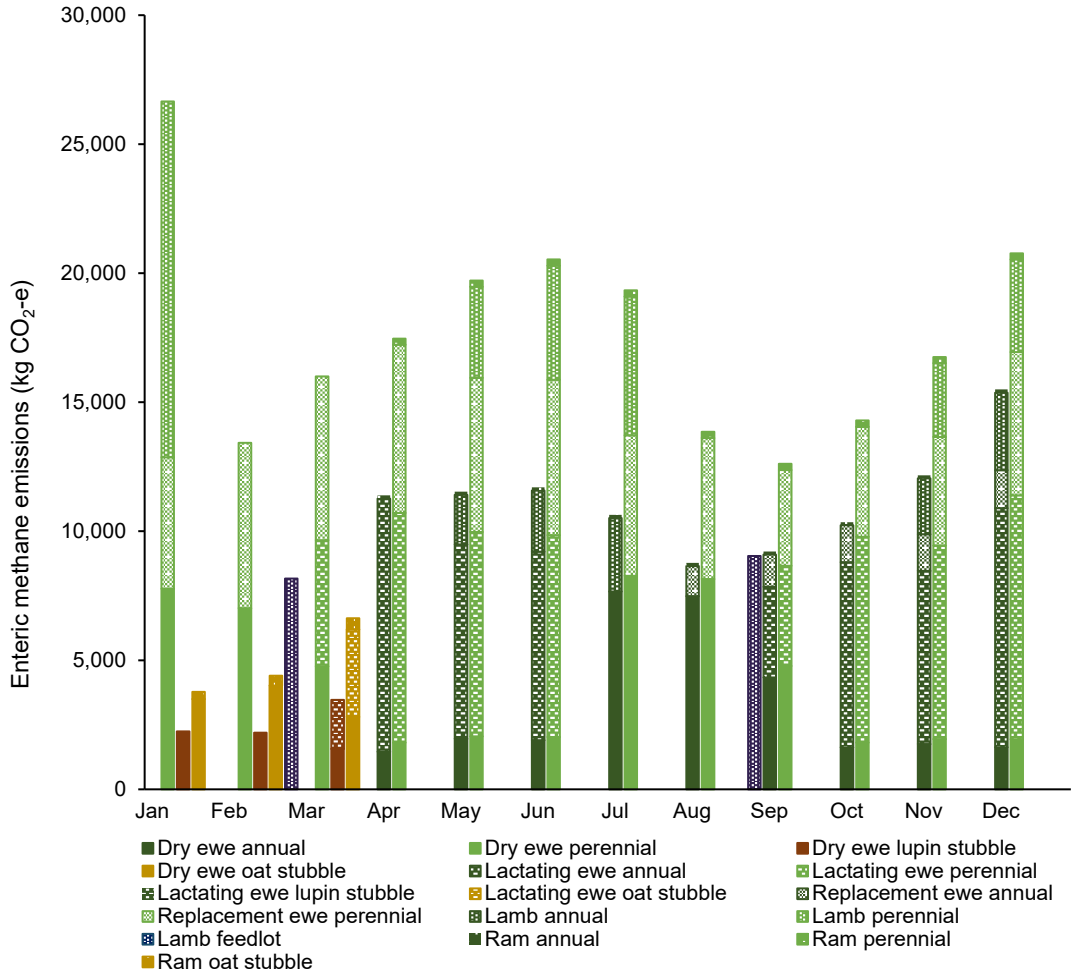


Figure 5.8 - Total monthly breakdown of enteric methane emissions (CO_2-e) within each stock class of the Dorper flock on each feedbase at the Bremer Bay sheep enterprises

The effect of supplementary feed on emissions can also be observed in January, where animal emissions from kikuyu were the highest of all the feedbases across the production year. The majority of these emissions were produced by the weaned

September lambs (Figure 5.8), all of whom were grazed on kikuyu pasture for the month, when neither annual pasture nor crop stubble were of sufficient quality to support the lambs in their final month of grazing. Despite the higher quality of the kikuyu during this period, supplementary feed was still required to support the growth of the lambs. As observed in Chapter Four, supplementary feed increases enteric methane production through the increased intake by the animal to meet requirements, above the intake of the feedbase. This is a trade-off of finishing lambs in the non-growing season.

5.3.2 Wickepin case study farm

5.3.2.1 Analysis of the individual feedbases

The annual pasture was the dominant source of emissions at Wickepin, contributing 71% of the total 1,145,073 kg CO_2-e produced each production year (Figure 5.9). The Wickepin enterprise reflected a more traditional model of livestock production, with livestock predominantly grazing annual pasture. The income crop stubble, grazed during the non-growing season, contributed 15% of total emissions, while lupin crop, grown for the purpose of supplementary feed and also grazed during the non-growing season, contributed 10%. Saltbush pasture, originally established to remediate salinity-affected area, produced only 4% of total emissions, reflecting the small area of established saltbush and low stocking rate during the months it was grazed.

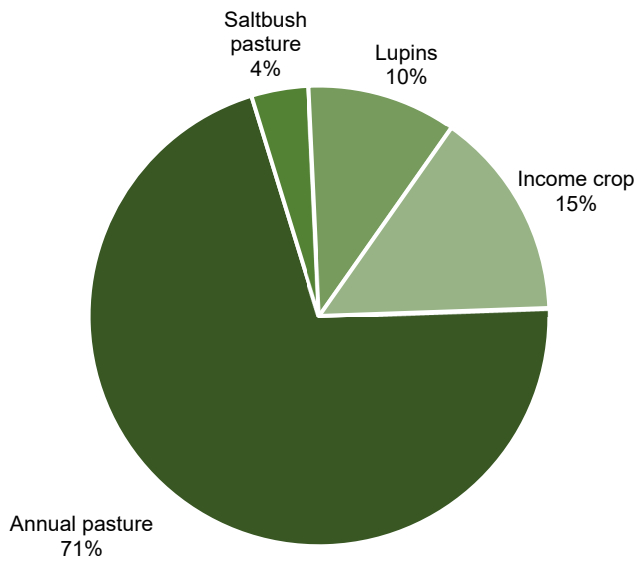


Figure 5.9 - Contribution of each feedbase (CO_2-e) to the overall carbon footprint of the Wickepin sheep enterprise

Figure 5.10 presents the EIs of the four feedbases at Wickepin. The corresponding EI values can be found in Appendix F. The annual pasture, as the primary grazing

source for the enterprise's Merino and SAMM flocks, produced the lowest EI of the feedbases, totalling 8.95 kg CO_2-e/kg LW. Saltbush pasture had a higher EI of 15.38 kg CO_2-e/kg LW. Unlike at Bremer Bay, whereby the EIs of annual and perennial pastures were similar (based on months grazed), the EI of saltbush was 72% higher than annual pasture. This reflects the secondary role of the perennial pasture at Wickepin, to support a small portion of the breeding flock during the non-growing season. This is explored later in this Section and in Chapter Six. The income crop stubble produced a slightly lower EI than saltbush pasture, totalling 15.05 kg CO_2-e/kg LW. This value is misleading however, as following the system boundaries established in Chapter Three, only emissions associated with the grazing of livestock on income crop stubble are included in the carbon footprint analysis. Emission sources such as the production and transportation of inputs, crop residue and liming application were attributed to the production of income crops. The primary hotspot of income crop stubble, enteric methane production, provides an indicator of the actual productivity of the crop. Enteric methane emissions contributed 14.14 kg CO_2-e/kg LW to the total EI of income crop stubble, 58% more than annual pasture, and the highest of the feedbases.

The lupin crop had the lowest productivity of all the feedbases at Wickepin with an EI of 37.59 kg CO_2-e/kg LW. Unlike the income crop stubble, lupin crops were produced for the purpose of providing on-farm supplementary feed. As such, emissions associated with the production and transport of crop inputs, machinery operation, liming application and the decomposition of crop residue were included in the analysis. Comparing the two crops without these emission sources, that is purely based on livestock emissions produced during the grazing of crop stubble, reveals that lupin stubble was actually more productive than the lower quality income crop stubble, producing 11.26 kg CO_2-e/kg LW, 25% less than the income crops. This reflects the higher quality of lupin stubble as opposed to the crops which comprised the income crop stubble (i.e. oats, wheat, barley and canola). Per the initial analyses conducted in Chapter Four, it can also be attributed to the fact that lupin stubble supported weaner wethers for one month prior to sale, as opposed to solely supporting the breeding herd, increasing the proportion of saleable liveweight supported on lupins as compared to income crop stubble.

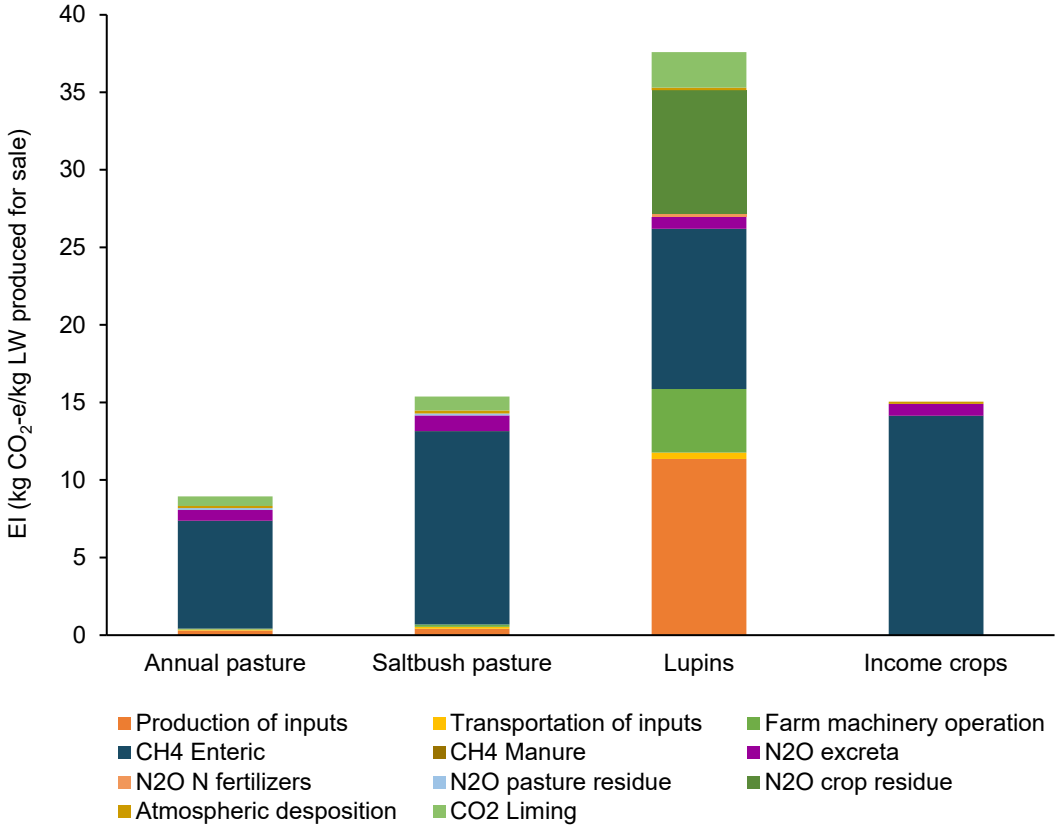


Figure 5.10 - The emissions intensity (EI; kg CO_2-e/kg LW produced for sale) of the pasture, supplementary feed crop and income crop stubble at the Wickepin sheep enterprise

The three primary hotspots for the annual and saltbush pasture were same as for the entire enterprise; enteric methane production, nitrous oxide emissions from excreta and carbon dioxide emissions following lime application. Despite being the primary hotspot for both pasture types, the contributions of enteric methane emissions to the EI of each differed substantially. Enteric methane contributed 6.95 kg CO_2-e/kg LW (77.73%) to the EI of annual pasture and 12.50 kg CO_2-e/kg LW (80.99%) to perennial pasture This was responsible for the majority of the observed difference in overall feedbase EIs between the two pastures and reflects the differences in the grazing management of the two pastures. Annual pasture played a critical role in the production of saleable liveweight at Wickepin, supporting the entire breeding herd for most of the production year, along with the production of the SAMM and Merino (aside from the final month) lambs destined for sale. By contrast, saltbush played a relatively minor role, supporting a portion of the mature dry breeding flock during the non-growing season only. This is explored further in Section 5.3.2.2.

The contribution of the second and third hotspots of the pasture types, nitrous oxide from excreta and carbon dioxide following liming, to the respective feedbase EIs was far less than enteric methane. The EI contribution of nitrous oxide from excreta was

0.69 and 1.10 kg CO_2-e/kg LW (7.70% and 6.58%) for the annual and saltbush pasture, respectively. For liming, the EI contribution was 0.62 and 0.90 kg CO_2-e/kg LW (6.98% and 5.85%) on the annual and saltbush pasture, respectively.

The three primary hotspots for income crop stubble were enteric methane production, nitrous oxide emissions from excreta and atmospheric deposition. These are all animal emissions, reflecting the exclusion of all emissions associated with the production of income crops in accordance with the system boundary. Of the three emission sources, enteric methane production was by far the most important, producing 94% (14.14 kg CO_2-e/kg LW) of the feedbase EI. The primary hotspots for the lupin crop, by contrast, were the production of inputs (11.40 kg CO_2-e/kg LW; 30.22%), enteric methane production (10.30 kg CO_2-e/kg LW; 27.46%) and crop residue (7.98 kg CO_2-e/kg LW; 21.22%). The large contribution of the non-animal emission sources to lupin crop EI, over 70%, as detailed in Figure 5.2, highlights the primary role of the lupin crop to produce supplementary feed and its secondary role as a grazing source for the flock.

An argument can be made for the allocation of emissions associated with the production of lupin grain across the feedbases according to consumption by livestock. If so, it was calculated that 47% of lupin grain was consumed on annual pasture, 4% on saltbush, 12% on lupin stubble and 38% on income crop stubble. The 84,249 kg CO_2-e produced annually as a result of the lupin crop production, can be allocated accordingly using these aforementioned percentages. It is included in the EI of the two pasture types and income crop stubble in the “production of inputs” emission source. The addition of emission impacts from lupin grain production has a minor effect on pasture, increasing the EI of annual pasture by almost 5% (9.38 kg CO_2-e/kg LW) and saltbush by just over 6%% (16.38 kg CO_2-e/kg LW), with no change to the primary hotspots. The allocation has a larger effect on income crop stubble, increasing the EI to 17.9 kg CO_2-e/kg LW (19% increase). This makes the production of inputs, driven by the lupins, the second hotspot, unsurprising given the reliance of livestock grazing this poor quality feedbase on the provision of supplementary feed (Chapter Four, Section 4.3.6). The allocation of emissions associated with the production of the lupins across feedbases decreases the EI of lupin stubble. While emissions associated with the grazing of stubble remain the same, the emissions associated with the production of lupins declines, with the overall feedbase EI falling by 62% to 14.34 kg CO_2-e/kg LW. Though this doesn’t change the hotspots it does change the order, with emissions from enteric fermentation the primary hotspot. It also makes lupin stubble more emissions efficient than income crop stubble, an expected result

given the higher proportion of saleable liveweight produced on lupin stubble to income crop stubble (Chapter Four, section 4.3.6).

5.3.2.2 Analysis of animal emissions at Wickepin

As for Bremer Bay, the carbon footprint analysis of Wickepin found that enteric methane production was the primary emission hotspot for the enterprise and also three of the four feedbases. This became the hotspot of all four feedbases if emissions associated with the on-farm production of the supplementary feed, lupins, were allocated according to consumption by livestock across the feedbases. In total, animal emissions contributed 81.66% to the whole-farm carbon footprint of Wickepin (Figure 5.2). The following sections examine these animal emissions in detail, across the stock classes and also the monthly distribution across the production year.

5.3.2.2.1 Breakdown of enteric methane emissions across stock classes

The breeding Merino ewes at Wickepin were the primary contributors of enteric methane, producing 89.02% of total emissions (Figure 5.11). Breaking this value down revealed that enteric methane contributed by dry and lactating Merino ewes was approximately 40% each, with the remainder contributed by replacement ewes. Though lactating Merino ewes produced on average 78% more enteric methane on a per animal basis than dry Merino ewes (Chapter Four, Section 4.3.6), the similar overall emission output by dry ewes can be attributed to a two key factors. Firstly, the lower weaning rates of Merino lambs meant that a higher portion of ewes remained dry through the production year. Secondly, the 600 cull ewes were kept on-farm over the growing season when pasture intake and thus emissions were higher, so they could be sheared in November prior to sale.

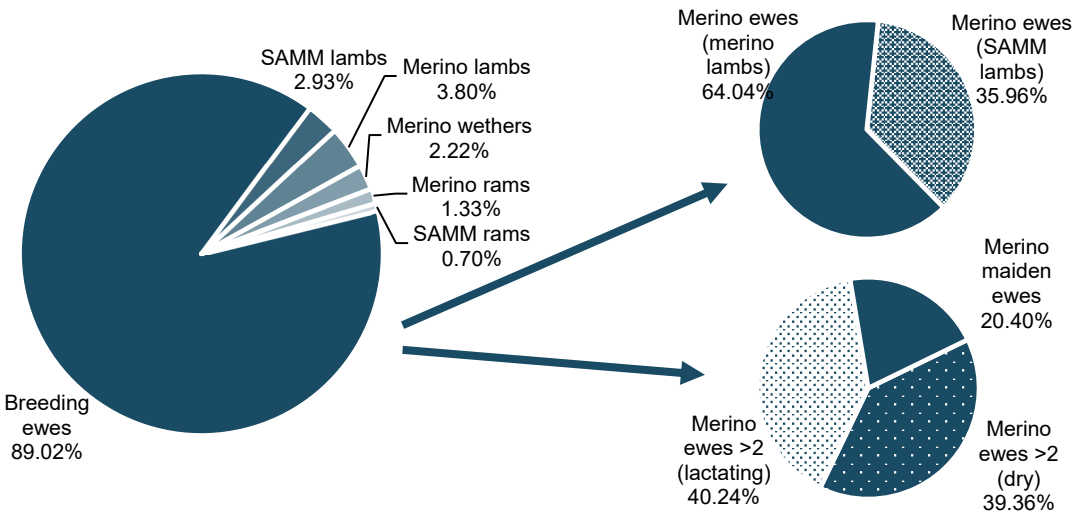


Figure 5.11 - Contribution of each stock class within the Merino and SAMM flocks to total annual enteric methane emissions (CO_2-e) produced at the Wickepin sheep enterprise

Of the two lambings that occurred each production year at Wickepin, in terms of enteric methane production, the Merino lambing (considering the lactating Merino ewes with a Merino lamb and also the Merino lambs) produced higher emissions than the respective SAMM lambing. However, taking into consideration that 1,600 Merino lambs were produced, compared to 800 SAMM lambs, total enteric emissions were higher for the production of SAMM lambs. Chapter Four (Section 4.3.6) investigated this further and found that in terms of actual liveweight produced, the production of a SAMM lamb produced less emissions than a Merino lamb. As for the multiple joinings at Bremer Bay, the trade-off here is emissions versus productivity. Whilst SAMM lambs produced higher daily enteric methane emissions than Merino lambs, they were sold at higher sale weights and with shorter turnoff periods. From an EI perspective, this increased productivity offset the concurrent increase in enteric methane output. The emissions benefits of producing the multiple breeds on an enterprise is considered in Chapter Six.

5.3.2.2.2 Distribution of animal emissions across the production year

The variation in whole-farm animal emissions across the production year is evident in Figure 5.12, with differences monthly differences of up to 46%. The dominant role of annual pasture in the production of livestock at Wickepin is evident. For the duration of the growing season, annual pasture was the sole grazing source for livestock. This period coincided with the highest monthly emissions of the production year, a reflection of the increased quality and availability of annual pasture during this period. It also reflected the SAMM and Merino lambing cycles, which meant there was a high proportion of lactating ewes and growing stock with increased intake requirements. As described in earlier sections, annual pasture was the most productive of the feedbases on the enterprise, and the high observed monthly emissions were offset by the greater proportion of liveweight produced on this pasture (84% of total saleable liveweight produced on-farm).

The role of both lupin and income crop stubble to support the maintenance requirements of the breeding flock during the non-growing season when annual pasture has senesced, is also evident. Chapter Four, Section 4.3.6, presented the feed intake and enteric methane produced by the respective stock classes on the various feedbases. It was observed that though the intake of sheep grazing stubble was restricted by the poor quality of the feedbase, the quantity of supplementary feed required to offset the deficit resulted in enteric methane emissions not too dissimilar to those produced by the same stock class grazing annual pasture during the growing season. Nitrous oxide emissions from excreta are noticeably lower during the non-

grazing season when the majority of the breeding flock were grazing the protein-deficient stubble, than the growing season when stock were grazing the green annual pasture with high CP content. As for enteric methane emissions, excreta nitrous oxide emissions were also higher from growing or lactating livestock. As annual pasture supported all lactating ewes and most of the growing stock classes, this compounded the difference between annual pasture and stubble. Chapter Four, Section 4.3.6 explains this in detail.

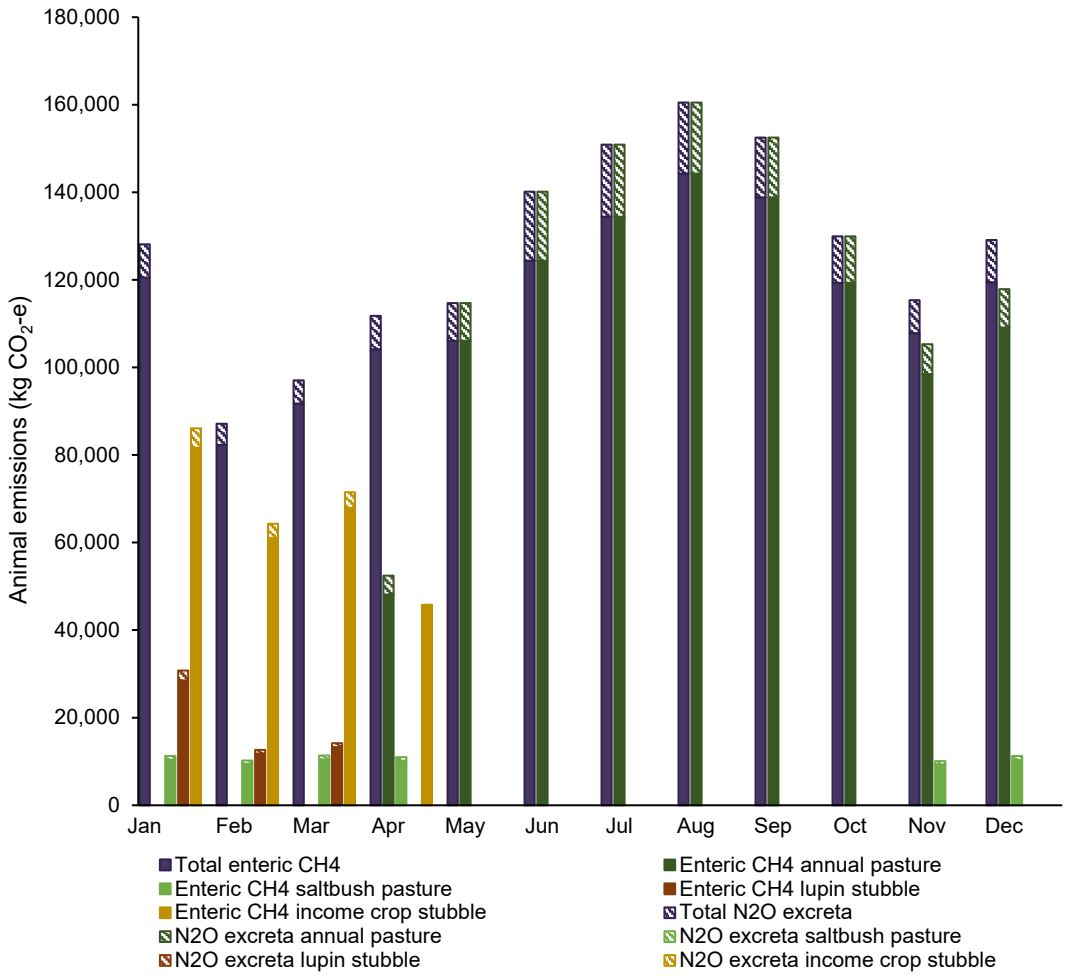


Figure 5.12 - Total monthly whole-farm and individual feedbase animal emissions (CO_2-e) at the Wickepin sheep enterprise

Where: solid bar is contribution through enteric methane production and striped bar is contribution through nitrous oxide emissions from dung and urine. Note: though methane from manure is included, its contribution is minor and not visible here.

Of all the feedbases, the saltbush contributed the lowest monthly animal emissions, as displayed in Figure 5.12. This can be attributed to the low stocking rate of the pasture and that only mature dry ewes were grazed. The feed intake results presented in Chapter Four, determined that stock grazing this pasture were able to meet their

daily requirements with less supplementary feed than the same stock class grazing stubble, resulting in lower animal emissions on a per kilogram of liveweight basis. The saltbush was utilised by Wickepin as a secondary feed source, with annual pasture and crop stubble prioritised. However, given these results, there seems to be an opportunity to increase the use of saltbush as a grazing source for the mature flock as a strategy to improve the efficiency of emissions production.

Figure 5.13, which displays the breakdown of monthly enteric methane emissions across the various stock classes grazed at Wickepin, provides insight into the effect of management decisions on the primary hotspot of the enterprise. The increased monthly emissions were associated with the SAMM lambing in April and then the Merino lambing in June. Previously, it was calculated that SAMM lambs, with the higher growth rates, produced higher daily emissions than the slower growing Merino ewes (Chapter Four, Section 4.3.6). Figure 5.13 shows that in August and September, the total enteric emissions from the four- and five-month old SAMM lambs were similar to the two- and three- month old Merino lambs. This also considers that SAMM lambs numbered 800 head to 1600 Merino lambs. Despite this, the faster turnoff of SAMM lambs and higher sale liveweight meant the enteric methane EI of producing SAMMs was lower than that of the Merino lambs.

Figures 5.12 and 5.13 show the decline in animal emissions over the final months of the growing season, reflective of the increasingly restricted feed intake across all stock classes as the pasture quality and availability declined. Whilst SAMM lambs were sold off-farm while pasture was still of good quality and availability, the weaned Merino lambs were retained until the end of January when wethers were sold and weaner ewes joined the breeding flock. Figure 5.13 details the ongoing emissions associated with retaining Merino wethers into the non-growing season. In December and January in particular, total farm animal emissions were boosted by the requirement to supplementary feed weaned merino lambs on top of the breeding flock. Emissions then fell as the weaner stock and cull ewes were sold off-farm and the remaining stock were retained at maintenance requirements. It is clear from this analysis that the interaction between such production events and feed supply will influence the whole-farm carbon footprint. Aspects of this are explored in Chapter Six.

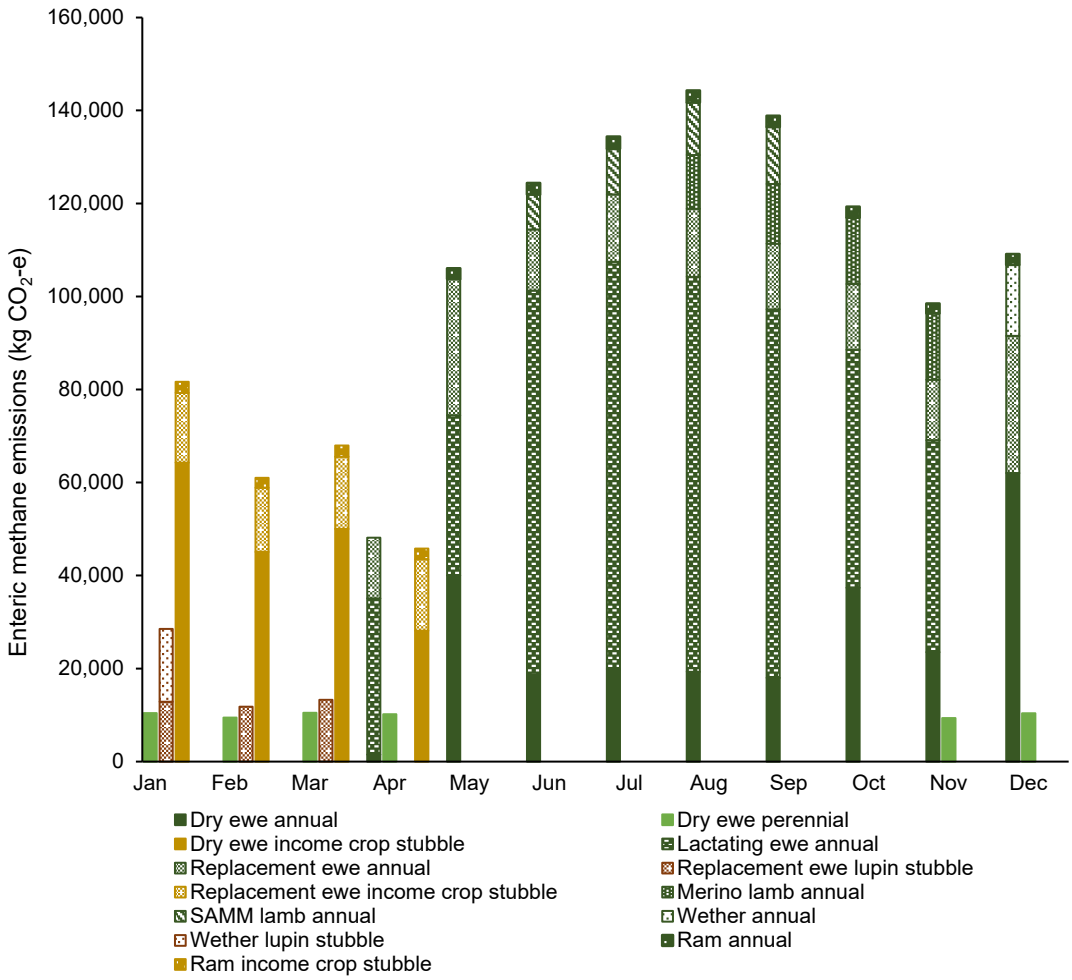


Figure 5.13 - Total monthly breakdown of enteric methane emissions (CO_2-e) within each stock class of the Merino and SAMM flocks on each feedbase at the Wickepin sheep enterprise

5.4 Cattle production carbon footprint analyses

As for the sheep enterprises. this Section examines in detail the whole-farm carbon footprint of the two cattle enterprises considered in this study. From this it was possible to explore the impact of various pasture types, the contribution of various stock classes and management practices to the carbon footprint of an enterprise.

It is important to note that, in line with the majority of livestock carbon footprint analyses (Hyland et al., 2016; Wiedemann et al., 2016b; Dougherty et al., 2018; Nieto et al., 2018; Bragaglio et al., 2018, for example), this study considered pre-farm emissions associated with purchased or agistment cattle outside the scope boundary. If it were possible to quantify these emissions, a process which would require a comprehensive carbon footprint analysis of the livestock systems these brought in cattle were sourced from in addition to the current analysis, the obtained carbon footprint would adjust accordingly. The degree to which this would alter the results

would be dependent on the efficiency of production in those systems under primary consideration and also on the method by which emissions would be allocated between the preceding system and the current system under consideration. While it is not possible to estimate the degree to which these pre-farm emissions would adjust the carbon footprint, it is important to remember that this variable could be subject to change.

5.4.1 Dongara case study farm

5.4.1.1 Analysis of individual feedbases

As discussed in Chapter Four, the breeding and pastoral herds at Dongara were rotationally grazed on three pasture types; annual pasture, subtropical grass pasture and tagasaste pasture. The subtropical grasses were the dominant grazing source, with over half the arable land dedicated to this pasture and responsible for 56% of total saleable liveweight produced. This is reflected in the carbon footprint of the enterprise, with emissions produced by subtropical grasses contributing 56.30% of the 4,051,957 kg CO_2-e produced each production year (Figure 5.14). Annual pasture, grazed throughout the production year, contributed 27.25% of total emissions, while tagasaste, which comprised the lowest portion of the farm's arable land, contributed only 14.55%. The feedlot, into which livestock were moved for a short period prior to sale, contributed the remaining 1.90% of the whole-farm carbon footprint.

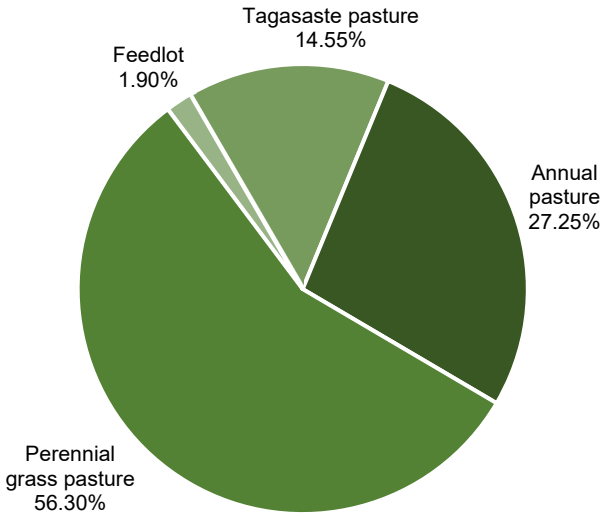


Figure 5.14 - Contribution of each feedbase (CO_2-e) to the overall carbon footprint of the Dongara beef cattle enterprise

Figure 5.15 presents the EIs of the four feedbases at Dongara. It is immediately evident that annual pasture is the least productive in terms of saleable liveweight, with

an EI of 12.12 kg CO_2-e/kg LW. This is line with the results presented in Chapter Four, Section 4.4.5, which found that annual pasture, whilst also grazed during the non-growing season as part of the farm's rotational grazing strategy, was unable to support the production of liveweight during those months, reducing its overall productivity. This contrasted to the two sheep enterprises which only grazed annual pasture during the growing season, boosting its productivity, with the lower productivity reflected in the stubbles which were grazed during the non-growing season. At Dongara, the grazing of annual pasture through its non-growing season period meant that it only produced 21% of total annual saleable liveweight, the same as the tagasaste pasture, despite covering double the arable farm area to that of tagasaste. The tagasaste and subtropical grasses, by comparison, were able to meet the requirements of the stock classes in both the breeding herd and the pastoral herd throughout most of the production year, resulting in lower EIs of 6.32 and 9.27 kg CO_2-e/kg LW, respectively. The feedlot EI was the second lowest of all the feedbases, after tagasaste, (6.69 kg CO_2-e/kg LW).

The similarity between the EIs of the tagasaste pasture and the feedlot can be attributed to two factors. Firstly, the high productivity of tagasaste, as described in Chapter Four, Section 4.4.5 and secondly, that the feedlot at Dongara is less centred on the intensive finishing of livestock. Though the Dongara feedlot finished weaner cattle from the small breeding herd, the majority of liveweight processed was that of livestock held in the feedlots on a maintenance ration in the days prior to sale. This was unlike the feedlot at Bremer Bay where weaner lambs were finished in feedlots specifically to meet target sale weights, and is reflected in the proportion of whole-farm saleable liveweight produced in the feedlot, totalling 18% at Bremer Bay as opposed to 3% at Dongara. Though the feedlot EI was still lower than other feedbases at Dongara, a reflection of its low input nature and that it only supported livestock destined for sale, the lower productivity in terms of the production of saleable liveweight, drove the higher EI than that typical of an intensive finishing system.

The primary hotspots of the three pasture types at Dongara aligned with the primary hotspots of the entire enterprise, as described in Section 5.2.1; enteric methane production, emissions associated with the production of inputs and nitrous oxide emissions from excreta. Of the three pasture types, the contribution of enteric methane to EI was the lowest for tagasaste, totalling 4.95 kg CO_2-e/kg LW. This was 31% less than that of subtropical grasses (7.21 kg CO_2-e/kg LW) and 48% less than that of annual pasture (9.46 kg CO_2-e/kg LW). This lower contribution by enteric methane was largely attributed to the efficiency with which tagasaste pasture

produced saleable liveweight as described earlier in this Section, and in part because it was able to produce this liveweight with comparatively less enteric methane (Chapter Four, Section 4.4.5). The second hotspot, the production of inputs, resulted from the annual application of fertiliser to the entire arable area of the enterprise. Emissions associated with the production of fertiliser contributed 85%, 86% and 94% of this hotspot, for the tagasaste, annual and subtropical grass pasture respectively. The lower contributions of the fertiliser to tagasaste and annual pasture reflected the higher input nature of these two pastures (i.e. machinery operation, other chemicals) as opposed to subtropical grasses.

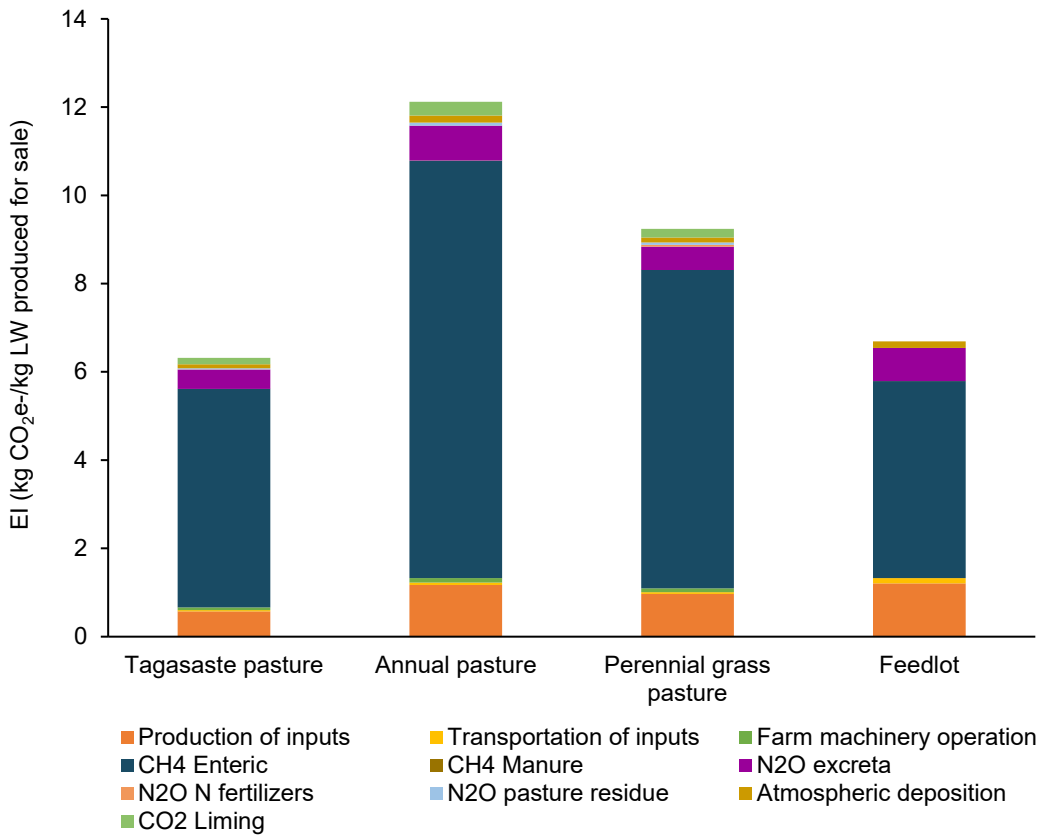


Figure 5.15 - The emissions intensity (EI; kg CO_2-e/kg LW produced for sale) of the pasture and the feedlot at the Dongara beef cattle enterprise

The third hotspot, nitrous oxide emissions from excreta, was responsible for a small proportion of total emissions produced on each pasture, between 5.72% and 6.93%, as opposed to enteric methane and emissions arising from the production of inputs which contributed between 77.82%-78.35% and 9.08%-10.44%, respectively. Regardless, it was possible to identify differences in the EI contribution of excreta emissions, with the contribution to tagasaste the lowest, totalling 0.44 kg CO_2-e/kg LW, followed by perennial grasses (0.53 kg CO_2-e/kg LW) and annual pasture (0.79 kg CO_2-e/kg LW). As for enteric methane, these differences can largely be attributed

to the respective efficiencies with which each pasture produced saleable liveweight. However, the CP content of each pasture also influenced these results. The increased CP content of tagasaste pasture as compared to annual pasture and subtropical grasses, was reflected in the differences between the EI contributions of each. For example, the EI contribution on tagasaste was only 16% lower than subtropical grasses, half of that exhibited for enteric methane, driven up by the increased CP intake and thus N excretion on tagasaste, as shown in Chapter Four. The results of this third hotspot demonstrate that whilst emissions from one emission source may be higher (i.e. higher excreta emissions on tagasaste), the declines in other hotspots and productivity gains (i.e. increased saleable liveweight production), can offset such increases. This highlights the importance of the whole-farm approach.

The primary hotspots of the feedlot differed from pasture, with enteric methane production (4.46 kg CO_2-e/kg LW; 66.73%), production of inputs (1.21 kg CO_2-e/kg LW; 18.04%) and nitrous oxide from excreta (0.75 kg CO_2-e/kg LW; 11.21%) the three key emission sources. Unlike for pasture, the emissions associated with the production of inputs was not related to chemical production. Rather it was from the production of the feed, contributing over 18% of the total EI of the feedlot.

5.4.1.2 Analysis of animal emissions at Dongara

The carbon footprint analysis of the Dongara enterprise found that, like the sheep enterprises, enteric methane production was the primary hotspot, contributing between 66.73% and 78.35% of the EI of each feedbase on the farm. Section 5.5.3 of this Chapter provides explanation as to the differences between sheep and beef cattle production. In total, animal emissions contributed 83.99% to the whole-farm carbon footprint of Dongara (Figure 5.3). The following two Sections examine these animal emissions in detail, across the stock classes within the breeding and pastoral herds and the monthly distribution across the production year.

5.4.1.2.1 Breakdown of enteric methane emissions across stock classes

The pastoral herd contributed over 80% of total enteric methane emissions produced at Dongara each production year, with the remainder attributed to the breeding herd (Figure 5.16). Of the stock classes grazed within the pastoral herd, pastoral heifers produced the highest emissions, totalling 44.91%, followed by pastoral steers (24.33%) and then pastoral bull calves (11.60%). Though pastoral steers produced more enteric methane on a per animal basis, pastoral heifers were supported in higher numbers and were held on the property longer (Chapter Four, Section 4.4.5), resulting in higher overall enteric methane emissions. The pastoral bull calves, contributing the

lowest overall enteric emissions of the pastoral stock classes, consumed the lowest amount of feed and were retained on the property for the shortest period, three months.

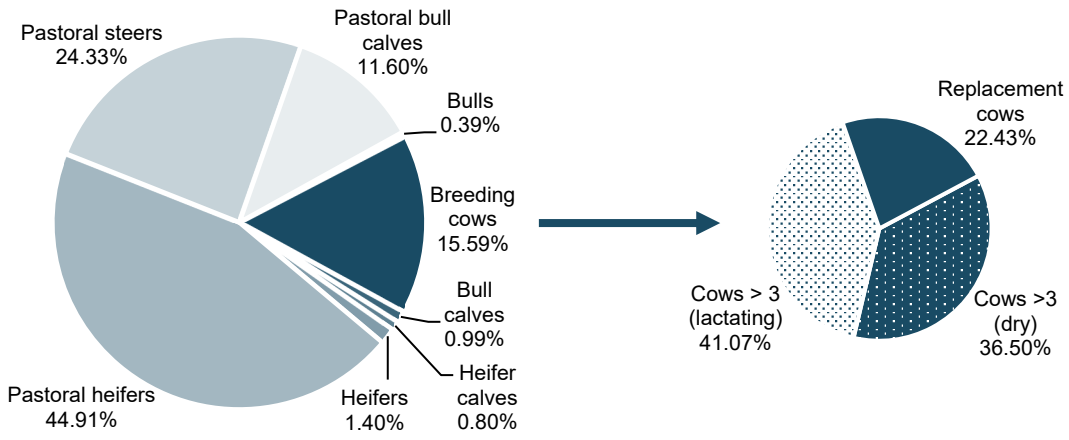


Figure 5.16 - Contribution of each stock class within the breeding and pastoral herds to total annual enteric methane emissions (CO_2-e) produced at the Dongara beef cattle enterprise

In the breeding herd, the breeding cows were the greatest source of enteric methane emissions, responsible for over 81% of the breeding herd's emissions. Figure 5.16 reveals that lactating cows contributed over 41% of emissions produced by the breeding cows at Dongara. Though lactation only occurred for 6 to 7 months of the year, this reflects the effect of higher intake on emission output during this period. The dry or pregnant mature cows produced only slightly lower emissions to the lactating cows. This is because the value not only includes emissions from dry stock during the non-calving months, but also emissions from non-lactating cull cows during the calving season. This coincides with the growing season, during which intakes and thus emissions are higher. Replacement cows, comprised of weaner heifers, to heifers at first joining, to heifers at second joining (23% of total breeding cow herd numbers), comprised the remaining 22.43% of total breeding cow emissions. The target product of the breeding herd, bull calves, only contributed 0.99% of total enteric emissions at Dongara, or 1.05% of breeding herd emissions.

Though the pastoral herd produced the greatest proportion of enteric methane emissions, in terms of emissions per kilogram of saleable liveweight produced, all three pastoral stock classes were more efficient than the bull calves or heifers produced by the breeding herd. Further detail regarding this comparison can be found in Chapter Four, Section 4.4.5.

5.4.1.2.2 Distribution of animal emissions across the production year

Examination of monthly animal emissions highlights how certain management practices can influence the overall carbon footprint of a beef cattle enterprise. Figure 5.17 shows the total monthly animal emissions for each feedbase across the production year at Dongara. Immediately it is evident that grazing was managed in such a way to coordinate animal demand with pasture supply, with monthly emissions varying by up to 91% across the production year. Animal emissions increased from the beginning of the annual growing season in June and declined from the end of the growing season in October. From March through to May, when annual pasture quality was at its lowest, emissions were also at their lowest. Though this variation throughout the production year can be partly attributed to variations in the quality and availability of the feedbases grazed, it is mostly a result of deliberate changes to the grazing intensity of pasture through specific management decisions.

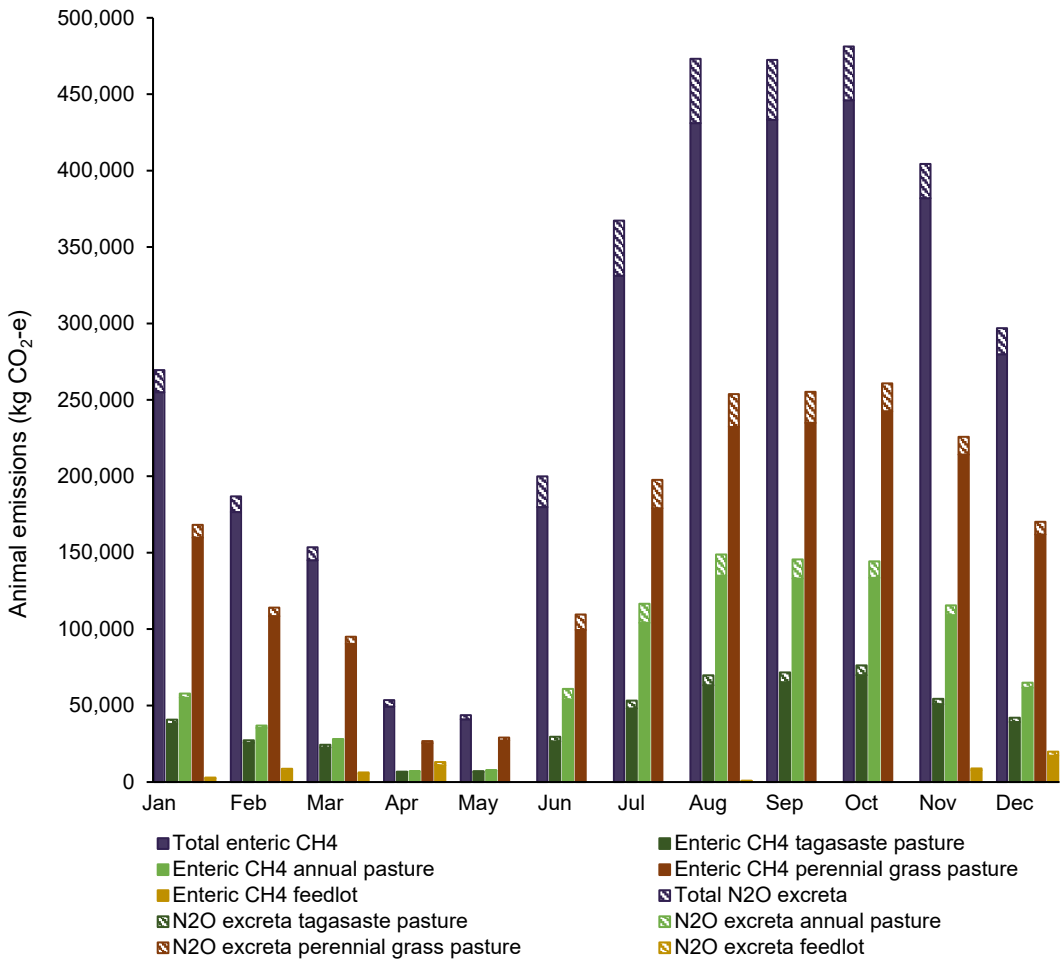


Figure 5.17 - Total monthly whole-farm and individual feedbase animal emissions (CO_2-e) at the Dongara beef cattle enterprise

Where: solid bar is contribution through enteric methane production and striped bar is contribution through nitrous oxide emissions from dung and urine. Note: though methane from manure is included, its contribution is minor and not visible here.

Section 4.4.5 in Chapter Four examined the correlation between monthly pasture attributes, the physiological status of stock classes and the monthly intake (and thus enteric methane output) of the various stock classes grazed at Dongara. Across the three pasture types, intakes of grazed stock were higher during the growing season, when pasture quality and quantity was greater than the non-growing season. This difference was less pronounced on tagasaste and subtropical grass pastures, which produced green feed throughout the production year, as opposed to annual pasture. Figure 5.17, in terms of total emissions across all livestock grazed, reflects these findings. The differences in the productivity of the pastures can be observed during the non-growing season months, where, despite covering double the arable land of the tagasaste pasture and supporting cattle as part of the enterprise's rotational grazing strategy, annual pasture produced similar monthly animal emissions as tagasaste. The role of the two perennial pastures, in particular subtropical grasses which acted as the dominant grazing source through the production year, in supporting cattle at Dongara is evident in Figure 5.17. Not only did they support liveweight production during the growing season alongside annual pasture, they also permitted the production of significant quantities of saleable liveweight for between four to five months beyond what a traditional annual pasture dominant enterprise would. The influence of perennials on the carbon footprint of an enterprise is examined in detail in Chapter Six.

Focussing on the enteric methane emissions hotspot, Figure 5.18 demonstrates how farm management practices also influenced the distribution of animal emissions over the production year. The first of the pastoral cattle were brought onto the farm in June, coinciding with the beginning of the growing season. The dominance of the pastoral cattle is evident through the months they are retained on-farm. The farm's efforts to match demand with supply is also evident, with pastoral cattle gradually sold off the farm from the end of the growing season, so that only the breeding herd remained by the end of March. As discussed previously, through the growing season, all three pastures were able to support the growth of the various pastoral stock classes. During the non-growing season, tagasaste and subtropical grasses continued to support the pastoral cattle, offsetting the deficit resulting from grazing the senesced annual pasture. The perennial pastures allowed the farm to retain pastoral cattle beyond the end of the growing season, maximising potential liveweight gains and enabling the targeting of certain market specifications.

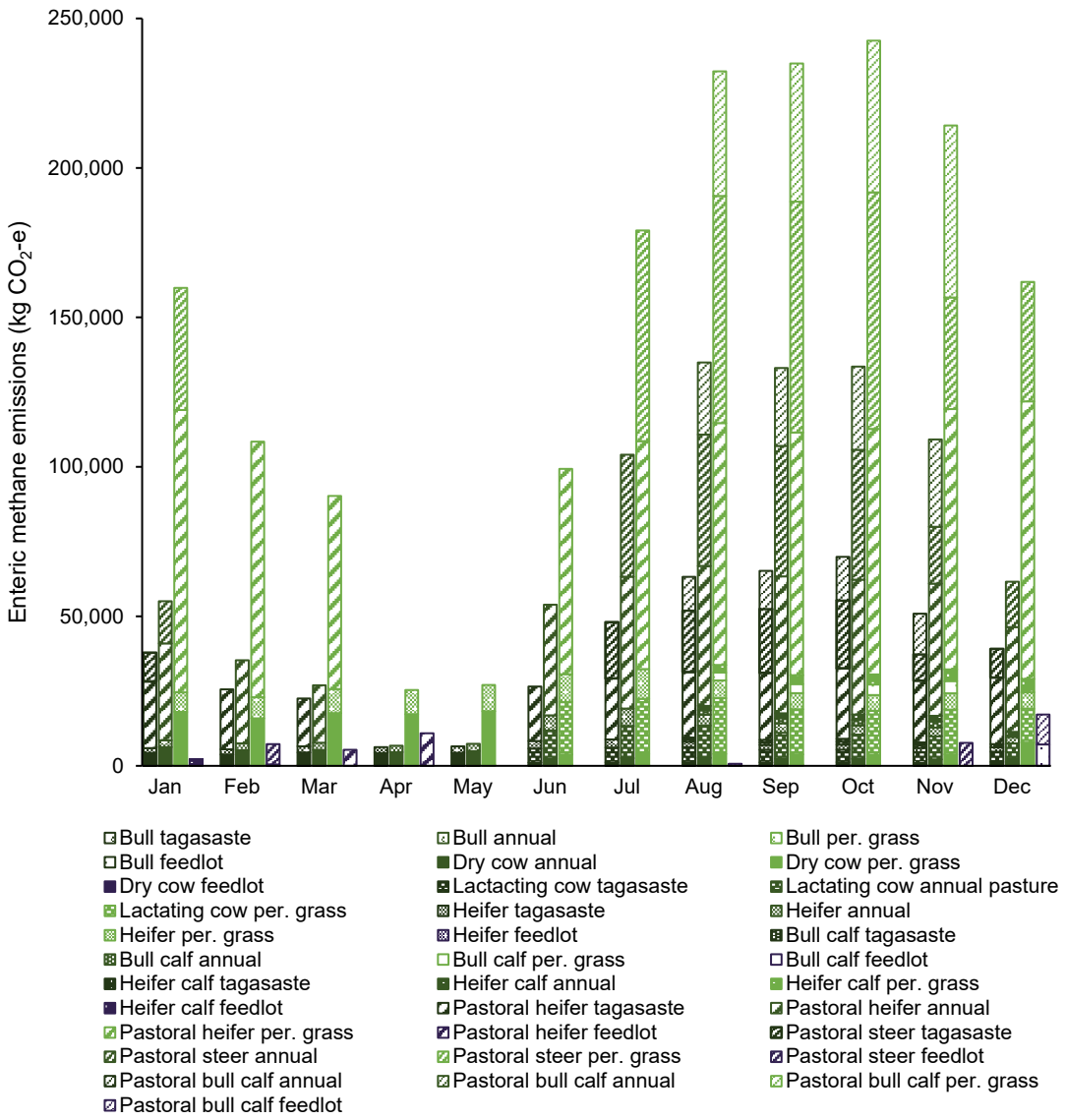


Figure 5.18 - Total monthly breakdown of enteric methane emissions (CO_2-e) within each stock class of the breeding and pastoral herds on each feedbase at the Dongara beef cattle enterprise

Dongara's management of its breeding herd according to pasture supply is evident in Figure 5.18. Calving occurred in June to coincide with the beginning of the growing season. All calves were weaned and sold or moved into the breeding herd by the beginning of January. This ensured that the grazing intensity of the breeding herd was highest when pasture quality and availability was at its maximum across the three pasture types and lowest in the non-growing season. This removed the need for supplementary feed and ensured that the perennial pastures were available for grazing by the pastoral cattle during the non-growing season. By reducing the grazing pressure to minimum levels late in the non-growing season (no pastoral cattle and a breeding herd with only maintenance energy requirements), the enterprise allowed the pastures to rest prior to the beginning of the next growing season and

commencement of peak liveweight production. The impact of such management decisions in terms of carbon footprint analysis is further examined in Chapter Six.

5.4.2 Lancelin case study farm

5.4.2.1 Analysis of individual feedbases

The primary pasture at Lancelin was tagasaste, with half the arable land of the beef cattle enterprise dedicated to its production. This was reflected in the whole-farm carbon footprint, where emissions associated with tagasaste contributed 60.35% of total emissions (Figure 5.19). The annual and subtropical grass pastures, which were also rotationally grazed throughout the production year and were allocated similar areas of the enterprise's arable land, contributed 18.92% and 20.44% to the overall footprint, respectively. By comparison, the feedlot, which was only used for a brief period during the weaning of bull and heifer calves, produced by the breeding herd, contributed only 0.29%.

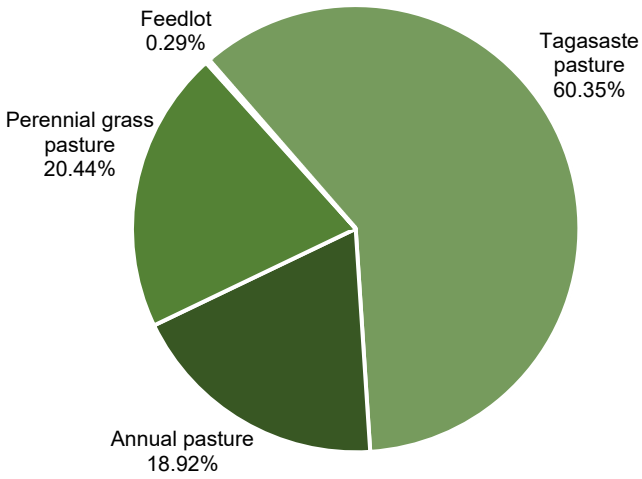


Figure 5.19 - Contribution of each feedbase (CO_2-e) to the overall carbon footprint of the Lancelin beef cattle enterprise

Despite producing total emissions amounting to more than three-fold higher than annual pasture, Figure 5.20 shows that the EI of tagasaste pasture was similar to annual pasture, producing 14.66 kg CO_2-e/kg LW compared to 14.21 kg CO_2-e/kg LW on annual pasture. As for Dongara, annual pasture was grazed throughout the production year. Though annual pasture at Lancelin was a primary grazing source for the breeding herd during the growing season when feed quality was high, during the non-growing season it was unable to support the production of saleable liveweight. This lowered the overall productivity of the pasture, as is reflected in its EI.

Despite covering a similar area of arable land to annual pasture, subtropical grasses produced 27% of total saleable liveweight produced at Lancelin, as compared to only 18% produced on annual pasture. The high productivity of the perennial grasses pasture can be attributed to its ability to support the requirements of livestock through the production year and its role as the primary grazing source for the agistment herd. This is reflected in the EI of the perennial grass pasture which was 9.84 kg CO_2-e/kg LW, the lowest of the three pasture types. As expected, the feedlot Lancelin, yielded the lowest EI of all the feedbases, producing 5.15 kg CO_2-e/kg LW.

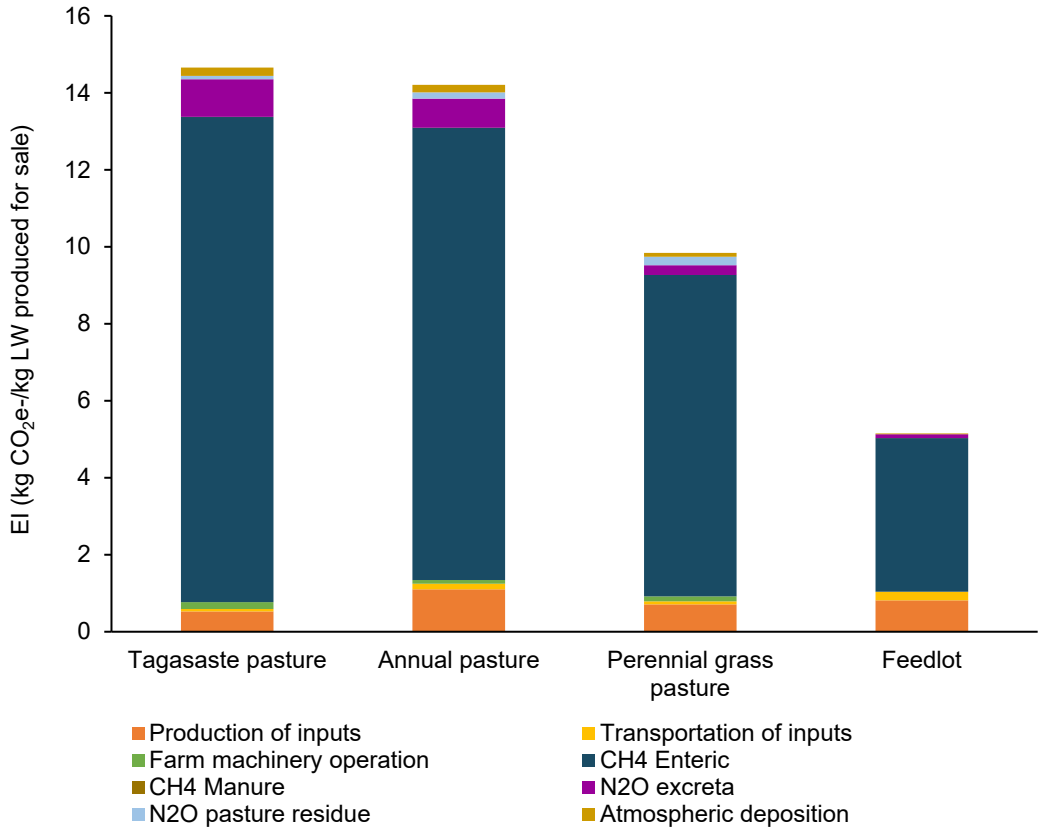


Figure 5.20 - The emissions intensity (EI; kg CO_2-e/kg LW produced for sale) of the pasture and the feedlot at the Lancelin beef cattle enterprise

The implications of the grazing management strategies employed at Lancelin are reflected in the feedbase EIs. For example, the EI of tagasaste was the highest of all the feedbases; however, this is not a reflection of its productivity, rather the livestock grazing it. As described in Chapter Four, tagasaste was the primary grazing source for the Lancelin breeding herd. While it supported the breeding herd during the growing season, which coincided with the calving and the production of saleable liveweight, the pasture was also the primary grazing source during the non-growing season when saleable liveweight was not being produced. This means that the EI of tagasaste is not representative of the important role that the pasture played in

supporting the maintenance of the breeding herd responsible for the production of the greater proportion of liveweight produced on-farm. This explains the large difference exhibited between the EI of tagasaste at Lancelin, as compared to Dongara, where livestock were rotationally grazed throughout the year with a high proportion of backgrounded cattle as opposed to breeding herd. Another example was that of the subtropical grass pasture. The EI of the subtropical grasses at Lancelin was the lowest of all the pasture types at the enterprise. This reflected that it served as the predominant grazing source for the agistment cattle and meant that produced emissions resulted primarily from the production of saleable liveweight, rather than the maintenance of liveweight to remain on-farm. This was a similar situation to that at Dongara, whereby pastoral cattle were the primary sources of saleable liveweight, and it reflected in the similar EIs of the subtropical grasses at the two enterprises. These findings demonstrate the potential role of grazing management on the carbon footprint of an enterprise and the importance of considering employed management practices when examining and comparing both feedbases and enterprises.

Enteric fermentation was the primary hotspot across the three pasture types and the feedlot at Lancelin, contributing between 77.48% and 85.99% towards the EI of each feedbase. The production of enteric methane in terms of saleable liveweight was similar for tagasaste and annual pasture, contributing 12.60 and 11.75 kg CO_2-e/kg LW, respectively. This again, reflects that a higher proportion of animal emissions produced on tagasaste pasture were produced by the breeding herd as opposed to stock destined for sale. For annual pasture, it reflects the lower productivity of the feedbase and represents the emissions produced during the non-growing season that were not offset by the production of saleable liveweight. Of the three pasture types, the contribution of enteric methane to feedbase EI was the lowest for subtropical grasses, totalling 8.35 kg CO_2-e/kg LW. As described in Chapter Four, the high productivity of the perennial grasses was likely because the pasture primarily supported the agistment herd, and so all liveweight gains contributed to saleable liveweight, lowering the EI of the pasture. This contrasted to the other two pasture types which primarily supported cattle within the breeding herd, of which a minor proportion were sold. The majority were responsible only for the indirect production of saleable liveweight, increasing the overall EI of these pastures. Enteric methane contributed 3.99 kg CO_2-e/kg LW, or 77% of the feedlot EI. Unlike the other feedlots/yards considered in this study, Lancelin feedlotted only weaner bulls produced by the breeding herd and only provided them with maintenance feed, hay. This meant that only a small proportion of whole-farm saleable liveweight was

produced in the feedlot (0.7%) and is reflected in the resulting EI, despite the low-input nature of this feedlot.

The second hotspot of the annual pasture (1.10 kg CO_2-e/kg LW; 7.77%), subtropical grasses (0.71 kg CO_2-e/kg LW; 7.21%) and the feedlot was emissions arising from the production of inputs (0.81 kg CO_2-e/kg LW;15.81%). This emission source was the third hotspot of the tagasaste pasture (0.53 kg CO_2-e/kg LW; 3.60%). The high contribution of emissions from the production of inputs to the EI of the feedlot was unsurprising and follows the findings of the other feedlots. However, the high contribution to annual pasture and subtropical grasses was less expected, particularly as the second whole-farm hotspot of the enterprise was excreta nitrous oxide. The breakdown of inputs revealed that the production of supplementary feed, which was only supplied to the agistment cattle, contributed to the increased emissions observed on both pastures. The agistment herd was predominantly grazed on annual pasture and perennial grasses, which is why the emissions were lower for tagasaste pasture. Furthermore, the actual contribution of the emissions from feed production to the EI of each pasture was higher for annual pasture (0.63 kg CO_2-e/kg LW) than subtropical grasses (0.30 kg CO_2-e/kg LW), reflective of the higher quantities of supplementary feed required to support the agistment stock on annual pasture.

The second hotspot for tagasaste was nitrous oxide emissions from excreta. This emission source contributed 0.98 kg CO_2-e/kg LW, or 6.66% to the EI of tagasaste. This higher contribution as compared to the other pastures, is reflective of both the higher average CP content of tagasaste and that it supported the largest proportion of the breeding herd through the production year. As the farm was in the process of expanding its breeding herd, there were higher numbers of growing and lactating cattle, both of which produced higher nitrous oxide emissions relative to mature stock as a result of higher intake, as described in detail in Chapter Four, Section 4.5.5. In terms of both area covered and livestock supported, tagasaste was the largest of the feedbases, explaining why this emission source was the second whole-farm hotspot despite accounting for a lower proportion of emissions than the other pasture types.

5.4.2.2 Analysis of animal emissions at Lancelin

As for the previous three enterprises, the carbon footprint analysis of Lancelin revealed that enteric methane production was the dominant emission hotspot. The combined contribution of animal emissions to the carbon footprint was 90.68%, the largest of all the farms. This value reflects the low input nature of Lancelin's operations and the role of animal emissions, specifically an expanding breeding herd, in

determining the overall GHG profile of the enterprise. The following Sections examine the emissions produced by the breeding and agistment herds at Lancelin, across the various stock classes and through the production year.

5.4.2.2.1 Breakdown of enteric methane emissions across stock classes

The dominant role of the breeding herd at Lancelin is evident in Figure 5.21. In total, stock classes within breeding herd contributed 76.04% of total enteric methane emissions. Breeding cows from the breeding herd are the largest producers of enteric methane across the enterprise, contributing 61.60% of total annual emissions. Of this value, replacement cows comprising first and second joined heifers, are responsible for 11% more enteric methane emissions than mature cows. As described in Chapter Four, Section 4.5.5, on a per kilogram of liveweight basis, the older heifers produced more methane than mature breeding cows, resultant from increased feed intake requirements. This meant that at Lancelin, which was in the process of building its breeding herd, the higher proportion of immature stock increased overall emissions on a liveweight basis more than an enterprise with a higher proportion of mature stock. Of the calves produced by the breeding herd, heifer calves produced 16% more enteric methane than bull calves, due in part to the slightly higher stock numbers, but also the longer weaning period (seven versus eight months) of heifer calves.

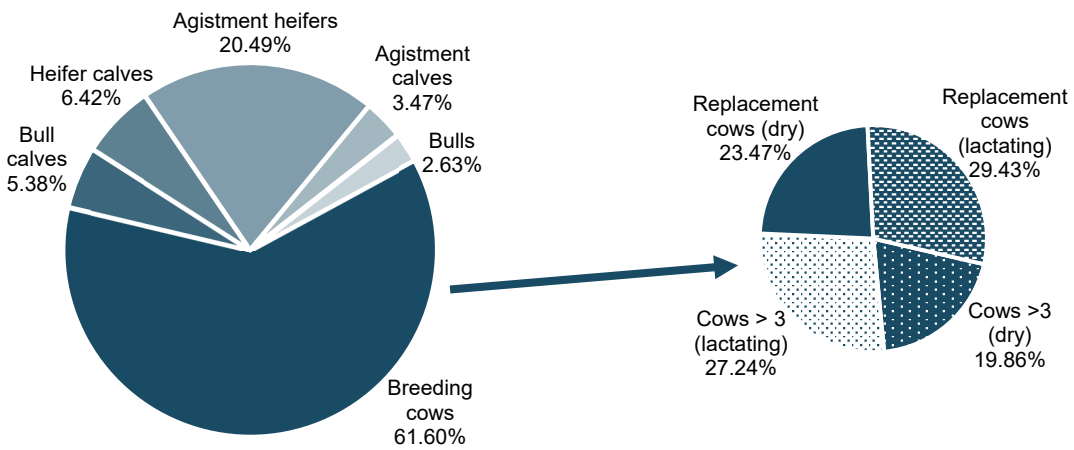


Figure 5.21 - Contribution of each stock class within the breeding and agistment herds to total annual enteric methane emissions (CO_2-e) produced at the Lancelin beef cattle enterprise

The agistment herd, on-farm for seven months of the production year, contributed the remaining 24% of enteric methane emissions. Agistment calves only contributed 3.47% of this, reflecting the fact that they only contributed enteric methane for three of these months. The results presented in Chapter Four, revealed that on a per animal basis, agistment heifers produced more methane than mature cows or heifers within

the breeding herd. The 20.49% of enterprise enteric methane emissions contributed by agistment heifers reflects these findings. Despite these increased emissions, the findings in Chapter Four also revealed that, per kilogram of saleable liveweight produced, the agistment herd was still more efficient than the breeding herd.

5.4.2.2.2 Distribution of animal emissions across the production year

The distribution of monthly animal emissions across the production year at Lancelin demonstrates its non-reliance on a traditional growing season (Figure 5.22). As described in Chapter Four and also earlier in this Chapter, tagasaste was the dominant feedbase of the enterprise and played an important role in supporting cattle at the end of the growing season, from September to December, when annual pasture was in decline. Its role in supporting cattle late in the non-growing season, from March to April, is also evident. By contrast, emissions from cattle on annual pasture were high from June to August, when the pasture quality was at its highest. Though animal emissions are also high in January and February, this is not a reflection of increased productivity, rather of the increased supplementary feed required to support the agistment herd on pasture (Chapter Four, Section 4.5.5). The role of subtropical grasses in supporting agistment cattle through the dry season, from November to April is evident, along with the minor role it played in supporting cattle during the growing season when subtropical grasses were dormant.

The monthly breakdown of stock classes presented in Figure 5.23 provides further explanation as to the effect of farm decisions on the emission hotspot, enteric methane production. In contrast to Dongara, the dominance of the breeding herd in the production of total monthly enteric emissions is evident. Breeding herd heifers were the primary contributors to monthly enteric emissions throughout the production year, reflective of the expanding nature of the breeding herd. Given this production of emissions and given that heifers do not begin to produce saleable liveweight until they are over 18 months at first calving, examining the emissions trade-off between a mature herd versus an expanding herd with a high proportion of heifers is a useful exercise.

As described earlier, the breeding herd was largely supported on tagasaste pasture and this is obvious in Figure 5.23. The July calving meant that feed demand by the breeding herd was highest at the end of the annual growing season through the beginning of the non-growing season, from October to November. Without the perennial pasture, equivalent production of saleable liveweight by the breeding herd would not have been possible without the provision of supplementary feed.

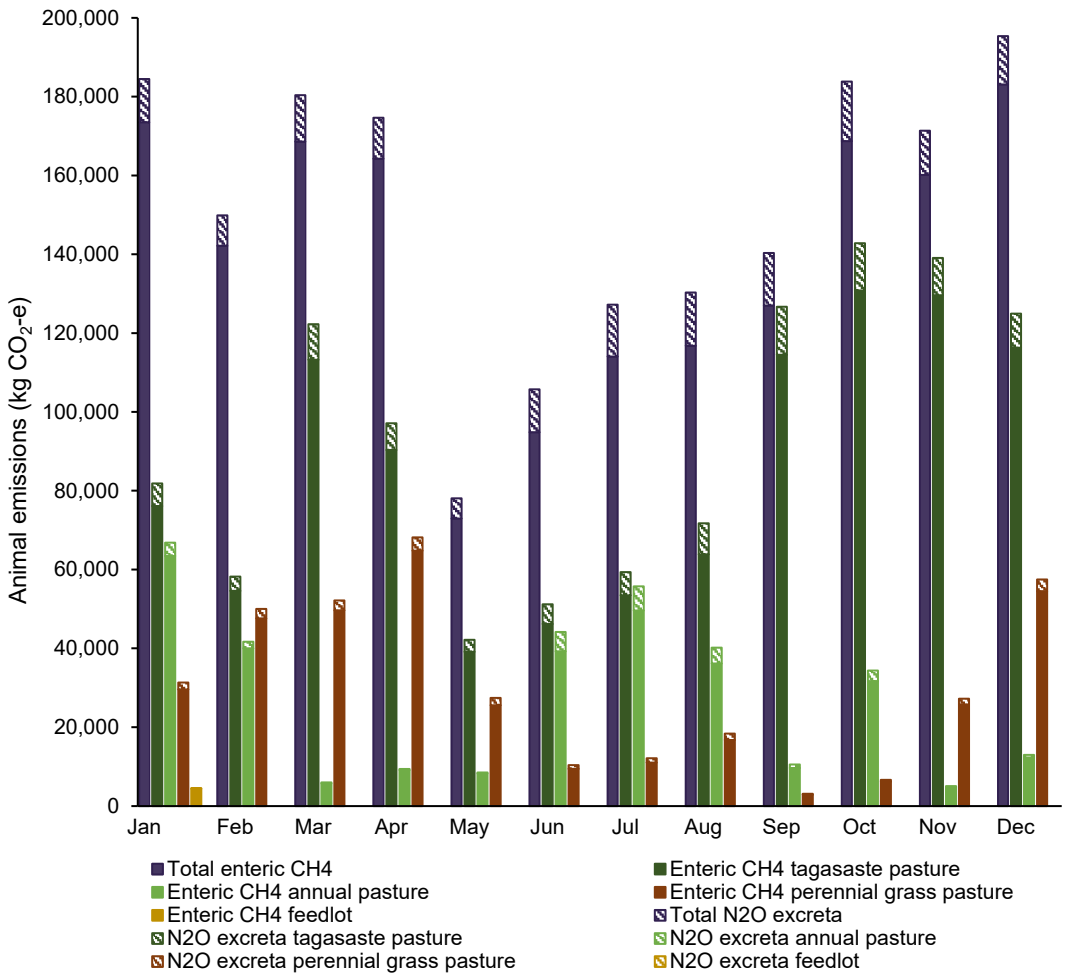


Figure 5.22 - Total monthly whole-farm and individual feedbase animal emissions (CO_2-e) at the Lancelin beef cattle enterprise

Where: solid bar is contribution through enteric methane production and striped bar is contribution through nitrous oxide emissions from dung and urine. Note: though methane from manure is included, its contribution is minor and not visible here.

Similarly, Lancelin was only able to take on agistment cattle during the non-growing season because of the availability of perennial pastures for grazing during this period. Figure 5.23 shows that enteric emissions from agistment stock were the highest for the perennial pastures, in particular subtropical grasses, reflective of their role in supporting liveweight production by the agistment herd. The increased contribution of the agistment herd emissions to overall monthly emissions is evident from January onwards, a result of the physiological status of the heifers (lactating) and the requirement to supplementary feed to meet the ensuing increased requirements. Chapter Six explores further the potential emissions burden the requirement for supplementary feeding places upon the production of saleable liveweight.

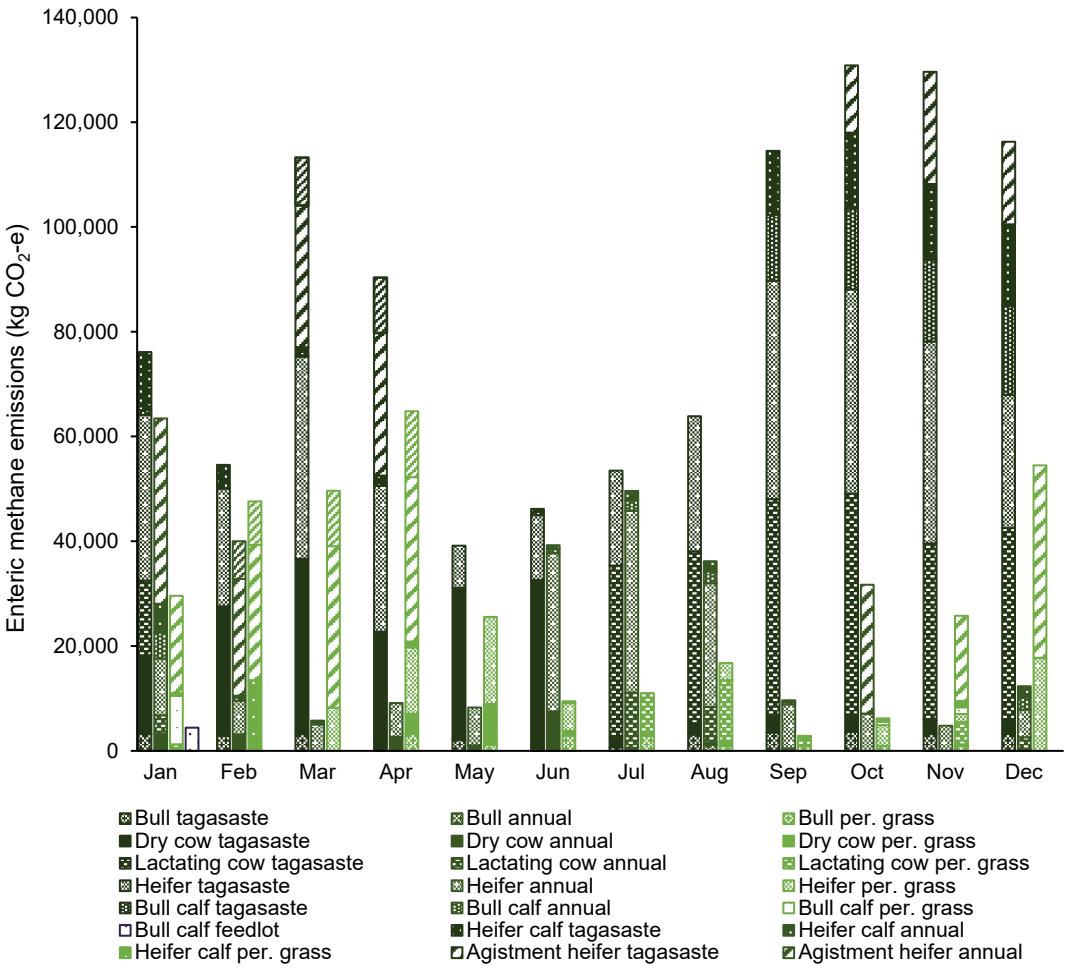


Figure 5.23 - Total monthly breakdown of enteric methane emissions (CO_2-e) within each stock class of the breeding and agistment herds on each feedbase at the Lancelin beef cattle enterprise

5.5 Comparisons with existing research

The earlier sections of this Chapter provided detailed breakdowns of the carbon footprints of the case study farms, through examination of whole-farm and feedbase EIs, hotspots and intra-annual variations. Such detailed analyses enabled identification of feedbase and flock/herd characteristics, along with management practices which influence the carbon footprint. Whilst comparisons within this study are possible, comparing these results to those of other LCA and carbon footprint studies is difficult and should be made with caution. As described in Chapter Two, observed differences between livestock systems examined in different studies are not always indicative of differences in productivity or reflective of different regions or adopted practices. Rather these differences can also be attributed to differences in adopted methodologies, assumptions and data sources. Lack of transparency regarding the data used often makes it difficult to make these distinctions. For these

reasons, this section has been delivered separately, not inter-dispersed within the carbon footprint results earlier in this Chapter. Despite the inability to directly compare results between studies (de Vries et al., 2015; Zervas & Tsiplakou, 2012), it is important to acknowledge these studies and to draw broader comparisons.

This Section, therefore, explores trends in the EIs and emission hotspots of Australian and international carbon footprint and LCA studies, along with relevant Australian non-carbon footprint whole-farm studies, making comparisons with the findings of the present study. Comparisons between beef and sheep production systems are also made, identifying characteristics or practices which have influenced the results of other studies. A number of these influencing factors are considered further in Chapter Seven.

5.5.1 Carbon footprints of ruminant livestock production

Globally, the carbon footprints of beef cattle production vary widely. Expressed as EI, estimates of the carbon footprint of beef production (kg CO_2-e/kg LW) fall within an extensive range[4], from 8.0 kg CO_2-e/kg LW (Nieto et al., 2018) to 26.3 kg CO_2-e/kg LW (Bragaglio et al., 2018), excluding impacts from LU and dLUC. In Australia, the distribution of EI ranges from 9.6 (Peters et al., 2010) to 14.0 kg CO_2-e/kg LW (Wiedemann et al., 2015c). Another whole-farm, but not carbon footprint, study calculated the EI of beef production to be as much as 25.8 kg CO_2-e/kg LW in the Kimberley region of northern Australia (Eady et al., 2016). However, this study did not consider pre-farm emissions or on-farm emissions associated with input use. The spread of values in both international and Australian studies reflect the diversity in production systems and locations examined, along with methodological approaches at both scales. The EIs obtained for both the Dongara and Lancelin beef production enterprises, 9.17 and 13.21 kg CO_2-e/kg LW respectively, fall within this range reported by other studies.

Far less research attention has been directed towards investigating the carbon footprint of sheep meat production. Between the studies which have considered meat production, EI values vary widely, largely a result of the requirement to allocate emissions between wool and meat production in most sheep systems. Though allocation methods differ, most international studies and earlier Australian studies have adopted economic allocation, which varies widely, both across countries and

[4] Global and Australian EI values for both beef and sheep meat production were sourced only from studies which had adopted the FU of kg LW or had provided sufficient information to allow conversion to this metric. Dairy studies, or those which had included an intensive feedlot stage external to the on-farm stage were also omitted, to align with the system boundaries of this study.

also temporally (LEAP, 2015b). Globally, the reported EIs of sheep meat production range from 8.6 to 143.5 kg CO_2-e/kg LW (Edwards-Jones et al., 2009; Ledgard et al., 2011). This latter value was calculated from a farm in the UK, with most of these emissions attributed to peat soil on the farmland. Excluding the unusually high EI values of that study, the next highest EI reported in sourced literature was 25.9 kg CO_2-e/kg LW (Ripoll-Bosch et al., 2013).

In Australia, most carbon footprint and whole-farm studies which have reported on sheep meat production, have done so as an allocated by-product of wool, with wool being the focal commodity (Alcock et al., 2015; Biswas et al., 2010; Brock et al., 2013; Doran-Browne et al., 2016; Eady et al., 2012; Henry et al., 2015b; Wiedemann et al., 2016c). A smaller number have focussed on sheep meat (Alcock & Hegarty, 2011b; Bell et al., 2012b; Peters et al., 2010; Wiedemann et al., 2015b; Wiedemann et al., 2016b) whilst others have examined combined sheep production system efficiency (Browne et al., 2011; Cottle et al., 2016; Doran-Browne et al., 2015; Harrison et al., 2014a; Harrison et al., 2014b). Together, the EI values of the reported carbon footprint studies range from 4.4 to 10.7 kg CO_2-e/kg LW (Peters et al., 2010; Wiedemann et al., 2015a). These Australian values fall below and in the lower end of the EI range of international studies, in line with other reviews of lamb carbon footprint studies (Clune et al., 2017). The calculated EIs of Bremer Bay and Wickepin fall within or close to the range of both the international and Australian studies, 8.18 and 10.60 kg CO_2-e/kg LW, respectively.

As described in Chapter Two, the large variations in EI of both beef and sheep production reflect different farm systems, scales of assessment (i.e. country to regional to case study farm) and methodological approaches. Amongst these, there are four methodological factors with potential to explain a portion of the large differences observed between international and Australian EIs, and also between the Australian EIs and those of the livestock systems examined in this study. These highlight the difficulty of comparing direct results, using the findings of the present study as a basis for comparison.

Firstly, with focus on sheep production, the economic allocation factors applied to meat and wool can differ considerably between countries. For example, the gap between wool price and sheep liveweight price is considerably larger in Australia than internationally, with wool of higher value. This results in a lower proportion of emissions allocated to meat production in Australia as opposed to internationally, with studies reporting this to differ by as much as 48% (Wiedemann et al., 2015a).

The second factor, again directed at sheep production, is the difference between economic allocation and, the more recently accepted and adopted, protein mass allocation. Wiedemann et al. (2015a) found that the difference in proportion of emissions allocated to meat using the two methods resulted in EI variations of up to 20%, while Cottle and Cowie (2016) found this EI difference to be, on average, 24%. In all Australian instances, the transition from economic to protein mass allocation resulted in a shift of emissions burden so that EI was greater for meat than wool production. This was also observed in the present study, with protein mass allocation to meat production of Wickepin case study farm emissions totalling 62% as opposed to 38% directed to wool. Most of the reviewed Australian studies which reported lower EIs had also employed economic allocation; including Peters et al. (2010), Eady et al. (2012) and Biswas et al. (2010)(4.4-4.7, 5.3 and 5.1-5.6 kg CO_2-e/kg LW, respectively). By contrast, a number of studies (including the present study) which adopted protein mass allocation observed higher EI values; for example, Wiedemann and Yan (2014) and Wiedemann et al. (2016b) calculated EI values of 6.2-7.9, 6.5-7.0 kg CO_2-e/kg LW, respectively. Similarly, Cottle and Cowie (2016), which presented their EI in terms of carcass weight, not liveweight, found that a transition from economic allocation to protein mass allocation increased EI by 75%. Consideration of allocation method means that while EIs may be, on average, lower in Australia than internationally, and lower for economic allocation than protein mass allocation within Australia, it is not a reflection on the relative emissions efficiency of the systems. Rather it just represents a shift of burdens across an enterprise. Potentially, the emissions efficiencies of respective sheep production systems may not differ as much as certain allocation factors would indicate.

A third factor proposed to explain a portion of the differences observed between international and Australian beef and sheep studies is the choice of methodology to calculate enteric methane emissions. As described in Chapter Two, most international studies adopt the IPCC Tier 2 methodology, which calculates enteric methane as a proportion of gross energy intake using a conversion factor which is dependent solely on the DMD focus of diet (concentrate versus roughage). Others apply Tier 1 methodology, which involves the application of a flat enteric CH4 rate to livestock, regardless of dietary DMD or animal physiological or physical state. For example, Cerri et al. (2016) a study examining extensive pasture-based beef production in Brazil, applied Tier 1 country-specific rates of 0.19 kg/CH_4/day (68 kg CH_4/head/yr) for cows and 0.13 kg CH_4/head/day (48 kg CH_4/head/yr) for young cattle less than 230 kg LW. Another Brazilian study examining similar systems, however, applied a

rate of 0.14 kg/CH_4/day (52 kg/CH_4/yr) across stock classes (de Figueiredo et al., 2017). These can be compared to the values obtained in the present study which, following the approach of Blaxter and Clapperton (1965), considered animal characteristics, dietary DMD and DMA. Averaging calculated monthly methane production to obtain values of daily methane across the production year, dry and lactating cows at the Dongara case study farm produced 0.18 and 0.22 kg/CH_4/day (66 and 80 kg/CH_4/yr) respectively. The average values at the Lancelin case study farm were 0.17 and 0.20 kg/CH_4/day (61 and 72 kg/CH_4/yr), respectively. Young cattle with liveweights less than 230 kg at Dongara and Lancelin generated an average of 0.10 kg/CH_4/day at each farm. Similarly a UK study, Edwards-Jones et al. (2009), allocated Tier 1 IPCC defaults of 0.02 kg/CH_4/day (8 kg/CH_4/yr) to pasture-based adult sheep. By comparison adult ewes and rams on pasture at both Bremer Bay and Wickepin produced an average of 0.03-0.04 (11-15 kg/CH_4/yr; dry versus lactating) and 0.04 kg/CH_4/day (14 kg/CH_4/yr), respectively. These values highlight differences between the IPCC Tier 1 approach and the methodology employed in the present study. Such differences are amplified when applied at a farm-scale, and the consequences of not considering influencing factors such as intra-annual variations, dietary DMD, DMA, increased production from lactating and growing animals become evident. The differences presented in these examples align with comparisons made in other studies. For example, Dougherty et al. (2018) found applying Tier 1, Tier 2 and country-specific methodology altered the final EI of sheep production in California by up to 14%. In a comparison of Australian-specific approaches, Brock et al. (2013) found that enteric methane predictions were 26% less when NGGI methodology and seasonal default values were applied, as opposed to GrassGro output in a daily time-step. The above demonstrate the importance of examining the methodology adopted by a study given the influence it can have on the carbon footprint of a livestock production system.

The final factor to be mindful of when observing the differences in the EIs, and thus carbon footprints, of different studies, is the selection of GWP. The release of each IPCC report has been accompanied by revised weightings for methane and nitrous oxide emissions. The most recent recommendations in AR5 place greater weight on methane and less on nitrous oxide than that of AR4. Given that enteric methane emission is the dominant hotspot of ruminant production systems, this can have profound effects on EI. For example, if in the present study, Bremer Bay was to adopt the AR4 GWPs, the whole-farm EI would decrease by 6%, from 8.1 to 7.6 kg CO_2-e/kg LW. Similarly, the EI of Lancelin decreased by 9% (13.2 to 12.0 kg CO_2-e/kg LW)

under the historical GWPs. The great majority of the most heavily cited livestock carbon footprint studies adopted the AR4 values, however as many do not provide detailed breakdown of emissions, it can be difficult to adjust the results accordingly. However, for those which do, the results can differ considerable. For example, Beauchemin et al. (2010) estimated the EI of beef production in Western Canada to be 13.0 kg CO_2-e/kg LW, similar to the EI of the Lancelin beef production enterprise. However, once the results were recalculated in accordance with current GWPs, the EI was reduced by 12% to 11.5 kg CO_2-e/kg LW. As recently as 2018, studies were publishing results with AR4 GWPs (Bragaglio et al., 2018; Nieto et al., 2018), highlighting the additional limitations of comparing the EIs of studies.

Despite the difficulties in distinguishing whether EI differences between studies result from methodological or production system factors, this limitation is eliminated when comparing EIs within the same study (Ledgard Sf, 2011). In the present study, the EIs of the four case study farms varied by up to 38%. Other studies have calculated the EI of multiple case study farms within regions and found similar variations. For example Wiedemann et al. (2015c) found the EI of 11 case study beef enterprises in eastern Australia varied by 26%, while Ridoutt et al. (2011) found differences of up to 20% between six beef enterprises in NSW, Australia. Internationally, Veysset et al. (2014) found EI varied by 52% amongst 59 beef enterprises located in central France, Jones et al. (2014) calculated variations of up to 75% between 60 sheep case study farms in the UK, while in central Argentina Nieto et al. (2018) the EI of 30 beef enterprise varied by 69%. Opio et al. (2013) conducted a global-scale study comparing the EI of ruminant production systems across world regions, finding variations of up to 82% for beef production and 52% for sheep meat production. Such intra-study comparisons highlight the scope for improvement which exists between production systems. It also enables identification of farm system characteristics or practices which have influenced these variations, which is what several the above studies, along with others, have undertaken. This means that whilst the direct results of the various studies cannot be compared to the present study (i.e. EI), referring to the carbon footprint influencing factors identified by these studies is useful when examining differences between the four enterprises. It also assists in the analysis of the effect of potential mitigation strategies.

5.5.2 Emission hotspots

In line with the four livestock production systems examined in the present study, enteric methane was the dominant emission hotspot of all the beef and sheep production studies examined. Ranging from 74-75% for the two sheep production

enterprises, Bremer Bay and Wickepin, and from 78-85% for the two beef cattle enterprises, Dongara and Lancelin, respectively, the high contribution of enteric methane to each farm's carbon footprint aligns with the findings of other extensive pasture-based systems. For example, Wiedemann et al. (2016b) found that the contribution of enteric methane to the EI of predominantly pasture-based lamb produced for export in southern Australia was between 83 and 89%. Similar results (79-89%, 86%) were obtained for merino enterprises located through southern Australia (Brock et al., 2013; Wiedemann et al., 2016c). None of these studies produced supplementary feed on-farm, so removing emissions from crop residue from the two sheep systems, enteric methane comprises 81% for Bremer Bay and 77% for Wickepin, in line with these studies. This proportion would be even higher if associated input emissions, such as fertiliser and lime, were also excluded.

Two Australian beef carbon footprint studies found that enteric methane contributed between 83 and 92% of the GHG profile of pasture-produced cattle in eastern Australia (Ridoutt et al., 2011; Wiedemann et al., 2015c), while Taylor and Eckard (2016) found this to be from 90-92% in northern Australia. Another study conducted on two case study farms in eastern Australia (Eady et al., 2011) calculated enteric methane contributions of 74% and 85%. The lower value was attributed to the high proportion of pre-farm emissions generated on that case study farm, a result of including emissions associated with purchased weaners prior to entering the farm. Similarly, the lower enteric methane contribution by Dongara (78%) can be explained by high pre-farm emissions (10%). This was almost double that of the other three enterprises and a result of the high emissions generated in the production of fertilisers applied through to pasture.

The high contribution of enteric methane to the EI of the four enterprises also aligned with the results of international studies that have focussed on grass-based sheep and beef production. In South America, where livestock is predominantly produced and finished on pasture with few inputs (Becoña et al., 2014; Cerri et al., 2016), enteric methane has been found to contribute between 85 and 98% to the GHG profile of beef production (Dick et al., 2015a; Ruviaro et al., 2015) and an average of 85% to sheep production (Toro-Mujica et al., 2017), as examples. In Thailand, where cattle are typically produced in low-input smallholder systems, enteric methane has been calculated to contribute 77% (Ogino et al., 2016). The carbon footprint of grass-produced sheep in Ireland (O'Brien et al., 2016) and California, United States (Dougherty et al., 2018) comprised 80-87% and 72% of enteric methane, respectively. Despite differences in system characteristics and approaches, the common factors in

all the considered Australian and international livestock production systems, along with the farms examined in this study, are the extensive, pasture-based and low input nature of production.

The high proportion of enteric emissions observed in this study, however, differ to the results of more intensive systems where, though still the primary hotspot, the contribution is smaller. Two other Irish studies, Casey and Holden (2006) and (Foley et al., 2011) found that enteric methane comprised only 49-63% of the EI of beef production in semi-intensive systems. Similar results have been found in both beef and sheep systems through Europe, where livestock are housed for a portion of the year and fed quality grain and forage, with weaners often finished in feedlots (Jones et al., 2014; Mogensen et al., 2015; Ripoll-Bosch et al., 2013; Veysset et al., 2010; Veysset et al., 2014). Canada (Beauchemin et al., 2010; Vergé et al., 2008), the United States (Capper, 2011; Pelletier et al., 2010; Stanley et al., 2018) and Japan, which also typically employ intensive systems, found that enteric methane contributed only 55-63%, 42-53% and 61%, respectively.

The differences in the magnitude of the contribution of this primary hotspot between extensive systems, such as the four livestock production systems examined in the present study, and more intensive systems, are often attributed to production efficiency gains. As described in Chapter Two, these include higher quality feed, faster growth rates and ensuing turnoff rates for slaughter or sale. However, it is also a result of the higher contribution of other hotspots in more intensive systems, namely manure emissions and, to a lesser extent, emissions arising from the production of inputs. In the present study, nitrous oxide emissions from excreta averaged approximately 6% for all the livestock production systems examined. Similar results have been obtained in other Australian studies, including 5-10% for sheep and 7-13% for beef production in eastern Australian (Brock et al., 2013; Eady et al., 2012; Wiedemann et al., 2015c).

By comparison, this emissions from excreta and manure is far more dominant in the international semi-intensive production systems; that is, systems with a pasture component but also a confinement or feedlot component. Examples of such systems in Europe observed nitrous oxide emissions from manure and excreta of between 15 and 30% for sheep production and 13-26% (Bragaglio et al., 2018; Hyland et al., 2016; Jones et al., 2014; Mogensen et al., 2015; Ripoll-Bosch et al., 2013; Veysset et al., 2010; Veysset et al., 2014). Similar results have been obtained in Canada (23%) and the United States (19-26%)(Beauchemin et al., 2010; Pelletier et al., 2010; Stanley et

al., 2018)[5]. The higher proportion of manure emissions can be partly attributed to the requirement for manure handling and storage in confinement or feedlot systems, common stages in intensive production systems but not required in pasture systems. It can also be attributed to the higher emissions generated on pasture in the regions central to these more intensive systems. In Australia, denitrification rates of nitrogen deposited on pasture by livestock in excreta have been found to be much lower than other regions, as reflected in the lower EF of 0.4% as compared to the IPCC assigned EF of 1%, or higher, adopted by most of the studies examined (DEE, 2019). In addition to this, emissions resulting in leaching and runoff is rare in Australia as compared to other regions such as Europe, South America and New Zealand. These differences help explain why higher rainfall regions such as New Zealand and countries within Central and Southern America, which have extensive pasture-based production systems similar to Australia, exhibit high contributions from manure emissions as would be expected from a more intensive system (Becoña et al., 2014; Bogaerts et al., 2017; Cerri et al., 2016; de Figueiredo et al., 2017; Huerta et al., 2016).

A third common hotspot identified by livestock carbon footprint studies is the production of inputs. As described earlier, this emissions source was relatively small for each of the considered livestock production systems considered in the present study, totalling 10% for Dongara and between 5 and 6% for Bremer Bay, Wickepin and Lancelin. This reflects the findings of other pasture-based, low input Australian production systems with Eady et al. (2011) reporting between 2-5% and Brock et al. (2013) reporting 2% on eastern cattle and merino enterprises, respectively. By contrast, in more intensive systems, the production of inputs can contribute similar, or even higher, emissions to that of manure. For example, Hyland et al. (2016) found that inputs were responsible for 24% and 22% of the EI of the sheep and beef enterprises examined in their UK study. Another UK study centred on sheep production found this hotspot to total 20% (Jones et al., 2014), while in the United States this totalled between 22-28% for beef and sheep production (Dougherty et al., 2018; Pelletier et al., 2010), in Thailand approximately 15% (Ogino et al., 2016) and for beef production in France, 20% (Veysset et al., 2014). In all cases where emissions from the production of inputs in the semi-intensive systems is high, the two key contributors are feed production and fertiliser production. Though on a smaller scale, the same was true for the present study with the production of plant chemicals comprising between 37-89% of this GHG emission source. Feed production was more

[5] While most carbon footprint studies present the contribution of enteric methane as a primary hotspot, few present results with a breakdown of contribution by each emission source. As such presentation of other hotspots, such as manure emissions, from reviewed studies is limited.

variable, Dongara and Wickepin did not purchase feed as a result of effective grazing management between pasture types and production of all supplementary feed from on-farm purpose grown crops, respectively. For Lancelin, which purchased supplementary feed for agistment stock, and Bremer Bay which purchased feed to supplement the feed grown, feed production comprised 37% and 24% of total emissions arising from input production, respectively.

Despite the seemingly small contribution of the production of inputs to the carbon footprint of each of the case study farms, in line with other studies centred one low-input extensive livestock production systems, the added emissions impact of growing crops for supplementary feed must be considered. Though the contribution of feed production was low or non-existent for Bremer Bay and Wickepin, the emissions associated with the production of the on-farm supplementary feed crops (excluding those related to animal emissions) were approximately 13% and 8%, respectively. Adding to these values, the contribution of all other inputs, this becomes 16% and 10%. As a low-input production system, these are high contributions. The emissions trade-off between producing supplementary feed on-farm and purchasing in feed (as many of the reviewed studies do), is worth exploring further.

5.5.3 Carbon footprint of sheep versus cattle production

It is widely accepted and demonstrated that ruminant livestock systems have greater environmental impacts than monogastric production systems, in terms of their carbon footprint (Steinfeld et al., 2006). Whether sheep or beef production systems are more emissions efficient remains under contention. In terms of gross emissions, cattle enterprises tend to have a larger footprint than sheep enterprises, a result of the higher emission output per animal unit of cattle as opposed to sheep. This is reflected in the present study where in all instances the unadjusted EQ was higher for the cattle enterprises than the sheep enterprises. Comparing the Lancelin and Wickepin enterprises for example, the EQ of Lancelin was 42% higher despite running approximately 1,000 head of cattle over the production year compared to the more than 5,500 head of sheep at Wickepin. However, in terms of emissions efficiency, this study found that there was no obvious correlation between enterprise type and carbon footprint. For example, excluding emissions from pasture residue, the farm with the lowest EI was the Bremer Bay Dorper enterprise, followed by the Dongara Angus beef and backgrounding enterprise. The least efficient farm was the crossbred cattle and agistment property, Lancelin, followed by the Wickepin Merino and SAMM enterprise. The EI of the two sheep enterprises differed by 23% while the beef enterprises differed by 30%. With no obvious difference between the sheep and beef enterprises, it

became obvious in the analysis conducted earlier in the chapter that the differences were rather a result of different characteristics and employed practices.

Other studies have also compared the carbon footprints of sheep and beef production systems. Ledgard et al. (2011) and Lieffering et al. (2011) each using the same methodological approach, conducted benchmarking analyses of New Zealand lamb and beef produced for export. They found that the EI of sheep production was 18% lower than that of beef production, attributing this difference to the greater fecundity of ewes, higher growth rate of lambs and the distribution of sheep emissions between both wool and meat, as opposed to just meat on cattle enterprises. Peters et al. (2010) compared the carbon footprint of a Merino supply chain in Western Australia to two beef supply chains in eastern Australia across two observed production years. They found that across both years the EI of the sheep enterprise was lower, from 15% to 37%, which they attributed to the shorter turnoff rates of the sheep as opposed to the cattle. However, the sheep enterprise also purchased in stock in the second year, which has likely contributed to the reduced EI and thus the higher observed gap between enterprise types. Not all studies have found sheep production to have lower carbon footprints, however, with a Canadian benchmarking study calculating the EI of national sheep production to be 17% higher than that of beef production (Dyer et al., 2014). The study was restricted in its analysis however, as the tool it employed to calculate emissions did not permit the user to examine potentially influencing factors (i.e. liveweights, slaughter weights, growth rates, intake breakdown) in order to compare the source of the observed differences between enterprise types. However, the study did highlight the dominance of beef production in the Canada's agricultural industry and the minor role of the sheep industry. Given its dominant role, the Canadian beef industry has observed significant efficiency gains, in terms of both productivity and emissions, in recent decades. A portion of beef slaughtered is also sourced from the Canadian dairy industry, estimated to reduce the EI of beef production by up to 11% (Legesse et al., 2016). Together, this could explain the lower carbon footprint of Canadian beef over sheep production.

Though each of the above studies identified potential factors to explain the observed differences between the carbon footprint of beef and sheep enterprises, benchmarking analyses based on national averages and comparisons of single case study farms don't account for the variation which may exist between production systems of the same enterprise type. Hyland et al. (2016) compared the carbon footprint of 15 Welsh case study farms which specialised in either sheep or beef production. Comparing the individual farm footprints between the two enterprise types

they found no observable trend, with some of the sheep case study farms presenting lower EIs than some beef farms. while in other instances the beef farms presented the lower EI. However, when they grouped low EI and high EI farms across each enterprise type, they found that in both instances sheep production presented a lower carbon footprint. It was only once farms were grouped according to the management practices they did (or did not) employ than a trend was identified between enterprise types. The findings of Hyland et al. (2016) show that the relative EI performance of a sheep versus beef farm is not determined by enterprise type, but rather the characteristics and management practices adopted by each. This aligns with the findings of the present study. For example, the Dongara beef enterprise exhibited a lower EI than the Wickepin sheep enterprise. As observed earlier in this Chapter and in Chapter Four, two reasons for this are the bringing in of backgrounding cattle at Dongara which diluted breeding herd emissions and also the strong grazing management strategies employed, which ensured pasture supply matched stock demand through the production year, maximising growth rates and optimising emission output. The influence of various practices adopted by the four livestock production systems on the respective carbon footprints is examined in the following Chapter Six.

5.6 Conclusion

The findings of the carbon footprint analyses undertaken in this study have highlighted the impact of geographical location, pasture type, livestock characteristics and management practices on the GHG profile of a livestock production system. The depth of analysis conducted was only possible because of the Frameworks developed in Chapter Three. These Frameworks enabled the carbon footprint to be examined at a whole-farm scale, averaged over the production year, with a detailed breakdown of each GHG emission source. The intra-farm analyses conducted at monthly timesteps enabled examination of the effect of pasture, livestock fluctuations, production events and management strategies on the emissions efficiency of each feedbase, and the resultant impact on the whole-farm carbon footprint.

This Chapter has highlighted the carbon footprint variation that exists not only between farms, but also within farms, across feedbases. Such intra-farm variation is not captured in carbon footprint analyses which apply regional data on an annual, or even seasonal, basis. Similarly, intra-farm variation cannot be fully captured without considering intra-annual fluctuations and detailed feedbase and livestock information. Whilst such detail may not be necessary for benchmarking analyses for example, it is

necessary when examining proposed mitigation strategies. With such inherent variation between and within farms, the suitability and impact of adaptation or mitigation strategies will differ. Conducting analyses at the scale and level of detail as presented in this study enables a targeted and farm-specific approach.

The following two Chapters explore further the impact of pasture systems, livestock management and grazing management practices, as identified in this Chapter, in determining the carbon footprints of livestock production systems.

6 THE MITIGATION POTENTIAL OF PERENNIAL PASTURE SYSTEMS

6.1 Introduction

Chapter Five presented the findings of the carbon footprint analyses for the four considered livestock production systems, including those of the intra-farm analyses which considered, amongst other factors, different pasture systems. Through these analyses it became evident that pasture type and its management play a role in determining both the productivity and carbon footprint of a livestock enterprise. This Chapter explores these findings further by conducting scenario analyses to examine in detail the impact of respective pasture systems at each enterprise and the potential for perennials to mitigate whole-farm emissions.

6.2 Influence of pasture system

Chapter Five identified that perennial pasture played a role in the productivity and carbon footprint of the investigated farming systems. While emissions tended to be higher on the perennials, driven by higher annual livestock intake and increased carrying capacity, the increased liveweight production enabled by out-of-season green feed offset these increased emissions. Exceptions included where the perennial pasture played a secondary role in the grazing management of the enterprise, such as at Wickepin, or where the increased livestock carrying capacity afforded by the perennial pasture was utilised for other purposes, such as supporting an expanding breeding herd at Lancelin; driving pasture consumption towards the maintenance of livestock rather than the production of saleable liveweight, for example.

The analyses demonstrated that the actual contribution of perennials differed according to the respective management strategies and establishment motivators of each case study farm. This Section explores the influence of the perennial systems on each farming system in the context of; emissions contribution compared to annual pasture, influence on the whole-farm carbon footprint and their role in enabling certain productivity-enhancing practices. To support this, scenario analyses were conducted whereby the carbon footprint of each enterprise was modelled with an annual pasture system only. Measures of productivity, such as supplementary feed requirements and stocking rates provide additional context.

6.3 Bremer Bay sheep production enterprise

Chapter Five showed that emissions arising from kikuyu pasture contributed to over half of Bremer Bay's whole-farm emissions, whilst annual pasture contributed approximately half of the remaining emissions, followed by crop stubble and the feedlot. Despite this, the carbon footprint, expressed as EI, of the kikuyu pasture was 8.28 kg CO_2-e/kg LW, only 2.2% higher than annual pasture (8.10 kg CO_2-e/kg LW; Chapter Five, Figure 5.4). This reflected the higher productivity of kikuyu which supported the production of 52% of total saleable liveweight production annually, as compared to annual pasture and crop stubble which, combined, supported only 30%.

The EI of annual pasture is misleading, however, as it only reflects grazing only over the growing season when the pasture was of higher quality and availability. It does not consider mature stock moved to graze stubble at the end of the growing season or weaned lambs and immature breeding flock moved to the higher quality kikuyu pasture during the same period. If mature livestock were instead retained on annual pasture, which was of similar quality to crop stubble over the same period, an increase in EI would be observed. This is because annual pasture, similar to crop stubble, would only be able to support the maintenance requirements of the stock in question, thus producing emissions with minimal or no concurrent production of saleable liveweight. Similarly, to support the growth of immature animals and weaned lambs destined for sale on annual pasture instead of kikuyu pasture, large quantities of supplementary feed would be required.

Kikuyu played an important role in livestock production at Bremer Bay. It supported livestock year-round, along with growing animals within the breeding flock and weaned lambs moved from annual pasture at the end of its respective growing season. The provision of out-of-season green feed by the kikuyu pasture clearly assisted in the farm's accelerated joining system, enabling weaned lambs produced in the second joining cycle access to kikuyu pasture in December and January when the annual pasture had senesced.

6.3.1 Modelling an annual only pasture system

To further examine the role of perennial pasture at Bremer Bay, a scenario was modelled whereby only annual pasture was established. Similar approaches have been adopted by other studies. For example, Taylor et al. (2016) examined the influence of the perennial forage shrub, Leucaena (*Leucaena leucocephala)*, on the carbon footprint of beef production, and modelled a scenario whereby only pre-existing native vegetation was present. Harrison et al. (2014a) modelled the effect of

changing the baseline of a perennial ryegrass and sub clover pasture system to annual pasture only scenario, on the EI of sheep production.

The assumptions of the “no perennial pasture” scenario are presented in Table 6.1. This scenario was modelled to produce the same quantity of annual saleable liveweight as the baseline in line with the requirement to at least maintain productivity. Supplementary feed accommodated any increased feed intake deficits during the non-growing season and thus acted as a measure of productivity, alongside the modelled changes to the carbon footprint. Throughout the remainder of this Chapter and Chapter Seven, “baseline” refers to the initial case study farm conditions as presented in Chapters Four and Five.

Table 6.1 - Key assumptions of the “no perennial” scenario at Bremer Bay

Baseline	**“No perennial” scenario**
Annual pasture component was 170 ha	Annual pasture component was 370 ha
Kikuyu pasture component was 240 ha	No kikuyu pasture
Livestock moved from the annual pasture to crop stubble (mature livestock) or kikuyu (weaner lambs and growing animals) over dry season.	Mature livestock moved to crop stubble over dry season to maintain stubble stocking rates per baseline. Remainder of Dorper flock retained on annual pasture.
On-farm lupin production (40 ha) for feedlot ration	Per baseline
On-farm oat production (90 ha) for supplementary feed	Increased on-farm oat production (130 ha) to account for almost 40% increase in supp. feed requirements (Table 6.2). Additional 40 ha was sourced from land previously allocated to kikuyu
	Same saleable liveweight production as baseline

Scenario findings

The modelled “no perennial” scenario produced a carbon footprint, expressed as EI, of 8.58 kg CO_2-e/kg LW produced for sale (Figure 6.1; Table 6.2), almost 5% higher than the baseline with kikuyu pasture. The increased carbon footprint was accompanied by a 48.4% increase in annual supplementary feed required by livestock grazing annual pasture and crop stubble. This represents the additional feed required by the Dorper flock to overcome additional feed deficits encountered in this scenario and to maintain the same saleable liveweight production as the Bremer Bay baseline. Unsurprisingly, the increased emissions associated with this increase in supplementary feed requirements, accounted for 73% of the exhibited increase in annual whole-farm emissions (Appendix F). These emissions were attributed to activities associated with increased oat crop production, such as crop residue, the production and transportation of inputs, along with increased farm machinery operation. Interestingly, nitrous oxide emissions from excreta also increased by 6.5%

from the baseline, accounting for 8% of increased whole-farm emissions. Examining the attributes of the feedbases grazed as presented in Chapter Four, it was possible to ascertain that this was because the CP content of annual pasture over the non-growing period was higher than both crop stubble and kikuyu; a result of the high legume content in the annual sward.

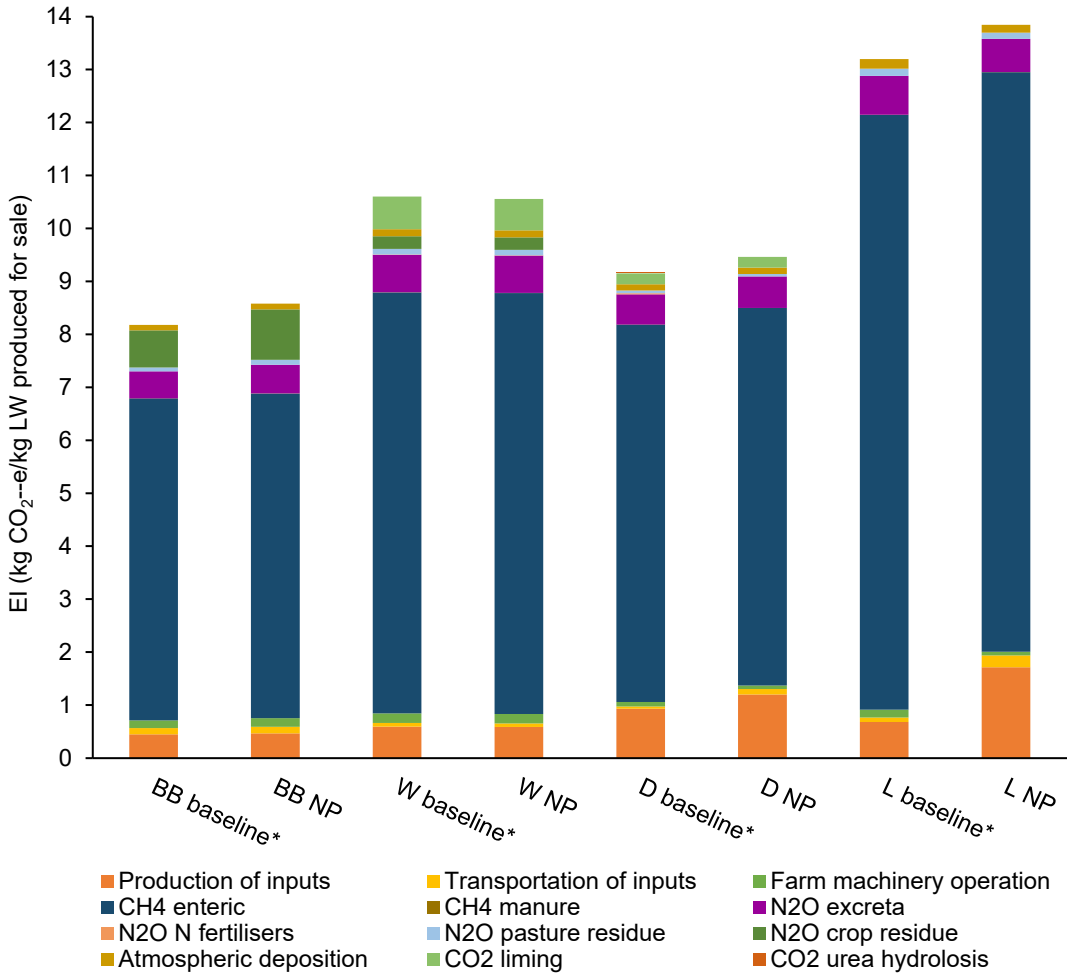


Figure 6.1 - The baseline carbon footprints, expressed as EI, of the Bremer Bay, Wickepin, Dongara and Lancelin livestock production enterprises, as compared to the respective "no perennial" scenarios.

Where: the Wickepin sheep production enterprise carbon footprint results are presented post allocation of emissions between meat and wool production. NP = no perennial. [6]

The changes to the EI of the individual feedbases as presented in Table 6.1, revealed that although annual pasture produced over 77% of total saleable liveweight in the scenario, almost three-fold more than the baseline, the revised EI was 8.24 kg CO_2-e/kg LW. This annual pasture EI was almost 2% higher than calculated in the baseline highlighting that the increase in liveweight produced on annual pasture was

[6] In each of the figures presented throughout this Chapter; "BB" refers to Bremer Bay, "W" is Wickepin, "D" is Dongara and "L" is Lancelin.

inadequate to offset the inefficiencies of carrying livestock on annual pasture through the non-growing season.

Table 6.2 - Influence of the "no perennial" pasture scenario on total liveweight (LW) produced, supplementary feed requirement, total emissions (EQ) and emissions intensity (EI) of the Bremer Bay sheep production enterprise, both at the whole-farm and individual feedbase scale

Scenario	LW produced for sale (kg)	Supplementary feed Pasture/crop stubble (kg)	Supplementary feed Feedlot ration (kg)	EQ (kg CO_2-e/ yr)	EI[a] (kg CO_2-e/ kg LW)
Bremer Bay baseline with perennial	**57,975**	**81,634**	**40,738**	**474,185**	**8.18**
Annual pasture	15,131	22,187	-	122,522	8.10
Kikuyu pasture	29,930	43,605	-	247,832	8.28
Crop stubble	2,419	15,841	-	82,341	34.04
Feedlot	10,500	-	40,738	21,490	2.05
Bremer Bay "no perennial"	**57,975**	**121,159**	**40,738**	**497,443**	**8.58**
Difference	***-***	***48.4%***	***-***	***4.9%***	***4.9%***
Annual pasture	44,676	100,283	-	367,952	8.24
Difference	*195.3%*	*352.0%*	*-*	*200.3%*	*1.7%*
Crop stubble	2,802	20,877	-	108,000	38.53
Difference	*15.8%*	*31.8%*	*-*	*31.2%*	*13.2%*
Feedlot	10,500	-	40,738	24,490	2.05
Difference	*0.0%*	*-*	*0.0%*	*0.0%*	*0.0%*

[a] Where whole-farm EI represents whole-farm emissions produced per kg of liveweight produced for sale across the case study farm, while the EI of the individual feedbases represents the emissions produced on that feedbase per kg of liveweight produced for sale on that feedbase.

Another indicator of the feasibility of an annual-only pasture system, aside from emissions and supplementary feed requirements, was threshold groundcover. In line with soil erosion considerations, scenarios were only considered if they did not increase the proportion of the year with groundcover below threshold levels beyond 7% (Moore, 2012). In the "no perennial" scenario, preliminary analyses in GrassGro determined that annual pasture could withstand stocking rates of up to 11.5 DSE/ha, almost double the modelled rates in the scenario. From a land degradation perspective, the annual pasture could thereby withstand higher stock numbers. However, increased numbers on annual pasture would be accompanied by significant increases in supplementary feed during the non-growing period and further increases to corresponding emissions. In addition, higher stocking rates over prolonged periods may affect other pasture parameters, such as the ability of the pasture to set seed and re-establish in the following season in the longer term.

This study did not include quantitative considerations of the economic implications of various farm strategies. However, an almost doubling of supplementary feed requirements in the modelled scenario would likely encounter both economic and feasibility issues. Other studies have found that a mismatch in pasture supply versus demand by livestock causes production stock to be retained for a longer period of time prior to achieving target sale liveweights due to slower growth rates (Alcock et al., 2015; Alcock & Hegarty, 2011b; Harrison et al., 2014a). Retaining stock longer had emission ramifications, increasing EI. Over the non-growing season, when stocking rates are close to threshold levels, feed available to the breeding flock was reduced and soil erosion risk increased, hindering productivity. Such a situation would be pertinent to Bremer Bay, whereby lambs are produced in the September lambing and remain on pasture until February. If supplementary feed was not increased to meet the increased feed deficit on the senesced annual pasture, lambs would either have to be sold at a lower liveweight, or retained on-farm longer in line with the lower growth rates on the poorer quality pasture. Either scenario would increase whole-farm EI, along with the EI of the annual pasture. As the annual pasture was well below threshold stocking rates, the ramifications on feed supply for the breeding flock or soil erosion would be unlikely, however from both a productivity and emissions perspective, this questions the viability of an accelerated joining system at Bremer Bay without the presence of a perennial pasture component.

6.4 Wickepin sheep production enterprise

The primary purpose of the saltbush at Wickepin was to remediate marginal salt-affected land. Following establishment, it had since become a source of green feed over the non-growing season, from November to May, for a small portion of the breeding flock's mature ewes. This secondary role of saltbush was reflected in Chapter Five, finding that it contributed only 4% of whole-farm emissions, far less than the annual pasture, which contributed 72%, and the remainder from the crop stubble to which the breeding flock (aside from the ewes on saltbush) were moved to over the non-growing season. Grazed solely by mature ewes, the saltbush only supported a small proportion of annual saleable liveweight, less than 3%. The lesser role of the saltbush in the grazing system at Wickepin was reflected in its EI, totalling 15.38 kg CO_2-e/kg LW produced for sale, as compared to the EI of the annual pasture which totalled 8.92 kg CO_2e-/kg LW. These results reflect a more traditional annual-based grazing model.

6.4.1 Modelling an annual only pasture system

It was hypothesised that the removal of the saltbush pasture from the livestock production system at Wickepin would have little effect on the carbon footprint. Along with the key assumptions in Table 6.3, it was assumed that the 110 ha previously allocated to saltbush pasture would not be re-allocated as this was already marginal land unsuitable for annual pasture or crop production.

Table 6.3 - Key assumptions of the "no perennial" scenario at Wickepin

Baseline	"No perennial" scenario
Annual pasture component was 2,320 ha	Per baseline
Saltbush pasture component was 110 ha	No saltbush pasture
Livestock moved from the annual pasture, either to crop stubble or saltbush, over dry season.	Livestock moved from annual pasture to crop stubble over the dry season, with exception of livestock previously on saltbush which remained on annual pasture to maintain stubble stocking rates per baseline
On-farm lupin production (300 ha) for feedlot ration	Per baseline
	Same saleable liveweight production as baseline

Scenario findings

The carbon footprint of Wickepin under an annual only pasture system was 10.56 kg CO_2-e/kg LW, or 0.4% less than the baseline with saltbush (Table 6.4; Figure 6.1). This smaller footprint resulted from a small decline in whole-farm emissions, which when examined across individual emission sources, could be attributed primarily to the emission savings associated with the establishment and maintenance of the saltbush pasture, including the production and transportation of inputs along with on-farm machinery operation. Together these sources accounted for 68% of the observed decline (Appendix F).

As saltbush was of similar DMD to annual pasture, averaged over the months grazed, the transfer of the ewes to annual pasture did not result in an increase in total supplementary feed requirement, as was exhibited at Bremer Bay. However, the excreta nitrous oxide emissions from ewes on annual pasture in the scenario were lower than when the ewes were on saltbush in the baseline, as the senesced annual pasture had a lower CP content to green saltbush (see Chapter Four). This decline in excreta emissions accounted for almost 30% of the total decline in whole-farm emissions exhibited between the baseline and "no perennial" scenario.

Table 6.4 shows that the EI of annual pasture increased by 1.7%, from 8.95 to 9.09

kg CO_2-e/kg LW in the "no perennial" scenario. This reflects the increase in the proportion of the mature breeding flock on annual pasture when saltbush is removed from the farming system and can be attributed to the breeding flock contributing more GHG emissions than saleable liveweight, driving up the EI of the feedbase. The EIs of crop stubble remained the same as the baseline, as neither stock numbers nor land dedicated to the production of supplementary feed were affected in the scenario.

Table 6.4 - Influence of the "no perennial" pasture scenario on total liveweight (LW) produced, supplementary feed requirement, total emissions (EQ) and emissions intensity (EI) of the Wickepin sheep production enterprise, both at the whole-farm and individual feedbase scale

Scenario	LW produced for sale (kg)	Supplementary feed Pasture/crop stubble (kg)	Supplementary feed Feedlot ration (kg)	EQ (kg CO_2-e/ yr)	EI[a] (kg CO_2-e/ kg LW)
Wickepin baseline with perennial[b]	**108,000**	**404,050**	**-**	**1,145,073**	**10.60**
Annual pasture	90,650	181,679	-	810,863	8.95
Saltbush pasture	2,981	14,819	-	45,847	15.38
Supplementary feed crop stubble	3,200	49,168	-	120,276	37.58
Income crop stubble	11,170	158,384	-	168,087	15.05
Wickepin "no perennial"[b]	**108,000**	**404,039**	**-**	**1,142,850**	**10.56**
Difference	***0.0%***	***0.0%***	***-***	***-0.2%***	***-0.4***
Annual pasture	93,630	196,488	-	851,541	9.09
Difference	*3.3%*	*8.2%*	*-*	*5.0%*	*1.7%*
Supplementary feed crop stubble	3,200	49,168	-	120,539	37.67
Difference	*0.0%*	*0.0%*	*-*	*0.2%*	*0.2%*
Income crop stubble	11,170	158,384	-	168,087	15.05
Difference	*0.0%*	*0.0%*	*-*	*0.0%*	*0.0%*

[a] Where whole-farm EI represents whole-farm emissions produced per kg of liveweight produced for sale across the case study farm, while the EI of the individual feedbases represents the emissions produced on that feedbase per kg of liveweight produced for sale on that feedbase.

[b] Post-allocation of emissions between meat and wool.

As for Bremer Bay, analyses were conducted to ascertain the maximum achievable stocking rates on annual pasture to remain under threshold groundcover levels, finding that maximum levels were 5.5 DSE/ha/yr. The average annual stocking rates of the Wickepin baseline and modelled scenario remained well below this level, averaging between 3 and 3.5 DSE/ha/yr. This demonstrates that annual pasture could withstand additional grazing pressure where saltbush was absent. Considering that most of the breeding flock were transferred to crop stubble over the non-growing period, further investigation is required to ascertain the effect of year-round grazing

on the annual pasture system and thus the carbon footprint influence of stubble grazing (see Chapter Seven).

Overall, the Wickepin enterprise represents a more traditional annual pasture-based livestock system typical of rain-fed dryland agriculture, whereby the farm relies primarily on annual pasture to produce livestock, supported by crop stubble grazing over the non-growing season. Unlike Bremer Bay, whereby perennial pasture played a critical role in the production of livestock and enabled different management practices such as accelerated joining, at Wickepin, saltbush as a grazing source was secondary to its role in the remediation of marginal land. There were no additional emissions, productivity, supplementary feed requirement or groundcover benefits to grazing saltbush at current levels. At the same time, there was no obvious disadvantage to grazing saltbush. This suggests that saltbush, as a pasture component, may be underutilised at Wickepin. Given that during the non-growing season, the breeding flock is grazed predominantly on crop stubble, saltbush may warrant potential in a scenario whereby crop stubble is not grazed. This is investigated further in Chapter Seven.

6.5 Dongara beef production enterprise

The two perennial pasture types at Dongara, subtropical grasses and tagasaste, played critical roles in the production of saleable liveweight by the breeding herd and the backgrounded cattle. Chapter Five found whilst both the annual and perennial pasture components were grazed rotationally year-round, the EIs of each varied considerably, with tagasaste and subtropical grasses producing 6.32 and 9.27 kg CO_2-e/kg LW produced for sale, respectively, whilst annual pasture produced 12.12 kg CO_2-e/kg LW. The higher productivity of perennial pastures over annual pasture was evident in these results. The farm did not provide supplementary feed as the rotational grazing strategy meant that any deficits encountered on annual pasture during the non-growing season were offset by perennials. Although the area established to annual pasture was double that of tagasaste, both pasture types produced approximately 20% of total annual saleable liveweight each. This reflected the ability of tagasaste to continue to support liveweight production throughout the year, as opposed to annual pasture, which while productive during the annual pasture growing season, could not sustain liveweight production outside of this five-month growing period.

6.5.1 Modelling an annual only pasture system

Given the primary role of perennials at Dongara, it was hypothesised that the

enterprise would be unlikely to sustain the same level of production without significant additional external inputs, if at all. As for the two sheep enterprises, a “no perennial” scenario was modelled to examine this (Table 6.5).

Table 6.5 - Key assumptions of the “no perennial” scenario at Dongara

Baseline	“No perennial” scenario
Annual pasture component was 950 ha	Annual pasture component was 3,000 ha
Tagasaste pasture component was 450 ha	No tagasaste pasture
Subtropical grass pasture component was 1,600 ha	No subtropical grass pasture
No supplementary feed provision on pasture	Supplementary feed provided to meet any additional feed deficits on annual pasture over the dry season
	Same saleable liveweight production as baseline

Scenario findings

The carbon footprint for the “no perennial” scenario at Dongara, was 9.46 kg CO_2-e/kg LW (Table 6.6; Figure 6.1), 3.2% higher than the baseline. In order to maintain the same levels of liveweight production as the baseline, over 880 tonne of additional supplementary feed was required to sustain cattle on annual pasture. This increase in supplementary feed was responsible for the majority of the 3.4% increase in whole-farm emissions. As the annual pasture was less input-intensive to perennial pasture, the scenario also observed a concurrent decline in emissions associated with pasture, totalling 101,294 kg CO_2-e. These sources included emissions arising from the production and transportation of inputs, machinery operation, nitrous oxide emissions from fertilisers and carbon dioxide from urea hydrolysis (Appendix F). Despite this decline, it offset only a portion of the additional 215,185 kg CO_2-e produced by the supplementary feed.

A third contributor to the observed change in the whole-farm emissions was an additional 11,517 kg CO_2-e of excreta nitrous oxide emissions resulting from the higher CP content of the supplementary feed provided to livestock as compared to the pasture grazed in the baseline. Enteric methane emissions did not exhibit a noticeable difference between scenarios, primarily a result of the additional supplementary feed provided to fill any feed gap.

Changes in the individual feedbases showed that the EI of annual pasture was 9.54 kg CO_2-e/kg LW, 21.3% lower than the baseline. This lower EI can be attributed to annual pasture in the “no perennial” pasture supporting over 97% of saleable liveweight produced annually. Despite this, the EI remained higher than both

subtropical grasses and tagasaste in the baseline scenario, highlighting the inefficiency of annual pasture as the sole grazing source.

Table 6.6 - Influence of the "no perennial" pasture scenario on total liveweight (LW) produced, supplementary feed requirement, total emissions (EQ) and emissions intensity (EI) of the Dongara beef production enterprise, both at the whole-farm and individual feedbase scale

Scenario	LW produced for sale (kg)	Supplementary feed Pasture/crop stubble (kg)	Supplementary feed Feedlot ration (kg)	EQ (kg CO_2-e/ yr)	EI[a] (kg CO_2-e/ kg LW)
Dongara baseline with perennial	**442,075**	**5,200**	**117,455**	**4,051,957**	**9.17**
Annual pasture	91,099	1,648	-	1,104,269	12.12
Subtropical grass pasture	246,147	2,770	-	2,281,246	9.27
Tagasaste pasture	93,335	782	-	589,531	6.32
Feedlot	11,494	-	117,455	76,911	6.69
Dongara "no perennial"	**442,075**	**887,624**	**117,455**	**4,183,711**	**9.46**
Difference	***-***	***16,969.7%***	***0.0%***	***3.3%***	***3.3%***
Annual pasture	430,581	887,624	-	4,106,801	9.54
Difference	*372.7%*	*53,760.7%*	*-*	*271.9%*	*-21.3%*
Feedlot	11,494	-	117,455	76,911	6.69
Difference	*0.0%*	*-*	*0.0%*	*0.0%*	*0.0%*

[a] Where whole-farm EI represents whole-farm emissions produced per kg of liveweight produced for sale across the case study farm, while the EI of the individual feedbases represents the emissions produced on that feedbase per kg of liveweight produced for sale on that feedbase.

The increased arable land dedicated to annual pasture, 3,000 ha as compared to the 950 ha in the Dongara baseline, meant that annual pasture supported the entire breeding and pastoral herds with a similar average annual stocking rate to the baseline of between 5 and 5.5 DSE/ha/yr. It was modelled that the pasture could withstand stocking rates of up to around 6.5 DSE/ha without breaching groundcover thresholds. In a scenario whereby supplementary feed met feed deficits, such as modelled in this Section, stocking rates remained sustainable from a soil condition perspective. However, it could have implications if alternate scenarios, such as where supplementary feed did not meet feed deficits, are considered.

The requirement for over 880 tonnes of externally sourced supplementary feed to maintain the same saleable liveweight production as the baseline demonstrates the difficulty in maintaining whole-farm productivity without perennial pasture. The annual pasture was demonstrated to support both the breeding and pastoral herds during the annual growing season, although there were large mismatches between pasture supply and cattle demand from November through to June, when annual pasture had

senesced. Where perennial pasture would typically support livestock during this period, supplementary feed was required in the annual only pasture system.

In the “no perennial” scenario, the greatest feed deficits were experienced by pastoral steers and heifers which were sold during this non-growing period. These stock classes were also responsible for 68% of total saleable liveweight produced annually. The large quantities of supplementary feed to produce these stock classes on annual pasture is likely to be unfeasible from both economic and emission perspectives, questioning the ability of the enterprise to background pastoral cattle without perennial pastures. The pastoral cattle were backgrounded to meet target market specifications, so either selling at lower liveweights or retaining them for longer indicated that they will not meet specifications. If retained for a longer period, this would not only drive up the EI of the enterprise, but also increase stocking rates, increasing soil erosion risks and likely reduce annual pasture supply for other cattle on the enterprise. Without pastoral cattle, the farm would have to seek alternative ways to produce saleable liveweight, such as increasing their breeding herd size. Such a venture would require a sizeable increase in breeding herd numbers to achieve even a portion of the saleable liveweight produced by pastoral cattle, accompanied by the increased farm management required to maintain breeding cattle. As previously demonstrated, the breeding herd is also more emissions-intensive than backgrounded cattle. An examination of the feasibility of alternate approaches from emissions and economic perspectives would be valuable to explore further the potential benefits of perennials in a farming system.

It must be noted that whilst the developed carbon footprint Frameworks were capable of automatically calculating changes to emissions and supplementary feed resulting from changes to feedbase attributes, they did not have the capacity to automatically adjust stock growth rates and liveweights in accordance with these feedbase changes. The Frameworks instead relied on manual changes to growth rates. Such functionality could be incorporated but was subject to time and workload limitations in the present study. This meant, for example, that whilst the influence of no perennials could be examined in terms of additional supplementary feed to meet the resultant feed deficits, along with other productivity measures, it was not possible to examine resultant changes to growth rates and liveweights without manual calculations.

Overall, the analyses conducted in Chapter Five and herein demonstrate the critical role that perennials play with regards to both the productivity and carbon footprint at Dongara. The enterprise had adapted its activities to match cattle feed demand with pasture supply, eliminating the requirement for supplementary feed and enabling the

finishing of backgrounding of pastoral cattle on summer-active perennials. This ability to support cattle when annual pasture was senesced had emissions benefits, with the EI of both subtropical grasses and tagasaste lower than annual pasture. Excessive amounts of additional supplementary feed were required to support livestock on annual pasture year-round, driving up the carbon footprint of the enterprise. The same level of production could not be maintained without this supplementary feed, questioning the ability of the farm to maintain pastoral cattle, the primary source of saleable liveweight. Either a reduction in the saleable liveweight produced or a concurrent increase in the breeding herd to offset this reduction would have significant implications on the productivity and carbon footprint of the enterprise.

6.6 Lancelin beef production enterprise

Livestock at Lancelin were rotationally grazed across subtropical grass, tagasaste and annual pastures throughout the production year. Unlike Dongara, however, the farmer selectively grazed different stock classes on different pasture types throughout the production year, according to specific production events such as calving and agistment. Chapter Five analysed the productivity and carbon footprint implications of such grazing management with pasture EIs calculated to be 14.21, 14.65 and 9.84 kg CO_2-e/kg LW produced for sale for the annual, tagasaste and subtropical grass pasture, respectively, reflecting grazing management decisions.

Annual pasture supported the breeding herd during the peak of the growing season from June to August but played a secondary role for the remainder of the year; as reflected in the higher EI. Tagasaste, whilst producing the highest EI of the pasture types, was the dominant grazing source for the breeding herd throughout the year, meeting the energy requirements of the herd without the need for supplementary feed. Though tagasaste supported the production of almost 55% of total liveweight sold annually, its EI was higher because it was grazed predominantly by the breeding herd and thus not a reflection of its poor productivity like the annual pasture, but rather its role in supporting the breeding herd. The breeding herd was in the process of building and as such was comprised of a higher number of non-calving heifers, thus producing more emissions than were offset by saleable liveweight. Finally, the summer-active subtropical grasses, grazed predominantly from November to May, enabled the farm to take on agistment cattle, supporting their high energy requirements. The productivity of the subtropical grasses is reflected in its low EI.

6.6.1 Modelling an annual only pasture system

The three pasture types played specific roles at Lancelin, with the impact of each

reflected in their EIs. The key assumptions of the “no perennial” scenario are presented in Table 6.7.

Table 6.7 - Key assumptions of the “no perennial” scenario at Lancelin

Baseline	**“No perennial” scenario**
Annual pasture component was 450 ha	Annual pasture component was 1,850 ha
Tagasaste pasture component was 1,000 ha	No tagasaste pasture
Subtropical grass pasture component was 400 ha	No subtropical grass pasture
No supplementary feed provision on pasture	Supplementary feed provided to meet any additional feed deficits on annual pasture over the dry season
	Same saleable liveweight production as baseline

Scenario findings

The whole-farm carbon footprint of the “no perennial” scenario was 13.84 kg CO_2-e/kg LW, almost 5% higher than the Lancelin baseline (Table 6.8; Figure 6.1). To achieve the same saleable liveweight as the baseline, over 838 tonnes of additional supplementary feed was required. The emissions arising from the production and transportation of this supplementary feed were responsible for most of the observed increase in total emissions, contributing an additional 186,749 kg CO_2-e (Appendix F). As for Dongara, this increase was offset slightly by the lower emissions associated with the establishment and ongoing maintenance of annual pasture as opposed to perennial pastures (25,410 kg CO_2-e, or 2.6% lower than the baseline).

The increase in whole-farm emissions was also partially offset by reduced enteric methane and excreta nitrous oxide emissions (59,207 kg CO_2-e, 14.9% lower than the baseline). This decline in animal emissions resulted from the modelled changes to the annual pasture following increased stocking rates in the scenario lowering pasture availability through the production year. This meant that whilst supplementary feed met any feed deficits over the non-growing period, pasture supply was also slightly lower over the growing season, reducing intake levels and thus emissions from all stock classes. Excreta emissions declined by a greater proportion from the baseline than enteric methane, a reflection of the transition of the breeding herd from predominantly grazing tagasaste with high CP content to annual pasture which, even when supplemented, was of lower CP content.

As expected, though the modelled annual pasture supported over 97% of annual saleable liveweight in the “no perennial” scenario, its EI was only 4.9% lower than the baseline. Annual pasture was able to support most stock classes over the growing

season from June through to October, although outside of this period it was unable to support liveweight production and this mismatch between pasture supply and cattle demands is reflected in the pasture EI.

Table 6.8 - Influence of the "no perennial" pasture scenario on total LW produced, supplementary feed requirement, total emissions (EQ) and emissions intensity (EI) of the Lancelin beef production enterprise, both at the whole-farm and individual feedbase scale

Scenario	LW produced for sale (kg)	Supplementary feed Pasture/crop stubble (kg)	Supplementary feed Feedlot ration (kg)	EQ (kg CO_2-e/ yr)	EI[a] (kg CO_2-e/ kg LW)
Lancelin baseline with perennial	**150,109**	**179,304**	**-**	**1,981,018**	**13.20**
Annual pasture	26,381	78,998	-	374,744	14.21
Subtropical grass pasture	41,147	53,624	-	404,946	9.84
Tagasaste pasture	81,566	36,548	-	1,195,612	14.65
Feedlot	1,104	-	10,133	5,686	5.15
Lancelin "no perennial"	**150,109**	**1,017,571**	**-**	**2,078,141**	**13.84**
Difference	***-***	***467.5%***	***-***	***4.9%***	***4.9%***
Annual pasture	149,005	1,007,438	-	2,072,455	13.91
Difference	*464.8%*	*1,175.3%*	*-*	*453.0%*	*-2.1%*
Feedlot	*1,104*	-	10,133	*5,686*	*5.15*
Difference	*0.0%*	*-*	*0.0%*	*0.0%*	*0.0%*

[a] Where whole-farm EI represents whole-farm emissions produced per kg of liveweight produced for sale across the case study farm, while the EI of the individual feedbases represents the emissions produced on that feedbase per kg of liveweight produced for sale on that feedbase.

The analysis of groundcover thresholds found that the annual pasture could withstand stocking rates up to around 9 DSE/ha/yr, above the calculated annual average of 6.3 DSE/ha in the scenario. However, the "no perennial" scenario assumed that saleable liveweight production remained that same as the baseline, only possible with significant quantities of purchased supplementary feed. Without additional supplementary feed, the annual pasture would not be able to meet the energy requirements of the agistment cattle following their arrival at the end of the growing season in October. It would also be unable to meet the energy requirements of the weaner bull calves to achieve target growth rates to sale, or the energy requirements of the heifers in the breeding herd. This has potential productivity and emissions implications; questioning the ability of the farm to agist cattle and expand a breeding herd in an annual-based system without significant external inputs.

Chapter Five, supported by the additional analyses of this Chapter, demonstrated that subtropical grasses and tagasaste played critical roles in reducing the whole-farm

carbon footprint of Lancelin, both through reduced emissions and improved productivity. The year-round supply of green feed enabled the farm to support an expanding breeding herd and the agistment of cattle throughout the non-growing period. Without perennial pasture, the farm would likely require economically unfeasible levels of supplementary feed to maintain saleable liveweight production. In the absence of this option, the farm would not be able to support the agistment of cattle or the finishing of production livestock at target sale weights and ages. It would also mean that retained heifers in the breeding herd would take longer to reach reproductive weight, increasing the proportion of emissions produced by the breeding herd to production stock. These factors will reduce productivity and drive up the EI of the enterprise.

6.7 Implications of study findings

The analyses conducted in this study found that perennial pasture influences the carbon footprint of livestock production enterprises. This can occur through a direct change to emissions, such as a decrease associated with the reduced requirement for supplementary feed, or even an increase associated with increased animal emissions following improved pasture quality through the non-growing season, for example. The greatest influence, however, results from the increased whole-farm productivity that accompanies perennial pasture and this offsets any concurrent increase in emissions resulting from increased feed intake.

The provision of green, quality pasture during months where annual pasture is senesced and unable to meet the energy requirements of livestock, means that farm managers can implement practices which would not be feasible in an annual-based system. For example, the kikuyu pasture at Bremer Bay enabled the farm to adopt an accelerated joining system while still matching pasture supply with livestock demand. The subtropical grasses and tagasaste at Dongara and Lancelin enabled the farms to background and agist cattle during the non-growing season, alongside on-farm breeding herds. The increased whole-farm productivity resulting from the additional saleable liveweight produced outside of the annual growing season, reducing the EI of the enterprises. This is more obvious at Dongara whereby the majority of saleable liveweight was produced by pastoral cattle on perennials, as opposed to Lancelin where the majority of saleable liveweight was instead produced by the breeding herd, accompanied by further increased emissions associated with maintaining an expanding on-farm breeding herd.

The analyses also demonstrate how the role of the perennials, and the effect on the

whole-farm carbon footprint and productivity, varies across different livestock systems. These differences are exhibited across different climatic zones, perennial pasture species, farming system and management practices. For example, at Dongara and Lancelin, where deep sands and long dry summers restrict annual pasture production and render the enterprises unsuitable for sheep or crop production, the perennial species play a critical role in the viability of the farming systems, with ensuing effects on the whole-farm carbon footprints. At Wickepin by contrast, the saltbush plays a secondary role, with crop stubble and supplementary feed predominant feed sources over the non-growing period. This is reflected in the minor influence of the saltbush on the carbon footprint and productivity of Wickepin. Despite the inability of the region to support most temperate and subtropical perennial species, increased application of saltbush could have a more active role in livestock production should the farm transition from stubble grazing.

No other examined carbon footprint or LCA studies have directly compared the role of different pasture types established simultaneously on a livestock production system. Some Australian studies, however, have examined the emissions impact of changing pasture type or the influence of altering the pasture species grazed at different stages of production (Harrison et al., 2014a; Taylor et al., 2016; Thomas et al., 2012). Taylor et al. (2016) found that herd EIs of steers and heifers grazing the perennial forage species, Leucaena, in north-eastern Australia, were lower than when the cattle grazed unimproved native vegetation. This was attributed to both the higher achievable growth rates of livestock on Leucaena, resulting in shorter periods to reach target sale weights, the greater carrying capacity on Leucaena and the anti-methanogenic properties of Leucaena which lowered enteric methane production per unit of intake. Harrison et al. (2014a) found that the combined wool and liveweight production EIs of a sheep enterprise in south-eastern Australia was higher in a modelled annual pasture only scenario as compared to perennial legume, perennial grass, or the baseline of mixed perennials. This was attributed to the lower carrying capacity of the annual pasture, particularly during the summer-autumn period whereby pasture availability and quality was lower than perennials, but also just prior to lambing. This increased supplementary feed requirements and restricted animal production. Similarly, Thomas et al. (2012) modelled in GrassGro the impact of converting annual pasture to kikuyu on a sheep enterprise in south-western Australia, finding that the increases in stocking rate and subsequent liveweight and wool production, outweighed any corresponding increases in animal emissions. Though these studies were not carbon footprint analyses (Taylor et al. (2016) and Thomas et

al. (2012) considered animal emissions only), they reflected the findings of the present study, that increased whole-farm productivity associated with perennial pasture will reduce the carbon footprint, as EI, through increased liveweight production.

Clearly the key benefit of perennials is the improved productivity provided and the resultant impact on EI. Similar improvements, through improved pasture quality and production rather than specifically the introduction of perennials, has been observed by a multitude of studies domestically (Alcock & Hegarty, 2006; Waghorn & Clark, 2006), but primarily internationally (Becoña et al., 2014; de Figueiredo et al., 2017; Dick et al., 2015a, 2015b; Modernel et al., 2013; Ruviaro et al., 2015; Toro-Mujica et al., 2017). Each of these studies found that pasture systems of higher quality and production produced lower EIs than unimproved or lower quality pastures. This was driven by both the increased carrying capacity and growth rates on the improved pastures. Most these studies were located in southern America, a region also predominantly dependent on broadacre livestock production. For example de Figueiredo et al. (2017) found that whilst total emissions increased under improved pastures from increased feed intake and stocking rates, EI decreased; a result of improved livestock productivity. This was also observed in the present study in the three enterprises for which perennials played an important role (Dongara, Lancelin and Bremer Bay). In each case, although total emissions from the perennials were higher on a per hectare basis than the annuals, on a saleable liveweight production basis, they produced lower EIs to the annual pasture (aside from the tagasaste at Lancelin which supported the breeding herd).

In southern Australia, and thus the case study farms in the present study, even with the most well managed annual pasture system, an enterprise still must contend with the feed gap during the non-growing season. The perennials fill this gap, improving average pasture quality and production across the farm. Similar results have been observed in Brazilian beef production, whereby the establishment of winter grasses to fill the autumn-winter gap decreased whole-farm EI (Dick et al., 2015b). This reduction, like the findings of the present study, was a result of increased growth rates and livestock production enabled by perennial pasture supply over the feed gap.

The importance of grazing management practices, particularly the alignment of pasture supply to livestock demand, was highlighted in the carbon footprint analyses conducted in the present study. For example, at both beef enterprises, calving occurred at the respective start of the annual growing season, coinciding with high quality pasture production. Backgrounding and agistment stock were retained on-farm over periods whereby perennial pasture production and quality could support

additional livestock at no detriment to the breeding herd. At Bremer Bay, inclusion of the kikuyu pasture into careful grazing management ensured two successful lambing cycles annually. In all instances, the analyses demonstrated that the exhibited productivity of these farms could not be maintained without perennials or without high levels of supplementary feed, along with carbon footprint ramifications. Other studies have also highlighted that the practice of matching pasture supply to livestock feed requirements, through its productivity benefits, can improve EI (Alcock & Hegarty, 2011a; Alcock et al., 2015; Ledgard et al., 2011; O'Brien et al., 2016). Alcock and Hegarty (2011a) and Alcock et al. (2015) for example, demonstrated that matching key production events, such as lambing, with periods of peak pasture production optimised the EI of sheep enterprises. Shifting lambing by one to two months for example, increased EI by up to 10.5%, a consequence of insufficient pasture supply to either meet the energy requirements of breeding ewes or to enable lambs to achieve target growth rates. In the present study, perennials extended the period during which pasture could support increased livestock feed demands, enabling farms to adopt these productivity-enhancing practices, delivering concurrent EI benefits.

6.8 Carbon sequestration potential of perennial pasture

Chapter Two outlined the difficulties associated with the inclusion of carbon sequestration in whole-farm carbon footprint studies. Despite this, LEAP recommends that where possible these values are incorporated into analyses but reported separately. As such, this Section attempts to model potential carbon sequestration at the examined livestock enterprises. To date, research investigating carbon sequestration and storage under subtropical perennial grasses has yielded highly variable results (Sanderman et al., 2013; Thomas et al., 2012). Given this uncertainty, it was assumed that there was no net flux of carbon in the subtropical grass pastures at Bremer Bay, Dongara and Lancelin. However, a review of literature revealed existing information regarding carbon sequestration rates of the woody shrub species saltbush and tagasaste grown at Wickepin, Dongara and Lancelin. As such, the abatement potential of these forage species was modelled for the three enterprises.

6.8.1 Modelling the sequestration potential of saltbush and tagasaste

To model the potential carbon sequestration of saltbush at Wickepin, the findings of Walden et al. (2017) were employed. This study measured soil carbon and provided subsequent estimates of the sequestration rate of saltbush at several sites, including the Wickepin case study farm. The carbon sequestration rate at the Wickepin site was

estimated to be 8.33 t C/ha over 13 years, or 0.64 t C/ha/yr and converted to carbon dioxide equivalent, equated to 2.35 t CO_2-e/ha/yr. These calculated values were obtained from the Wickepin site with the same saltbush density as applied in the current study (2000 shrubs/ha), and so no adjustment for plant density was required.

The potential sequestration, after protein mass allocation, by the 110 ha of saltbush pasture was calculated to be 159.24 t CO_2-e/ha/yr. This reduced the carbon footprint of Wickepin, expressed as EI, to 9.13 kg CO_2-e/kg LW produced for sale, an offset of almost 14% (Table 6.9; Figure 6.2). Clearly there is potential for saltbush pasture to offset for the carbon footprint of the enterprise.

Table 6.9 - Changes to the carbon footprint, expressed as total emissions (EQ) and emissions intensity (EI), following the application of modelled forage shrub sequestration rates at the Wickepin, Dongara and Lancelin livestock production enterprises

Scenario	C sequestration rate (t CO_2-e/ ha/yr)	EQ (kg CO_2-e/yr)	EI (kg CO_2-e/ kg LW)	Difference (%)
Wickepin baseline[a]	**-**	**1,145,073**	**10.60**	**-**
Wickepin saltbush sequestration	2.35[b]	985,834	9.13	-13.9%
Dongara baseline	**-**	**4,051,957**	**9.17**	**-**
Dongara tagasaste sequestration – adjusted for tree density	8.26[c]	333,249	0.75	-91.8%
Dongara tagasaste sequestration – unadjusted	5.33[d]	1,655,707	3.75	-59.1%
Lancelin baseline	**-**	**1,981,018**	**13.20**	**-**
Lancelin tagasaste sequestration – adjusted for tree density	22.72[c]	-20,736,471	-138.14	-1,146.8%
Lancelin tagasaste sequestration - unadjusted	5.33[d]	-3,343,982	-22.28	-268.8%

[a] Post-allocation of emissions between meat and wool.
[b] Sourced from Walden et al. (2017).
[c] Sourced from Wocheslander et al. (2016) and adjusted per Thomas et al. (2015).
[d] Sourced from Wocheslander et al. (2016).

To model the potential abatement potential of the tagasaste pasture systems at Dongara and Lancelin, the sequestration rates estimated in Wocheslander et al. (2016) were adopted and adjusted according to the specifics of each enterprise. The estimated sequestration rates were obtained from 22-year-old tagasaste plantings at a site in Moora, Western Australia, approximately 110 km from Lancelin and 200 km from Dongara. The tagasaste plantings had not been grazed and as such, following

a report by the EverCrop Carbon Plus project (Thomas et al., 2015), which used the study findings, the estimated sequestration values were halved to account for the grazing of the tagasaste at both Dongara and Lancelin. These values were then adjusted for the planting density of Dongara (1,299 trees/ha) and Lancelin (3,571 tree/ha), as calculated in Chapter Four; yielding sequestration estimates of 8.26 t CO_2-e/ha/yr at Dongara, and 22.72 t CO_2-e/ha/yr at Lancelin.

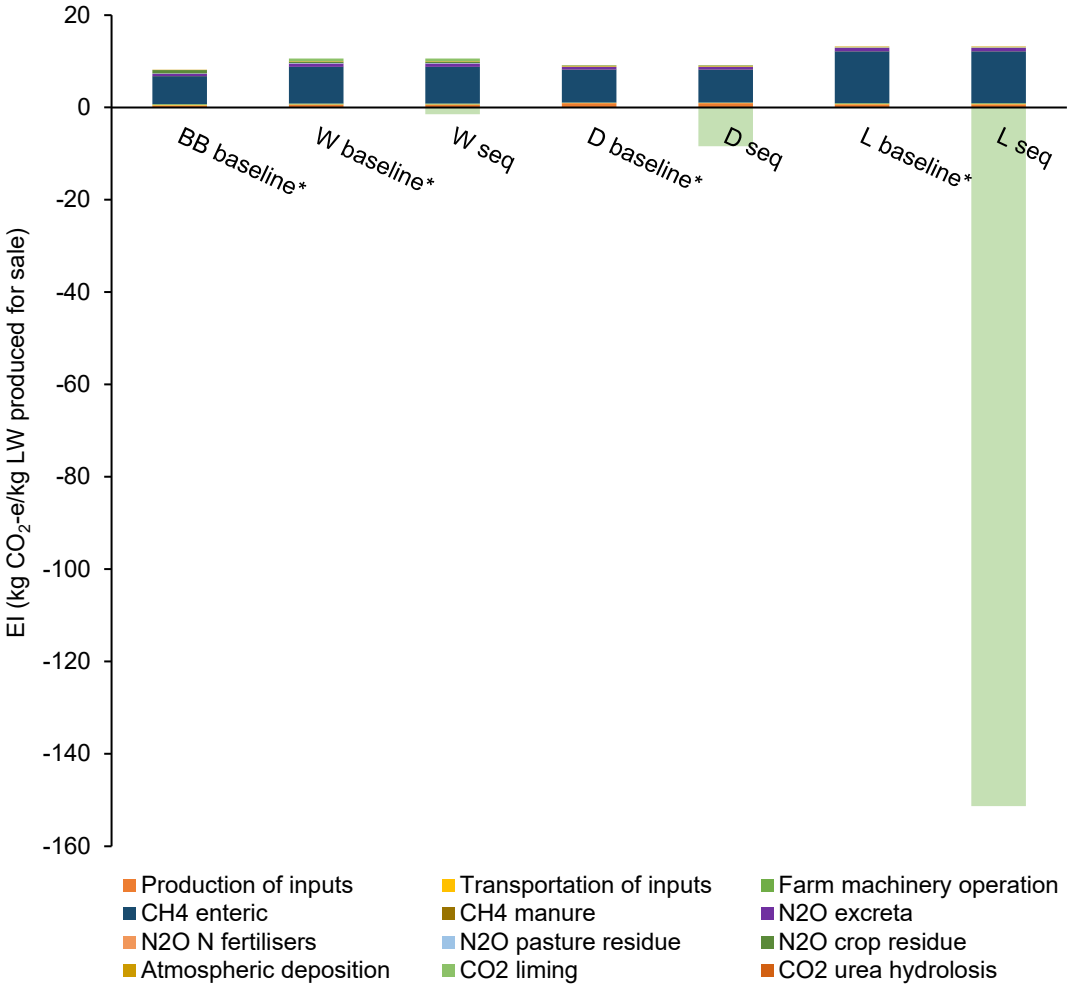


Figure 6.2 - The baseline carbon footprints, expressed as EI, of the Bremer Bay, Wickepin, Dongara and Lancelin livestock production enterprises, as compared to the respective "carbon sequestration" scenarios.

Where: the Wickepin sheep production enterprise carbon footprint results are presented post-allocation of emissions between meat and wool production. Seq. = sequestration.

Using these calculated values, the estimate sequestration potential of the 450 ha of tagasaste at Dongara totalled 3,718.01 t CO_2-e/ha/yr and reduced the carbon footprint, expressed as EI, by almost 92%, to 0.75 kg CO_2-e/kg LW. The sequestration potential of the 1,000 ha of tagasaste at Lancelin was estimated to be 22,717.49 t CO_2-e. Expressed as EI, this reduced the carbon footprint to -138.14 kg CO_2-e/kg LW, a reduction of 1,147%.

6.8.2 Methodological and data considerations when estimating sequestration potential

The inclusion of carbon sequestration estimates for the saltbush and tagasaste demonstrated significant potential for carbon offsets. However, the high uncertainty of including such estimates in carbon footprint analyses, as described in earlier Chapters, meant that these results are preliminary and must be interpreted cautiously.

In the case of saltbush, the estimated sequestration rate was based on direct site measurements averaged over 13 years. The study found that sequestration rates had increased in later years and recommended the evaluation of longer-term plantings. By contrast, two other southern Australian studies employed saltbush sequestration rate estimates of 0.07 t CO_2-e/ha/yr, averaged over 100 years (Henry et al., 2015b; Mayberry et al., 2019). Unlike Walden et al. (2017) which estimated sequestration from direct measurements of biomass and soil C, Henry et al. (2015b) developed their estimates of saltbush sequestration from biomass change obtained from observations of photographs taken of the case study sites, across a non-disclosed period of time. The final sequestration rates were annualised over 100 years and considered biomass C only, excluding soil C. These values were then also applied to nation-wide estimates of the abatement potential of chenopod shrubs by Mayberry et al. (2019), which were then used in the argument for achieving a carbon-neutral red meat industry by 2030 (see Chapter Two). Considering that the estimated sequestration rate of saltbush across the sites directly measured in Walden et al. (2017) differed by up to 74%, there would be increased uncertainty accompanying the application of location-specific values estimated through visual estimates of biomass change, which exclude soil C as per Henry et al. (2015). This uncertainty would be even greater when applying such estimates on a nationwide scale per Mayberry et al. (2019) as it does not consider the many factors likely to influence carbon sequestration across Australia, such as climate, soil type, year of planting, planting density.

The period of amortisation will also play an important role. If, for example, the 100-year amortised sequestration rates of 0.07 t CO_2-e/ha/yr used in Henry et al. (2017) and Mayberry et al. (2019) are applied to Wickepin, the total abatement potential of saltbush is only 4.74 t CO_2-e/yr, reducing the EI by less than 1%, to 10.56 kg CO_2-e/kg LW. It was not possible to adjust the value for the planting density or biomass at Wickepin as this information was not provided in the original study. The lifetime of the plants themselves must also be considered, as a 100-year period may not be appropriate for shorter lived shrubs like saltbush. If, the 20-year amortised value also presented by Henry et al of 0.30 t CO_2-e/ha/yr is instead applied, the total abatement

potential at Wickepin is 20.33 t CO_2-e/yr, reducing EI by 1.8% to 10.41 kg CO_2-e/kg LW. The values of Walden et al. (2017) were applied in this study as they were deemed the most appropriate and accurate, however these values are likely to change over a longer period of observation. The high variability both within Walden et al. (2017) and between other studies highlights the role that adopted methodology, C-inclusions and exclusions, selected amortisation period, location and accompanying growth patterns will have on sequestration estimates.

The sequestration estimates of tagasaste are also likely to demonstrate high variability. Though the estimated rates at Dongara and Lancelin were halved to account for the grazing of tagasaste following the approach of another study using the same data (Thomas et al., 2015), they were still quite high as compared to estimates of other trees. No other sequestration estimates were found for tagasaste within literature, however the sequestration rates of trees ranged from 7.7 to 17.4 t CO_2-e/ha/yr over a similar time period of 20 years, or up to 31.8 t CO_2-e/ha/yr over 30 years for a tree plantation (Eady et al., 2011). Other estimates range from 5.87 to 9.17 t CO_2-e/ha/yr (Doran-Browne et al., 2017; Doran-Browne et al., 2016). However, the variations in these studies are a result of differences pertaining to tree species, location, climatic conditions, methodological approach and planting density. In addition to this, the sequestration rate varies considerably over the lifetime of a tree (Doran-Browne et al., 2016). As such, it is difficult to compare such values. It is possible that a 50% reduction of values obtained from unmanaged tagasaste may underestimate the effect of grazing. It is also possible that the higher density of the tagasaste on both farms may influence these higher values. If for example, the original value obtained by Wochesländer et al. (2016), adjusted for grazing only and not planting density, is applied to Dongara and Lancelin, different values are obtained. Sequestration is calculated to be 5.33 t CO_2-e/ha/yr. The revised abatement potential at Dongara using this rate is calculated to be 2,396.25 t CO_2-e/ha/yr, resulting in an EI reduction of 59% to 3.75 kg CO_2-e/kg LW (Table 6.5). At Lancelin, this unadjusted rate revised the abatement potential to 5,325.00 t CO_2-e/ha/yr, resulting in an EI reduction of 269%, or -22.28 kg CO_2-e/kg LW. Though the abatement potential at both farms is less using this unadjusted rate, tagasaste still offsets a significant portion of the farm's carbon footprint, or even converts the farm into a carbon sink.

Despite the current uncertainty regarding modelling soil C fluxes, there is clearly scope for the incorporation of the abatement potential of forage shrub and tree species into the carbon footprint calculation of livestock production systems. Other methods of sequestration, as promoted in the federal ERF scheme, such as native

revegetation or tree plantations, require a re-allocation of a portion of arable land away from livestock production and are often not economically viable to the farmer (Doran-Browne et al., 2017; Farquharson et al., 2013). However, forage species are established for purposes other than abatement, whether it be environmental with an additional benefit of dry season feed such as saltbush, or the provision of quality year-round feed such as tagasaste. The previous Chapters demonstrated the role of perennials in improving the productivity and carbon footprint of livestock production systems. The abatement potential of these species is an added benefit of their on-farm use. This requires further research to streamline calculation methodology and improve spatial and temporal data quality to enable the farm-specific sequestration estimates required for the accurate inclusion of this emission offset.

6.9 Conclusion

The scenario analyses conducted in this Chapter demonstrated that perennial pasture as a component of a grazing system can result in a lower whole-farm EI than a system with annual pasture only. This aligns with the findings of Chapter Five. This difference can be a result of a direct change to gross emissions, such as lower emissions associated with the reduced requirement for supplementary feed by livestock grazing perennial pasture, or even higher emissions associated with increased animal emissions whilst grazing perennials of higher quality than annual pasture through the non-growing season, for example. Primarily though, the observed EI benefit of perennial pasture occurs through increased whole-farm productivity which can offset any concurrent increase in emissions resulting from increased feed intake.

The preliminary estimates of C sequestration under perennial shrubs indicate that there are potentially considerable benefits, with sequestration in some cases calculated to entirely offset the footprint or even convert it to a net emissions sink. Further research attention into developing a standardised methodology and improved data availability is required to improve the accuracy of such farm-specific estimates in Australian production systems.

The impact of perennials on a livestock system is farm-specific, dependent on the characteristics and farming practices of the enterprise in question. Perennials can, for example, enable a farm to shift focus to employ more productivity-driven practices (i.e. agistment, accelerated joining), convert previously marginal land to that suitable for livestock production and reduce supplementary feed requirement. The following Chapter explores further the impact of other farming practices, in some cases enabled by perennials, identified in the present and previous Chapters.

7 THE INFLUENCE OF FARM PRACTICES ON THE CARBON FOOTPRINT OF LIVESTOCK PRODUCTION SYSTEMS

7.1 Introduction

Chapter Five presented the carbon footprint analyses for the four considered livestock production systems. The whole-farm carbon footprint of each system was calculated and inter- and intra-farm variability highlighted, driven by pasture type and management practices. Whilst Chapter Six explored the role of pasture type in detail, this Chapter examines the impact of selected farm practices on both the carbon footprint and productivity of livestock production systems. In doing it identifies strategies with mitigation potential, fulfilling the third objective of this study, to:

Examine the mitigation potential of identified strategies on the carbon footprint of livestock production systems and provide regionally appropriate recommendations for application.

Specifically, this Chapter assesses the mitigation potential of several grazing and reproductive management practices. The selection of these practices was based on their identification in existing research, their short-term applicability, ease of access and relative cost-effectiveness. In some cases, one or more of the farms may have already adopted an identified strategy, making it possible to examine any changes to emissions and productivity because of this implemented strategy. The final section of this Chapter considers other factors which may influence the adoption of the strategies presented and provides recommendations for further research.

7.2 Investigating the mitigation potential of selected farm practices

The review conducted in Chapter Two found that the most promising mitigation strategies available for immediate uptake by farmers were those associated with improved reproductive efficiencies and grazing management. The effectiveness of these strategies typically centres around their ability to alter the flock/herd structure to increase the proportion of production livestock as opposed to breeding livestock (i.e. increased fecundity, weaning rates, replacement animals) or improve growth and turnoff rates (i.e. improved pasture management, feedlotting).

Whilst other strategies, typically technological advances (i.e. dietary supplements or

vaccines which act to inhibit animal emissions, or breeding for selected traits), offer the potential for greater direct impact on emissions, their adoption is limited by cost and lack of widespread evidence of effectiveness beyond laboratory or controlled trials (Beauchemin et al., 2020; Eckard & Clark, 2020b). Many of these strategies are more suited to intensive systems as there are implementation difficulties on extensive systems with lower levels of management. Similarly, while there may be other, non-livestock system specific strategies, with mitigation potential, such as the use of renewable or alternate fertilisers, these typically target processes within the livestock farming system which are not hotspots. As such their relative impact on the whole-farm carbon footprint will be less than those which target the system hotspots. Improving reproductive and grazing efficiencies, by contrast to these alternate strategies, typically lower the carbon footprint whilst improving productivities and are well-evidenced on-farm and available for immediate uptake (Beauchemin et al., 2020). As such, such the uptake of strategies are prominent pathways promoted to achieve the carbon reduction goals set by Australia's red meat industry (Mayberry et al., 2019).

There is scope for significant reductions in emissions through the application of farm practices with the potential for improved efficiencies. Gerber et al. (2013) suggested that a carbon footprint reduction of up to 30% may be possible if producers adopted the practices employed by the 10% of producers with the lowest footprints. The findings of LCA and carbon footprint studies which have quantified the impact of and compared practices employed across case study farms align with this assertion. In these studies, regardless of country or region, the primary differences between case study farms with high and low EIs were their respective reproductive and grazing efficiencies (Becoña et al., 2014; Hyland et al., 2016; Jones et al., 2014; Nieto et al., 2018; Toro-Mujica et al., 2017; Veysset et al., 2014). For example, Hyland et al. (2016) found that if the higher emitting farms in its UK study adopted the practices of the lower, the EI of the sheep and beef production enterprises could be reduced by 30.5% and 15%, respectively. The practices identified by these studies, such as increased reproductive rates, joining ages, pasture management, lamb growth rates, concentrate feeding, have also been identified as the most effective strategies for mitigation on modelled Australian farms in non-carbon footprint studies (Alcock et al., 2015; Cottle et al., 2016; Cullen et al., 2016; Harrison et al., 2014a; Harrison et al., 2014b; Ho et al., 2014). The effectiveness of these strategies is because they target the primary hotspot, enteric methane. Strategies which target hotspots will have a greater impact on the carbon footprint of an enterprise than lesser emission sources,

such as the production and transportation of inputs (Beauchemin et al., 2011). Along with productivity and emissions benefits, these strategies have evidenced whole-farm impact and as such demonstrate the ability to overcome the economic, farmer perception and knowledge transfer barriers to adoption.

Chapters Four and Five identified multiple farm practices and characteristics which influenced the carbon footprint of the respective enterprise under examination. These included the establishment of perennial pastures and the pasture management that they afford, feedlot finishing livestock, accelerated joining systems, breed and time of lambing/calving, backgrounding of stock and herd structure. A number of these practices align with those identified in the studies outlined above. The impact of perennial pasture and grazing management on the carbon footprint of livestock production systems was outlined in the previous Chapter. Two more grazing management strategies were selected for closer examination:

- feedlot finishing
- stubble grazing

Alongside these, two strategies targeting reproductive efficiencies were examined:

- the alteration of herd structure through replacement animals
- weaning rates

The purpose of this approach was to recognise those farmers already employing strategies with mitigation potential, even though the motivator may have been economic and/or enhanced productivity. Demonstrated evidence of the effectiveness of proven farm strategies may increase the potential for adoption by other farmers.

7.3 Feedlot finishing of livestock

In the present study, Bremer Bay and Dongara employed on-farm feedlot finishing of livestock. Though not external industrial feedlot operations as described by Wiedemann et al. (2016a) or confinement feeding to combat dry periods (Ghahramani & Moore, 2013; Moore & Ghahramani, 2013), these on-farm feedlots operated to finish livestock to meet target specifications for sale. Chapter Five found that the feedlots produced the lowest EIs of all the feedbases. This resulted from the high feed conversion efficiency of livestock in feedlots, enabling higher growth rates and shorter turnoff periods. Following these findings, this Section explores the effect of feedlot finishing livestock on the carbon footprints of all the farms.

7.3.1 Bremer Bay sheep production enterprise

Scenario assumptions

In the Bremer Bay baseline, Dorper lambs were finished in an on-farm feedlot. As lambs were produced over two annual lambing cycles in March and September, feedlotting occurred in August and February, respectively, with the provision of a high-quality feedlot ration to ensure higher growth rates. To examine the emissions impact of feedlotting lambs, a "no feedlot" scenario was developed, applying the key assumptions presented in Table 7.1.

In recognition of the importance of maintaining productivity, the scenario was modelled so that the same amount of saleable liveweight was produced as the baseline. Initial analyses determined that even without the feedlot and thus the requirement for feedlot ration components, supplementary feed requirement on pasture and crop stubble increased by over 40% (Table 7.2). Whole-farm supplementary feed requirements were thus 11% higher in the scenario, increasing the demand for farm-produced oats by 19%.

Table 7.1 - Key assumptions of the "no feedlot" scenario at Bremer Bay

Baseline	"No feedlot" scenario
Lambs finished in feedlot for one month prior to sale at 5.5 months	Lambs finished on pasture prior to sale at 6.5 months
Lambs produced in March lambing finished across annual and perennial (no supp. feed)	Per baseline
Lambs produced in September lambing finished on perennial pasture (with supp. feed)	Per baseline
On-farm lupin production (40 ha) for feedlot ration	No lupin production (0 ha) as no feedlot. Instead 20 ha dedicated to increase oat production and 20 ha to pasture.
On-farm oat production (90 ha) for supplementary feed	Increased on-farm oat production (110 ha) to account for 19% increase in oats required for supplementary feed.
Breeding flock grazed both lupin and oat stubble in dry season	Breeding flock grazed oat stubble only
	Same saleable liveweight production as baseline

Scenario findings

The results of the "no feedlot" scenario are presented in Figure 7.1 and Table 7.2. Compared to the Bremer Bay baseline whereby lambs were finished in the feedlot, the carbon footprint of finishing lambs on pasture, expressed as EI, was 8.86 kg CO_2-e/kg LW produced for sale, or 8.3% higher. Changes to enteric methane output was responsible for most of this increase and enteric methane was 11.3% higher in the

scenario (Appendix F). The increase was primarily a result of the increased enteric methane emissions from Dorper lambs on pasture for the additional two months, as compared to the baseline where they were finished in the feedlot over one month.

This follows the findings of Chapter Four, that though daily enteric methane production of lambs was higher when in the feedlot due to increased intake, this was offset by the increased liveweight gain and shorter time to sale. By contrast, though daily enteric methane production by lambs was lower on pasture, the longer time to sale resulted in higher net emissions. A portion of the increased whole-farm enteric methane output can also be explained by increased livestock numbers on oat stubble. Of lower quality than lupin stubble in the baseline, stock grazing oat stubble required greater amounts of supplementary feed, increasing intake and enteric methane production. The second emission source responsible for the observed change in whole-farm emissions was excreta nitrous oxide, which increased by 7.5%, driven by similar reasons to the exhibited increase in enteric methane.

Table 7.2 - Influence of the "feedlot" pasture scenario on total liveweight (LW) produced, supplementary feed requirement, total emissions (EQ) and emissions intensity (EI) of the four livestock production enterprises

Scenario	LW produced for sale (kg)	Supplementary feed Pasture/crop stubble (kg)	Supplementary feed Feedlot ration (kg)	EQ (kg CO_2-e/ yr)	EI (kg CO_2-e/ kg LW)
Bremer Bay baseline	**57,975**	**81,634**	**40,738**	**474,185**	**8.18**
Bremer Bay no feedlot	57,975	136,721	-	513,481	8.86
Difference	-	*40.3%*	*-100%*	*8.3%*	*8.3%*
Wickepin baseline[a]	108,000	404,050	-	1,145,073	10.60
Wickepin feedlot Merino	108,000	381,284	24,897	1,128,192	10.45
Difference	-	-5.6%	100%	-1.5%	-1.5%
Wickepin feedlot Merino & SAMM	108,000	379,060	53,554	1,116,308	10.34
Difference	-	*-6.2%*	*100%*	*-2.5%*	*-2.5%*
Dongara baseline	***442,075***	***5,200***	***117,455***	***4,051,957***	***9.17***
Dongara feedlot pellet	*442,075*	*3,063*	*1,019,897*	*3,975,880*	*8.99*
Difference	-	*-41.1%*	*88.5%*	*-1.9%*	*-1.9%*
Dongara feedlot lupin	*442,075*	*3,063*	*956,648*	*3,865,696*	*8.74*
Difference	-	*-41.1%*	*87.7%*	*-4.6%*	*-4.6%*
Lancelin baseline	*150,109*	*179,304*	-	*1,981,018*	*13.20*
Lancelin feedlot lupin	*150,109*	*171,543*	*51,714*	*1,973,170*	*13.14*
Difference	-	*-4.3%*	*100%*	*-0.4%*	*-0.4%*

[a] EQ and EI values presented for Wickepin are post-allocation between wool and meat production.

In line with the changes to arable area dedicated to crop and pasture, emissions from

crop residues declined by 2.5% while pasture residues increased by 2.0%. Reflecting the higher input nature of producing the lupin crop, emissions from the production and transportation of inputs were lower in the “no feedlot” scenario (4.4% and 3.3%, respectively). This offset any increased inputs on the land converted to oats or pasture. Despite the observed changes, these emissions were minor contributors to the whole-farm carbon footprint and only offset a minor portion of the increased animal emissions associated with finishing livestock on pasture.

Overall, this analysis demonstrated that feedlot-finishing Dorper lambs instead of finishing them on pasture lowered the whole-farm carbon footprint, improved feed conversion efficiency of lambs and reduced the net requirement for supplementary feed across the farm (pasture, stubble and feedlot).

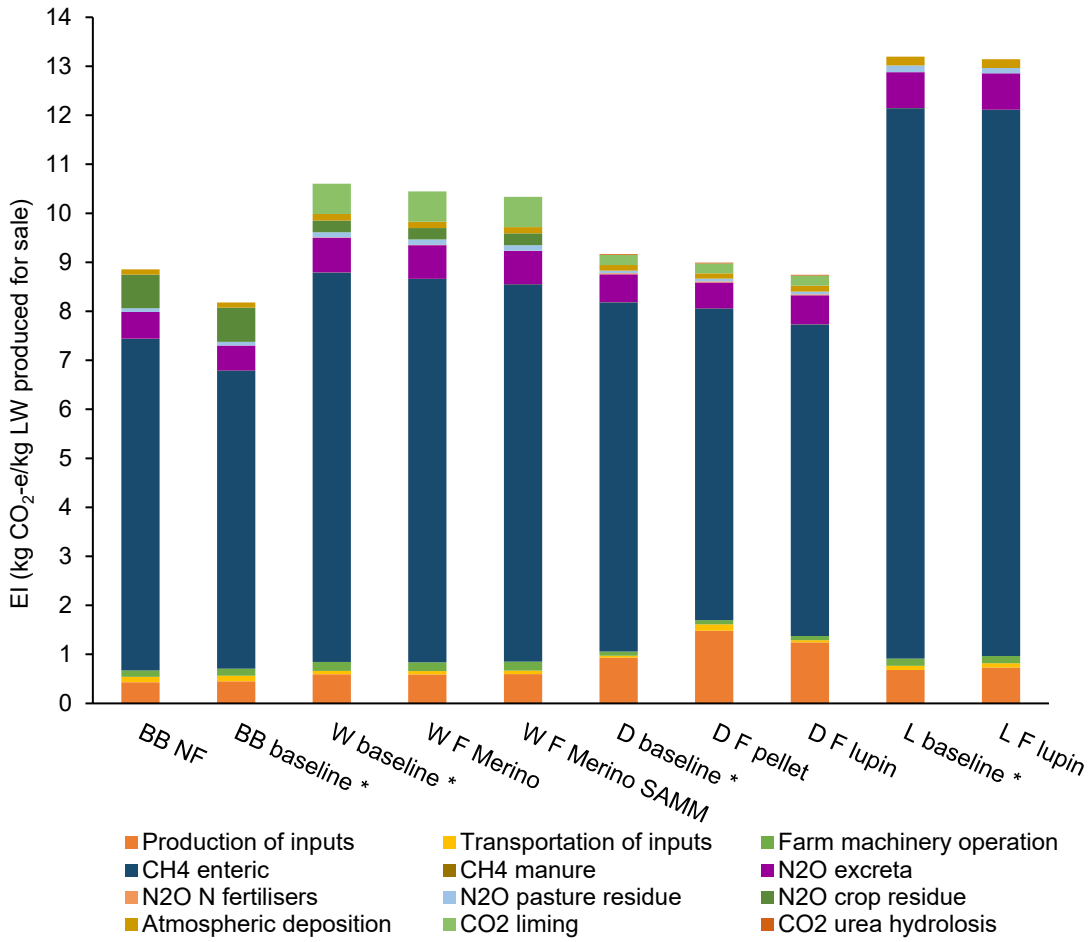


Figure 7.1 - The baseline carbon footprints, expressed as EI, of the four livestock production enterprises, as compared to the respective “feedlot” scenarios.

Where: the Wickepin sheep production enterprise results are presented post-allocation of emissions between meat and wool. NF = no feedlot; F = feedlot.

7.3.2 Wickepin sheep production enterprise

Scenario assumptions

In the Wickepin baseline, Merino and SAMM lambs were sold off pasture or crop stubble. To model the effect of instead finishing lambs in an on-farm feedlot, two scenarios were developed, “feedlot Merino” and a “feedlot Merino SAMM”. Table 7.3 outlines the key assumptions associated with each scenario.

Scenario findings

The carbon footprint for the “feedlot Merino” scenario was 10.45 kg CO_2-e/kg LW produced for sale, 1.5% less than the Wickepin baseline without feedlot finishing of lambs (Figure 7.1; Table 7.2). The feedlot finishing of both Merino and SAMM lambs in the “feedlot Merino SAMM” scenario, yielded a lower carbon footprint of 10.34 kg CO_2-e/kg LW, 2.5% less than the baseline. In both scenarios a decline in enteric methane was responsible for most of these reductions (Appendix F).

Table 7.3 - Key assumptions of the “feedlot” scenarios at Wickepin

Baseline	**“Feedlot Merino” scenario**	**“Feedlot Merino SAMM” scenario**
Merino wethers moved to lupin stubble from annual pasture at beginning of January	Merino wethers moved to feedlot from annual pasture at beginning of December	Per “feedlot Merino” scenario
Merino wethers sold off lupin stubble at the end of January at 8 months of age	Enabled by increased growth rates in feedlot, Merino wethers sold from feedlot at end of December at 7 months of age	Per “feedlot Merino” scenario
SAMM lambs sold off annual pasture at weaning at 6 months of age	Per baseline	SAMM lambs weaned on pasture at 4 months and moved to feedlot for 5 weeks before attaining target sale weight
No feedlot ration provided	Feedlot ration provided. Assumed to be same as Bremer Bay baseline.	Per “feedlot Merino” scenario
Merino wethers, maiden ewes and older replacement ewes grazed lupin stubble in dry season	Maiden ewes, older replacement ewes grazed lupin stubble during dry season, along with additional mature ewes from income crop stubble to match SR of baseline	Per “feedlot Merino” scenario
On-farm lupin production (300 ha) for supplementary feed	Decreased on-farm lupin production (290 ha) to account for 3% decline in lupin requirement (Table 7.4). The extra 10 ha allocated to annual pasture.	Per the baseline as only 1% decline in lupin requirement, equivalent to 3 ha (Table 7.4), thus no change assumed.
	Same saleable liveweight production as the baseline	Same saleable liveweight production as the baseline

Finishing Merino lambs in a feedlot in December meant that a month of grazing senesced annual pasture and another of grazing lupin stubble was avoided, eliminating the need for large quantities of supplementary feed. Moving a portion of the Merino flock from the income crop stubble to the higher quality lupin stubble also reduced supplementary feed requirements. The shorter period with lambs on pasture, accompanied by increased Merino flock numbers on the lupin stubble meant that enteric methane output declined by 1.6%. Supplementary feed requirement on pasture/stubble declined by 5.6% from the baseline. This meant that despite the additional requirement to provide feedlot ration, the overall change in supplementary feed required across the farm was less than 1% higher than the baseline. Enteric methane emissions and pasture/stubble supplementary feed requirements fell by 3.6% and 6.2% in the “feedlot Merino SAMM” scenario. The additional 2% reduction in enteric methane from the “feedlot Merino” scenario reflected both the earlier weaning of SAMM lambs, which reduced the period during which ewes were lactating and thus producing higher emissions, and the finishing of SAMM lambs almost a month earlier. Despite having to provide a feedlot ration for the finishing of both Merino and SAMM lambs, supplementary feed requirements only increased by 6.6%.

Other notable changes to emission sources were lower nitrous oxide emissions from excreta and atmospheric deposition, falling by between 3.0-4.4% and 3.1-4.4%, respectively. In the “feedlot Merino” scenario, this was likely a result of the change from pasture/stubble supplementary feed with a high CP content to a feedlot ration with an average CP content of almost half, combined with the shorter period to sale. The lower supplementary feed intake by livestock on lupin stubble compared to income crop stubble in the baseline also reduced CP content intake. In the “feedlot Merino SAMM” scenario, SAMM lambs were provided a feedlot ration with a lower CP content than the combined pasture and milk diet observed in the baseline, albeit a smaller difference than the change in diet for Merino lambs. The shorter period during which ewes were lactating also reduced their CP content intake.

In both scenarios there was a demonstrated emissions benefit to feedlot finishing lambs. The time of year at which finishing occurred played a role in the carbon footprint reduction. Feedlot finishing lambs in the non-growing season for example, reduced the carbon footprint and net supplementary feed requirement across the farm, whilst allowing a portion of the breeding herd to graze lupin stubble otherwise reserved for stock destined for sale. Whilst feedlot finishing lambs during the growing season also reduced the carbon footprint, the benefit was less pronounced and required extra provision of feed ration during a period where pasture is of high quality.

7.3.3 Dongara beef production enterprise

Scenario assumptions

As described in Chapter Four, in the Dongara baseline, weaner bulls produced by the breeding herd were retained in an on-farm feedlot for finishing for sale. All other cattle destined for sale, including cull animals from the breeding herd and backgrounded animals, were finished on pasture. To examine the effect of feedlot finishing all production stock a "feedlot pellet" scenario was developed (Table 7.4). The impact of a more emissions efficient feedlot ration was explored in the "feedlot lupin" scenario.

Scenario findings

The carbon footprint of the "feedlot pellet" scenario was 8.99 kg CO_2-e/kg LW produced for sale, 1.9% lower than the baseline (Figure 7.1; Table 7.2). Accompanying this decrease was an 88% increase in supplementary feed requirement across the farm. The only supplements provided to cattle on pasture in the baseline were mineral licks, so this increase was a direct result of finishing cattle in the feedlot. Unlike the sheep enterprises, Dongara did not produce its own supplementary feed, rather it was purchased. The arable land was unsuitable for crop production so modelling the effect of producing supplementary feed was not possible.

Table 7.4 - Key assumptions of the "feedlot" scenarios at Dongara

Baseline	"Feedlot pellet" scenario	"Feedlot lupin" scenario
Weaner bulls finished in feedlot for a month following weaning and sold at 6.25 months of age in December	Per baseline	
Feedlot ration comprised of pellets and hay	Per baseline	Feedlot ration comprised of lupins and hay
All other stock (i.e. cull animals, pastoral cattle), finished on pasture	Pastoral cattle finished in feedlot for month prior to sale. Cull cattle finished on pasture. Sale dates altered to account for higher growth rates in feedlot	Per "feedlot pellet" scenario
	Same saleable liveweight production as baseline	Same saleable liveweight production as baseline

Examination of the individual emission sources revealed that whole-farm enteric methane emissions were almost 11% lower when cattle were feedlot finished, equivalent to 8.4% of the baseline (Appendix F). Nitrous oxide emissions from excreta and atmospheric deposition also declined by 7.3% and 7.2%, respectively. The lower animal emissions resulted from the shorter period to sale of all production stock. Despite this, the increased emissions associated with the production and transportation of the additional supplementary feed (58.8% and 226.5% from the

baseline, respectively) largely offset this decline in animal emissions.

As the calculated EF of the pellets was quite high, “feedlot lupins”, modelled a lupin feedlot ration (Table 7.4). The farm already purchased lupins from a neighbouring property to produce mineral licks, so it was assumed that the additional lupins were also sourced from this location. The carbon footprint of this scenario was 8.74 kg CO_2-e/kg LW, 4.6% lower than the baseline. This was more than double the reduction achieved by the “feedlot pellet” scenario. Lupins have a higher DMD to the pellets and as such 7% less was required to achieve the same sale weights. Combined with the less emissions-intensive nature of lupin production and the shorter distances to transport lupins, in the “feedlot lupin” scenario emissions associated with the production and transportation of inputs were only 33.5% and 8.3% greater than the baseline, a fraction of when pellets were supplied. The lower impact of the lupins was partially offset by the increase in nitrous oxide emissions from excreta and atmospheric deposition (3.7% and 3.6%, respectively), a result of the higher CP content of the lupins to the pellets.

Overall, feedlot finishing the production stock at Dongara reduced the carbon footprint of the enterprise regardless of the feedlot ration provided. A comparison of rations demonstrated the importance of considering the production process of purchased feed. The significant quantities of supplementary feed required to feedlot finish cattle questions the economic feasibility of applying this practice and there may be additional implications of retaining larger cattle yards. It is clear that the best approach is dependent on multiple factors, as exhibited in the two feedlot scenarios.

7.3.4 Lancelin beef production enterprise

Scenario assumptions

In the Lancelin baseline all livestock were sold directly off pasture. To model the effect of an on-farm feedlot, a “feedlot lupin” scenario was developed (Table 7.5).

Table 7.5 - Key assumptions of the "feedlot" scenario at Lancelin

Baseline	"Feedlot lupin" scenario
All livestock sold directly off pasture	Production livestock finished in feedlot for a month prior to sale. Weaner Cull animals sold directly off pasture
Weaner bulls from breeding herd weaned at end of December on pasture and sold end of January	Weaner bulls from breeding herd weaned at end of November and sold end of December
Weaner bulls from purchased heifers weaned in July and sold end of January	Per baseline
Feedlot ration comprised of pellets and hay	Feedlot ration comprised of lupins and hay
All other stock (i.e. cull animals, pastoral cattle), finished on pasture	Per "feedlot pellet" scenario
	Same saleable liveweight production as baseline

Scenario findings

Compared to the Lancelin baseline with no feedlot, the carbon footprint of the "feedlot lupin" scenario was 0.4% less, totalling 13.14 kg CO_2-e/kg LW produced for sale (Figure 7.1; Table 7.2). To accommodate livestock in the feedlot, an additional 20% of supplementary feed was required. A decline in whole-farm enteric methane emissions was largely responsible for this lower footprint, despite only a 0.7% (12,596 kg CO_2-e) decline from the baseline scenario. This decline in emissions was attributed to the shorter turnoff of weaner bulls in the feedlot and the earlier weaning of farm-produced weaner bulls, reducing the time that breeding cows spent lactating. However, more than half of this decline in enteric methane production was offset by an increase in nitrous oxide emissions from excreta and atmospheric deposition, resulting from the higher CP content of the supplied lupins (0.8% and 0.5%, respectively), along with the increased emissions associated with the production and transportation of the feedlot ration (7.1% and 5.8%, respectively). The total increase associated with these emissions sources was 8,339 kg CO_2-e, offsetting 66% of the decline in enteric methane output. However, this highlights the importance of focussing on interventions which target enteric methane, as given its role as the dominant hotspot, it will often outweigh changes to other emission sources (Beauchemin et al., 2011; Harrison et al., 2016).

Overall, whilst the introduction of a feedlot at Lancelin reduced the carbon footprint, the effect was smaller than that observed in the other farms. This was because the breeding herd was the largest contributor to the enterprise's carbon footprint (see Chapter Five). This meant that whilst feedlotting weaner bulls reduced emissions, because their contribution to overall farm emissions was small and the effect of feedlotting this stock class isolated (i.e. it did not have flow-on effects to other

components of the farm, such as at Wickepin whereby other stock can graze higher quality lupin pasture if lambs are feedlotted), the net effect on the carbon footprint was smaller than farms with a higher proportion of production livestock.

7.3.5 Implications of findings

Finishing production stock in on-farm feedlots decreased whole-farm emissions by between 0.4 and 8.3%, accompanied by lower supplementary feed requirements, for the same level of production as pasture. The effect was greatest at Bremer Bay, where the accelerated joining system saw two lambing cycles enter the feedlot annually. The high proportion of lambs to breeding flock also meant that the emission savings by the lambs in the feedlot had a considerable effect on the whole-farm carbon footprint. By contrast, the effect was the smallest at Lancelin, where, though weaner bulls were the primary product of the enterprise, the expanding breeding herd dominated the carbon footprint. Examining enteric methane emissions, the primary hotspot of both enterprises, highlights the impact of these differences; at Bremer Bay with no feedlot the breeding flock contributed 82% of total emissions while lambs contributed 18%, by contrast at Lancelin the breeding herd contributed 66% whilst the weaner bulls only contributed 5%. The emissions efficiency of feedlot finishing is thus dependent on the breeding herd structure and livestock class intended for the feedlot. This is farm-specific and must be considered when determining whether the introduction of a feedlot is an appropriate mitigation strategy for an enterprise.

A second consideration is the importance of time of the year that livestock are feedlotted. As a result of the two lambing periods at both Bremer Bay and Wickepin, for example, lambs would require feedlotting across two different months in the production year. In both instances, the emissions savings were greater when livestock were feedlotted during periods where pasture quality was poor. For example, at Wickepin, finishing the SAMM lambs in the feedlot in August when pasture was of high quality had a lower emissions benefit to finishing Merino wethers in the feedlot in December instead of lupin stubble. No reviewed study had directly examined the emissions benefit of time of feedlotting. However, Harrison et al. (2014a) examined the effect of feedlotting weaner lambs when green pasture availability fell below certain thresholds, finding that while this increased growth rates, overall the effect on EI was smaller than other examined strategies. It must be noted that in this study the duration between weaning and sale was only three weeks in December, so the period within which this strategy could be applied was narrow. They suggested that the strategy would have greater merit if it occurred over spring where removing lambs could have pasture regeneration benefits. However, based on the results of the

present study, removing livestock from pasture when it is at highest quality and quantity as opposed to when it is of poorer quality is unfeasible from a farm management perspective and in terms of emissions efficiency. In line with this, Cottle et al. (2016) examined confinement feeding livestock during the dry season when pasture availability fell below a pre-defined threshold, finding that doing so was an effective strategy to reduce EI, particularly in the extensive livestock production zones of southern Australia. They attributed this to better pasture regeneration when overgrazing was prevented, along with improved animal production.

The primary benefit of feedlot finishing livestock for sale is the higher growth rates and faster turnoff rates afforded by the improved feed conversion efficiency. The mitigation potential of practice of grain-finishing or finishing livestock in feedlots has been the focus of multiple carbon footprint and LCA studies. Though many of these studies were conducted in regions where intensive finishing of stock is common practice, such as the USA (Capper, 2011; Foley et al., 2011; Pelletier et al., 2010), Canada (Beauchemin et al., 2011), Europe (Casey & Holden, 2006; Nguyen et al., 2010; Ripoll-Bosch et al., 2013; Veysset et al., 2010) and Japan (Ogino et al., 2004; Ogino et al., 2007), some focus has also been applied in broadacre systems in southern America (Modernel et al., 2013) and Australia (Peters et al., 2010; Taylor & Eckard, 2016; Wiedemann et al., 2016a). These studies vary in their methodological approach, geographical distribution and farm characteristics. However, their findings align with the present study, that increased liveweight gains enabled by higher quality feed reduce the carbon footprint of a livestock system.

From emissions and productivity perspectives, feedlotting is a viable option. However, this study did not consider economic or other environmental implications. Ogino et al. (2016) found that in parallel to declines in EI, grain-finishing beef in Thailand increased energy consumption almost four-fold and increased acidification impacts. Feedlot finishing Australian beef, while more emissions efficient, is more energy and water intensive than pasture-finished beef (Wiedemann et al., 2016a). From an economics perspective however, grain-finishing lambs can be more profitable than pasture-finishing, driven by the shorter turnoff periods (Alcock & Hegarty, 2011b).

7.4 Crop stubble grazing

Crop stubbles can provide an alternate source of feed for livestock during the dry season. Grazing stubbles can also provide benefits to cropping systems, through weed control, manure deposition and residue management. The two sheep production enterprises examined in this study, Bremer Bay and Wickepin, used

stubble grazing as a grazing management strategy. Chapter Five found that at both enterprises, crop stubble presented the highest EI of all feedbases, a result of the low quantities of saleable liveweight produced on stubble, the higher animal emissions resulting from the poor-quality feed and ensuing high supplementary feed requirements. Overall, crop stubble contributed 18% of total emissions produced at Bremer Bay and 25% of emissions produced at Wickepin, yet produced only 4% and 13% total saleable liveweight, respectively. Of these emissions, livestock-associated emissions contributed 30% of crop stubble emissions at Bremer Bay and 71% at Wickepin. Given these findings, this Section examines the effect of the practice of stubble grazing on the whole-farm carbon footprints.

7.4.1 Bremer Bay sheep production enterprise

Scenario assumptions

Bremer Bay produced lupin and oats for the sole purpose of providing on-farm supplementary feed. The stubble of both crops was grazed by mature animals in the breeding flock. To examine the effect of stubble grazing on the carbon footprint of Bremer Bay, a "no stubble graze" scenario was developed in line with the key assumptions presented in Table 7.6. Initial analyses revealed that supplementary feed requirements were 9% lower when livestock were not grazed on stubble (Table 6.7), resulting in a 7% decline in the requirement for farm-produced oats, equivalent to five hectares less (rounded the nearest five ha) dedicated to oat crop production.

Table 7.6 - Key assumptions of the "no stubble graze" scenario at Bremer Bay

Baseline	"No stubble graze" scenario
Mature animals in the breeding herd moved from annual pasture to graze lupin and oat stubble from January through to the end of March	Mature animals in breeding herd remain on annual pasture through the production year
Weaner lambs and replacement ewes moved from annual pasture to graze kikuyu from January through to end of March	Per baseline
All stock supplementary fed oats and hay whilst grazing kikuyu or stubble from January to March	Per baseline, all stock supplementary feed oats and hay whilst grazing kikuyu or annual pasture from January to March
On-farm lupin production (40 ha) for feedlot ration	Per baseline
On-farm oat production (90 ha) for supplementary feed	Decreased oat production (85 ha) to account for 7% decline in oats requirement. The additional 5 ha allocated to pasture.
	Same saleable liveweight production as baseline

Scenario findings

Table 7.7 shows that, from an emissions perspective, there was no clear benefit to

grazing stubble over the dry season versus annual pasture. The carbon footprint of the "no stubble graze" scenario, expressed as EI, was 8.12 kg CO_2-e/kg LW produced for sale, approximately 0.7% lower. Examining the changes in annual emissions across the farm (Figure 7.2; Appendix F), in line with the lower supplementary feed requirements of the scenario, emissions associated with the production of oats (i.e. production and transportation of inputs, machinery operation) and purchased supplementary feed were lower that when stubble is grazed. As these comprise a small proportion of total farm emissions, the net effect was minor.

Whole-farm enteric methane emissions were also lower where livestock grazed annual pasture (1%), a reflection of the lower supplementary feed requirement by stock on the higher quality annual pasture. The higher CP content of annual pasture to crop stubble meant that nitrous oxide emissions from excreta and atmospheric deposition were 1.9% higher in the "no stubble graze" scenario, not enough to offset the lower emissions from the other sources. Overall, the difference between the baseline and the "no stubble graze" scenario was minimal.

Despite the minor influence of stubble grazing on the carbon footprint of Bremer Bay, almost 9% additional supplementary feed was required to maintain stock on the crop stubble, as opposed to a situation whereby stock remained on annual pasture during the dry season. That is, the same amount of saleable liveweight could be produced with less supplementary feed and a slightly lower carbon footprint if stock were retained on annual pasture. This demonstrates some of the multiple factors that could influence the farmer's decision to employ stubble grazing or not.

Table 7.7 - Influence of crop stubble grazing on total liveweight (LW) produced, supplementary feed requirement, total emissions (EQ) and emissions intensity (EI) of the two sheep production enterprises

Scenario	LW produced for sale (kg)	Supplementary feed		EQ	EI
		Pasture/crop stubble (kg)	Feedlot ration (kg)	(kg CO_2-e/ yr)	(kg CO_2-e/ kg LW)
Bremer Bay baseline	**57,975**	**81,634**	**40,738**	**474,185**	**8.18**
Bremer Bay "no stubble graze"	57,975	74,601	40,738	470,723	8.12
Difference	-	*-8.6%*	-	*-0.7%*	*-0.7%*
Wickepin baseline[a]	**108,000**	**404,050**	**-**	**1,145,073**	**10.60**
Wickepin "no stubble graze"	108,000	363,891	-	1,140,896	10.56
Difference	-	*-9.9%*	-	*-0.4%*	*-0.4%*
Wickepin "no stubble graze with added saltbush pasture"	108,000	362,461	-	1,134,285	10.50
Difference	-	*-10.3%*		*-0.9%*	*-0.9%*

[a] EQ and EI values presented for Wickepin are post-allocation between wool and meat production.

7.4.2 Wickepin sheep production enterprise

Scenario assumptions

In the Wickepin baseline, all livestock were removed from annual pasture at the beginning of January to graze crop stubble through to April or May, with young stock moved to lupin stubble and mature stock to income crop stubble. The exception was the dry Merino ewes grazing saltbush pasture from December to May. Table 7.8 outlines the assumptions of the "no stubble graze" scenario with no stubble grazing and an "added saltbush" scenario whereby farm area allocated to saltbush is doubled.

Table 7.8 - Key assumptions of the "no stubble graze" scenarios at Wickepin

Baseline	"No stubble graze" scenario	"Added saltbush " scenario
Weaner Merino lambs and maiden ewes moved from annual pasture to lupin stubble from January to March (weaner wethers sold end of January)	Weaner Merino lambs and maiden ewes remain on annual pasture through production year (weaner wethers sold end of January)	Per "no stubble graze" scenario
Mature animals from breeding flock moved from annual pasture to income crop stubble from January to April	Mature animals from the breeding flock remain on annual pasture throughout the production year	Per "no stubble graze" scenario
Merino weaners and lactating ewes with SAMM lambs on foot moved back to annual pasture in April. Remainder of flock moved back to pasture in May	N/A	N/A
Portion of dry Merino ewes grazed saltbush pasture (110 ha) from December through to May before being moved back to annual pasture	Per baseline	Increased dry Merino ewes grazed saltbush to account for added area allocated to saltbush (220 ha)
Supplementary feed comprising lupins and hay provided to all stock from January to May (December to May on saltbush)	Per baseline	Per baseline
On-farm lupin production (300 ha) for the purpose of supplementary feed	Decreased lupin production (270 ha) to account for 10% less lupin (Table 7.7). The additional 30 ha allocated to annual pasture and not income crop stubble in line with the system boundaries of the study	Per "no stubble graze" scenario
	Same saleable liveweight production as baseline	Same saleable liveweight production as baseline

Scenario findings

The difference in the carbon footprint between the Wickepin baseline and "no stubble graze" scenario was only 0.4%, with the carbon footprint of the latter 10.56 kg CO_2-e/kg LW produced for sale (Table 7.7; Figure 7.2). Emissions associated with the production and transportation of inputs, along with the operation of machinery were

lower when stubble was not grazed. This was attributed to the smaller area dedicated to lupin production and these emissions were not outweighed by the simultaneous increase in area dedicated to annual pasture production. Enteric methane was only 0.5% lower in the scenario, whilst nitrous oxide from excreta and atmospheric deposition were 3.8% and 3.9% higher, respectively. As for Bremer Bay, these differences could be explained by the DMDs and CP contents of the annual pasture and crop stubble. Overall, differences between the baseline and scenario were minor.

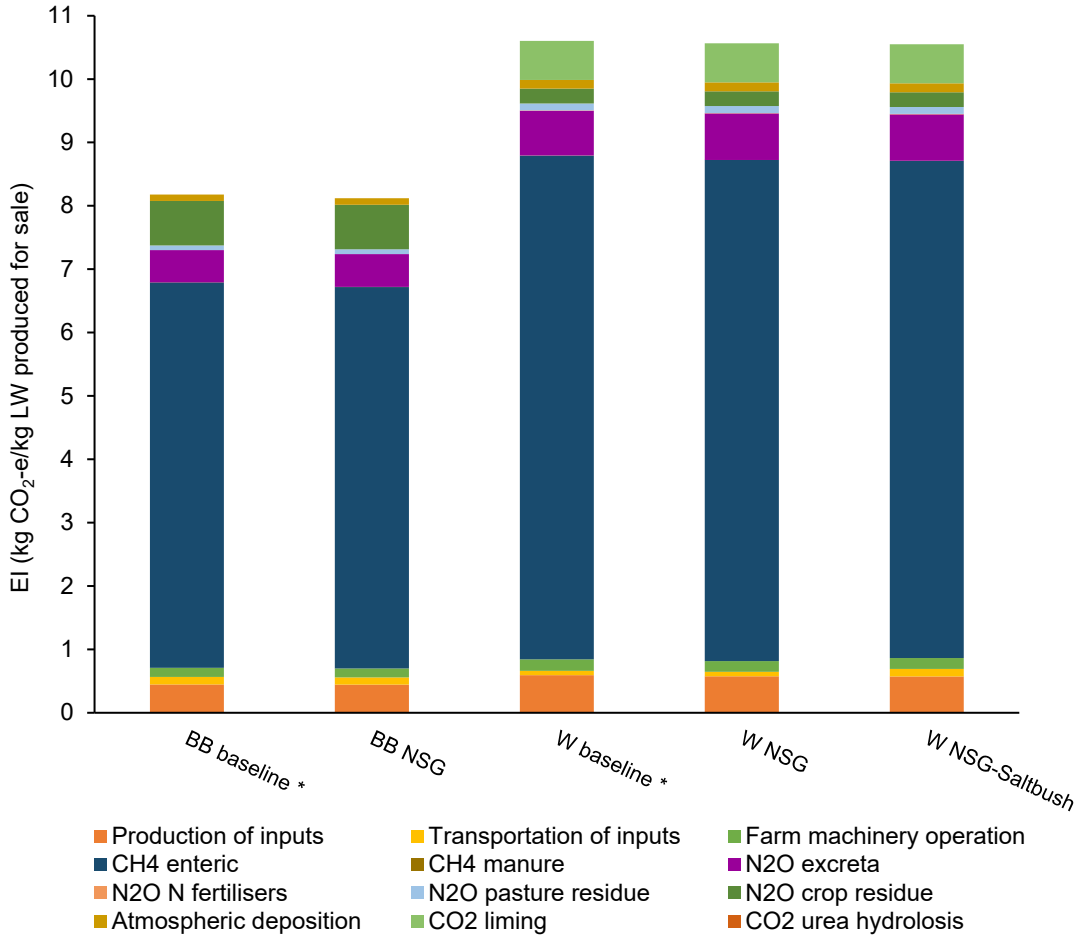


Figure 7.2 - The baseline carbon footprints, expressed as EI, of the Bremer Bay and Wickepin sheep production enterprises, as compared to the respective "no stubble grazing" scenarios. Where: Wickepin results are presented post-allocation of emissions between meat and wool. NSG = no stubble grazing; NSG-Saltbush = no stubble grazing with added saltbush.

Chapter Six found that the difference in carbon footprint between the Wickepin baseline and a scenario with no perennial pasture system was minor, a result of both the small area dedicated to saltbush and the stubble grazing employed by the farm. The carbon footprint of the "added saltbush" scenario, whereby the area dedicated to saltbush was doubled in the absence of stubble grazing, was 10.50 kg CO_2-e/kg LW (Table 7.7; Figure 7.2), 0.9% lower than the Wickepin baseline and double the

reduction achieved in the “no stubble graze” scenario. Whole-farm enteric methane emissions were 1.3% lower than the baseline whilst the supplementary feed requirement was 10.3% less. This decline in enteric methane emissions was almost three-fold the decline observed in the “no stubble graze” scenario and likely due to the higher quality of the saltbush pasture over the period grazed by the additional dry ewes, as opposed to where they were modelled to graze annual pasture for this period. The similar supplementary feed requirements of the “no stubble graze” and the “added saltbush” scenarios can be attributed to the additional month stock that were provided supplementary feed on saltbush (December) as opposed to annual pasture, which is doubled in the “added saltbush” scenario. Although the overall differences in carbon footprint are minor, it suggests that the area dedicated to saltbush could be increased whilst reducing whole-farm emissions, providing an alternative to crop stubble or even annual pasture during the dry season.

As described in Chapter Six, there could be a potential carbon sequestration benefit to increasing the farm area dedicated to saltbush production. If similar sequestration rates as those in Chapter Six are applied here, doubling the saltbush area could increase whole-farm sequestration by an 159.24 t CO_2-e/yr. Chapter Six determined that with the inclusion of modelled carbon sequestration, the EI of the baseline Wickepin scenario was 9.13 kg CO_2-e/kg LW, 13.9% less than when it was excluded. The inclusion of carbon sequestration to the “no stubble grazed” scenario would result in a revised EI of 9.09 kg CO_2-e/kg LW. Adjusting for the additional potential sequestration of the “added saltbush” scenario, the carbon footprint, expressed as EI, would be 7.55 kg CO_2-e/kg LW, 17% less than the baseline with sequestration. These calculations demonstrate that the consideration of carbon sequestration enhances the farm productivity and mitigation potential of increasing arable land dedicated to saltbush.

7.4.3 Implications of findings

The analyses conducted in this Section found no net emissions benefit of grazing crop stubble. Whole-farm carbon footprints were 0.4 to 0.7% higher when stubble was grazed as opposed to pasture, even when the pasture comprised senesced annual species of poor quality and availability. In both cases, the supplementary feed requirement was also higher when stubble was grazed. The Wickepin analysis went a step further, demonstrating that increasing saltbush production was less emissions-intensive and required less supplementary feed than when stubble was grazed or when annual pasture instead supported the dry season grazing. This benefit was realised even though the added saltbush was equivalent to only 10% of the area

dedicated to annual pasture and only supported 28% of the mature breeding flock, highlighting the potential for perennial pasture as an alternative to stubble during the dry season.

Although the literature review identified two studies which considered stubble grazing as a component within the farming system (Cottle & Cowie, 2016; Wiedemann et al., 2016c), no study has examined the effect of stubble grazing from a whole-farm emissions perspective. The findings of the present study question the viability of stubble grazing from both an emissions and productivity perspective. Though in the case of these two farms, grazing annual pasture throughout the dry season did not significantly impact the net primary productivity of the pasture, this may be an additional consideration in other farming systems. There may also be emissions or productivity consequences resulting from the removal of the co-benefits of stubble grazing (i.e. weed management, manure deposition, stubble load management) to crop production in the subsequent season. Though outside the scope of this particular study this should also be a consideration. Together, these analyses highlight the trade-offs that must be considered when farmers are making the decision whether to employ this practice.

7.5 Age distribution of the breeding herd

Waghorn and Hegarty (2011) stated that the most efficient mitigation strategies in livestock systems are those that reduce the proportion of energy expended on animal maintenance and increase the proportion directed toward animal production. Chapter Four determined that, on average, replacement breeding livestock consumed less and produced lower emissions than mature breeding livestock. Given that replacement livestock are typically able to produce offspring prior to reaching mature liveweight, this means that they did so by also producing lower net emissions. To further examine this, this Section explores the influence of reducing the average age of the breeding flock/herd by increasing the replacement rate of each enterprise.

7.5.1 Bremer Bay sheep production enterprise

Scenario assumptions

To model the effect of increasing the replacement rate at Bremer Bay, an “increased maiden ewe” scenario was developed whereby the replacement rate was doubled (Table 7.9), increasing the proportion of immature ewes in the breeding flock from 29% to 52%. Initial analyses showed that retaining more lambs meant that less lambs were finished in the feedlot prior to sale, reducing the feedlot ration requirement by 14.3% (Table 6.8). Although total intake by mature ewes declined in response to the

lower numbers, this was offset by the higher numbers of immature ewes and retained weaner ewe lambs during the period supplementary fed.

Table 7.9 - Key assumptions of the replacement rate scenario at Bremer Bay

Baseline	"Increased maiden ewe" scenario
150 Dorper ewes replaced annually	300 Dorper ewes replaced annually
On-farm lupin production (40 ha) for feedlot ration	Decreased lupin production (35 ha) to account for 14.3% decline in lupin requirement (Table 7.10). The additional 5 ha allocated to pasture.
On-farm oat production (90 ha) for supplementary feed	Per baseline as though oat requirement increased by 1%, equivalent to less than 5 ha, thus no change assumed.

Scenario findings

The carbon footprint, when expressed as EI, decreased by 9.5% to 7.40 kg CO_2-e/kg LW produced for sale when the replacement rate was doubled (Table 7.10; Figure 7.3). However, when expressed as total emissions the decrease was smaller, at 3.7%. This difference can be attributed to the increase in saleable liveweight production in the "increased maiden ewes" scenario. Though the number of lambs sold declined, the corresponding number of annual cull ewes increased, with the net result a 6.5% increase in saleable liveweight production.

Table 7.10 - Influence of breeding flock/herd replacement rate on total liveweight (LW) produced, supplementary feed requirement, total emissions (EQ) and emissions intensity (EI) of the Bremer Bay, Dongara and Lancelin livestock production enterprises

Mitigation scenario	LW produced for sale (kg)	Supplementary feed Pasture/crop stubble (kg)	Supplementary feed Feedlot ration (kg)	EQ (kg CO_2-e/ yr)	EI (kg CO_2-e/ kg LW)
Bremer Bay baseline	**57,975**	**81,634**	**40,738**	**474,185**	**8.18**
Bremer Bay increased maiden ewe	61,725	82,459	34,918	456,648	7.40
Difference	*6.5%*	*1.0%*	*-14.3%*	*-3.7%*	*-9.5%*
Dongara baseline	***442,075***	***5,200***	***117,455***	***4,051,957***	***9.17***
Dongara increased heifer	454,575	5,254	117,570	4,092,696	9.00
Difference	*2.8%*	*1.0%*	*0.1%*	*1.0%*	*-1.8%*
Lancelin baseline	***150,109***	***179,304***	**-**	***1,981,018***	***13.20***
Lancelin increased on-farm heifer	156,469	178,890	-	1,964,461	12.55
Difference	*4.2%*	*-0.2%*	-	*-0.8%*	*-4.9%*
Lancelin increased purchased heifer	187,935	180,730	-	1,964,461	10.46
Difference	*25.2%*	*0.9%*	-	*-0.8%*	*-20.8%*

The changes to individual emission sources (Appendix F), indicated a 4.0% decline in whole-farm enteric methane production, accounting for over 80% of the decline in observed emissions. This can be attributed to the change in flock structure. Chapter Four showed that across feedbases, average monthly enteric methane production by mature Dorper ewes was higher than that of replacement ewes, regardless of the physiological status (i.e. growth of replacement animals). This difference was more pronounced when mature ewes were lactating or grazing crop stubble and had higher intake requirements (Section 4.2.6, Chapter Four). As such, the decline in enteric emissions from mature ewes was 16% greater than the concurrent increase in replacement ewe emissions.

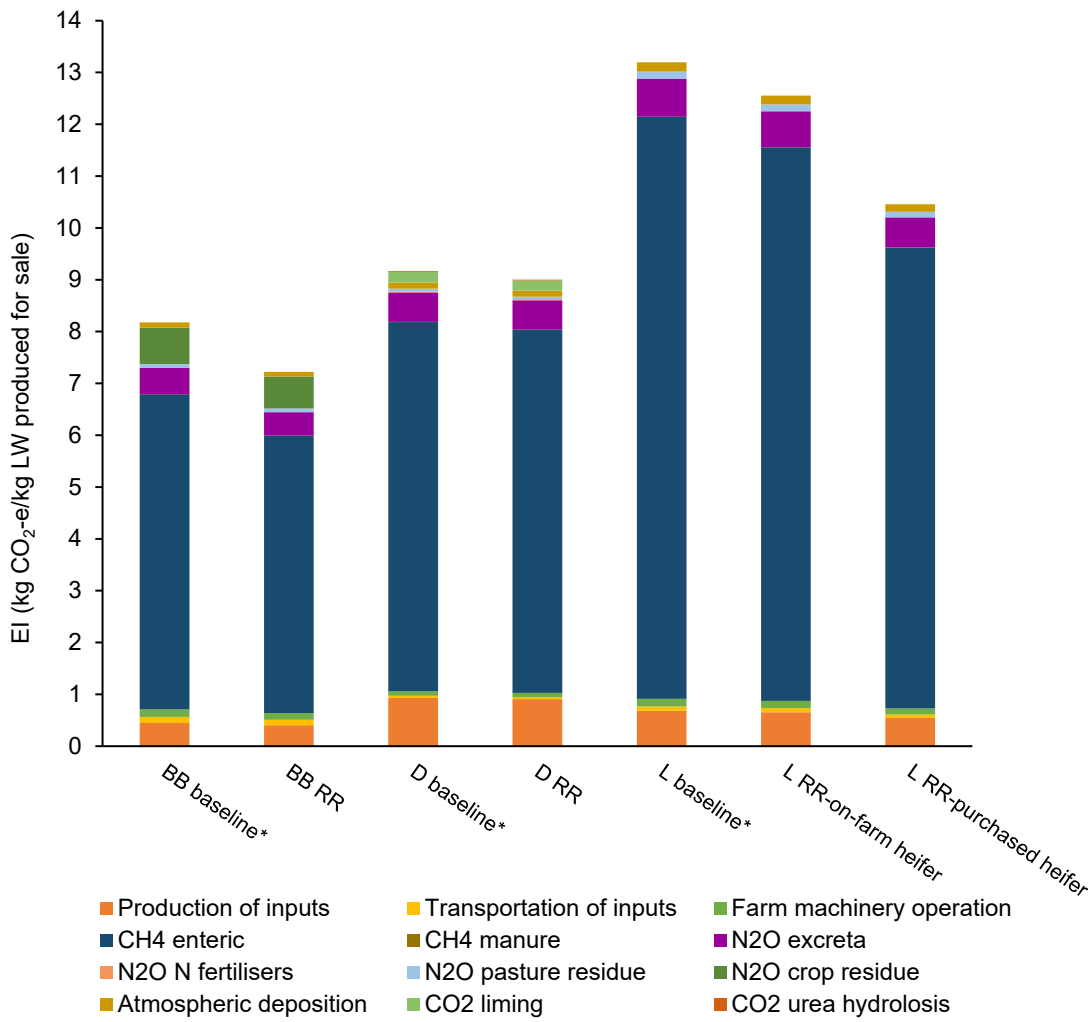


Figure 7.3 - The baseline carbon footprints, expressed as EI, of the Bremer Bay, Dongara and Lancelin livestock production enterprises, as compared to the respective "replacement rate" scenarios.

Where: RR = replacement rate.

Other notable changes to emission sources in the scenario included a 4.0% decline in total excreta nitrous oxide emissions, representing almost 7% of the decline in

whole-farm emissions. This could also be attributed to the change in flock structure, with the decline in nitrous oxide emissions from mature ewes offsetting the increase from immature ewes. A third emission source impacted by the modelled scenario was nitrous oxide emissions from crop residues, which declined by 4.9% from the baseline, accounting for 9% of the decline in whole-farm emissions. This was a direct result of the lower proportion of arable land dedicated to lupin production.

Overall, the primary benefit of increasing the replacement rate at Bremer Bay was the resultant change in breeding flock structure. This resulted in a simultaneous decline in whole-farm emissions driven by lower animal emissions and increased saleable liveweight production, with minimal change to supplementary feed requirements.

7.5.2 Wickepin sheep production enterprise

It was not possible to model increased replacement rates on the carbon footprint of the Wickepin enterprise. This is because the carbon footprint Frameworks assume that sheep reach their mature weight at two years of age. Chapter Three outlines the rationale behind this assumption further. At Wickepin, maiden ewes were first joined at 19 months of age and lambed five months later, at two years of age. Thus it was not possible to examine the effect of maintaining increased immature ewes because lambing itself was not assumed to occur until assumed mature weight.

7.5.3 Dongara beef production enterprise

Scenario assumptions

To model the effect of increasing the replacement rate, an “increased heifer” scenario was developed (Table 7.11), where the annual average proportion of heifers (one to three years of age) in the breeding herd increased from 24% to 51%.

Table 7.11 - Key assumptions of the replacement rate scenario at Dongara

Baseline	“Increased heifer” scenario
30 mature cows replaced by farm-produced 15-month heifers	80 mature cows replaced by farm-produced 15-month heifers
The remaining 50 15-month heifers sold	All heifers retained on-farm

Scenario findings

The carbon footprint of the “increased heifer” scenario was 9.00 kg CO_2-e/kg LW produced for sale, 1.8% below the baseline (Table 7.10; Figure 7.3). Total emissions, however, increased by 1%. The decrease in EI was thus attributed to the 2.8% increase in saleable liveweight production, a result of the sale of mature cows as opposed to heifers.

The breeding herd was responsible for 92% of the increase in enteric methane production, the emission source responsible for almost all of the observed increase in whole-farm emissions (Appendix F). Unlike Bremer Bay whereby maiden ewes were first joined within their first year and able to achieve multiple joinings prior to reaching mature liveweight, heifers at Dongara were first joined in their second year and calved for the first time in their third year, before achieving mature weight in their fourth year. This meant that while the number of heifers at calving age (third year) increased in the scenario, so too did the number of second-year heifers not of calf-bearing age, increasing the proportion of breeding herd on-farm producing emissions.

A breakdown of enteric methane production revealed that the decline in emissions produced by the lower number of mature cows was offset by the increase in third-year calving heifers. As described in Chapter Four, the enteric methane production of third-year dry heifers was higher than dry mature cows because of their increased intake requirements for growth. Over the production year, this meant that average enteric methane emissions between the two stock classes were similar. In addition to this, the concurrent increase in second-year heifers also increased enteric emissions, responsible for almost 97% of the increase in whole-farm enteric methane emissions. The second emission source influencing whole-farm emissions was excreta nitrous oxide, accounting for 7%. This was attributed to the higher number of non-calving heifers present on the enterprise in the modelled scenario.

Overall, whilst the EI of Dongara decreased under increased replacement rates, this was attributed to the increase in liveweight sold, rather than lower whole-farm emissions. These findings highlight the importance of reproduction strategy when assessing the suitability of this mitigation strategy. Clearly where livestock are joined later and are thus retained on-farm longer prior to first calving, the emissions advantage is less as the number of non-producing livestock is higher.

7.5.4 Lancelin beef production enterprise

Scenario assumptions

As described in Chapter Four, Lancelin was building its breeding herd. Annually, the number of retained heifers was higher than the number of sold cull cows and heifers. It also purchased heifers with calves on foot. To model the effect of increasing the replacement rate of the breeding herd, “increased on-farm heifer” and “increased purchased heifer” scenarios were developed (Table 7.12). The first scenario increased the annual average proportion of heifers (one to three years of age) in the breeding herd from 56% to 67%. The number of calves produced was lowered by

almost 7%, a reflection the lower weaning rate of heifers to mature cows at Lancelin. The second scenario also increased the annual average proportion of heifers (one to three years of age) in the breeding herd from 56% to 67%.

Table 7.12 - Key assumptions of the replacement rate scenarios at Lancelin

Baseline	"Increased on-farm heifer" scenario	"Increased purchased heifer" scenario
56 third-year heifers sold	All third-year heifers retained-on farm, 56 extra mature cows sold	Per baseline
60 purchased heifers with 53 calves on-foot	Per baseline	120 purchased heifers with 103 calves on-foot, 59 extra mature cows sold (post deaths)

Scenario findings

The carbon footprint of the "increased on-farm heifer" scenario was 12.55 kg CO_2-e/kg LW produced for sale, 4.9% lower than the baseline (Table 7.10; Figure 7.3). Whole-farm emissions decreased by less than 1%. In line with Bremer Bay and Dongara, the decrease in EI was primarily a result of increased saleable liveweight production, not lower whole-farm emissions. In this scenario, total saleable liveweight production increased by 4.2%, reflecting the increase in cull cows sold versus smaller heifers. The lower numbers of bull calves, resulting from the lower weaning rate of heifers, slightly offset this increase in liveweight sold.

The 0.9% observed decline in enteric methane emissions was responsible for most (89%) of the change in whole-farm emissions (Appendix F). The lower enteric methane output from mature cows was offset by increased emissions from the retained third-year heifers. Retaining third-year heifers rather than second-year heifers per Dongara, meant that second-year heifer numbers equalled the baseline. However, as the number of calves produced by heifers was lower, the decline in enteric emissions from calves was responsible for the majority (71%) of the observed decline in whole-farm emissions. Excreta nitrous oxide also accounted for 9% of the observed change. The rationale behind this decrease was the same as for that of enteric methane production.

The "increased purchased heifer" scenario resulted in a larger decline in EI, with a reduction of 20.8%, to 10.41 kg CO_2-e/kg LW, as compared to the baseline. As for the "increased on-farm heifer" scenario, this decline was predominantly a result of an increase in saleable liveweight production not a decline in total emissions, which reduced by only 0.8%. While cull cow sold liveweight in this scenario was similar to the "increased on-farm heifer" scenario, surplus on-farm heifers were also sold in line

with the baseline, along with additional bull calves and heifers produced by the purchased heifers. This meant that total liveweight sold was 25% higher than the baseline and 20% higher than the “increased on-farm heifer” scenario.

Enteric methane and excreta nitrous oxide emissions were responsible for 93% and 6% of the decline in whole-farm emissions, respectively (Appendix F). These declines were greater than in the “increased on-farm heifer” scenario, with enteric methane and excreta nitrous oxide emissions falling by 0.9% each, as opposed to 0.9 and 1.3% where on-farm heifers were retained. Lower emissions resulting from lower numbers of mature cows and farm-produced calves offset the additional emissions produced by retained purchased heifers and accompanying calves. The purchase of heifers had the benefit of additional liveweight without having to account for the additional emissions associated if heifers were produced by the breeding herd.

Overall, whilst increasing the replacement rate at Lancelin reduced whole-farm EI, this reduction was mostly attributed to the increases saleable liveweight production not a decline in whole-farm emissions. When retaining on-farm heifers, this increase resulted from the increased sold liveweight of cull cows. The purchase of additional heifers for replacement however, also increased calves sold and maintained the sale of other on-farm breeding herd stock classes. The choice of replacement stock also played a role; replacing older heifers does not require a concurrent increase in younger heifers of non-reproductive age, avoiding increased emissions from this non-producing stock class.

7.5.5 Implications of findings

Altering the age distribution of the breeding flock/herd through an increase in the replacement rate reduced the EI of the enterprises by between 2 and -21%. The concurrent change in total emissions, however, ranged from a 4% decrease to a 1% increase. As such, changes to the EI reflected increased saleable liveweight production (3 to 25%) rather than lower whole-farm emissions. The additional saleable liveweight resulted from the increased sale of mature breeding sheep/cows, rather than younger breeding stock classes, such as heifers at Dongara and Lancelin, or weaners previously destined for sale, such as at Bremer Bay.

The success of this strategy relates to the FU adopted to measure emissions efficiency, with liveweight aggregated across stock classes in the present study. By contrast, Dougherty et al. (2018), in their LCA of Californian sheep production, found that increasing replacement rates, assuming a steady-state herd, from 15% to 35% increased EI between approximately 14-35%. This compares to the steady-state

farms Bremer Bay and Dongara which yielded EI declines despite increases in the replacement rate of between 19%-38% and 13%-35%, respectively. A critical difference to Dougherty et al. (2018) was that the FU of that study was kg of market lamb produced, meaning that only the decreased quantity of sold lambs was considered, not the coinciding increase in cull ewe sales. Browne et al. (2015) found that decreasing the replacement rate of dairy cows through increased cow lactation periods lowered EI. However, the FU of this study was kg milk fat and protein and so there was no net benefit if replacement rate was increased. The study also assumed that milk production remained constant under extended lactation, unlike Wall et al. (2012) which determined that EI actually increased under extended lactation due to the lower milk production attainable in later lactation. The assumptions made, along with the FU considered, clearly play a role. Economic consideration of shifts in product output could be worthwhile in such situations, for example, the trade-off between greater quantities of lower value mutton to lesser declines in higher value lamb.

The effectiveness increased replacement rates varied across the enterprises, highlighting the differences in farm practices. For example, the EI at Dongara declined by less than 2%, whereas at Lancelin it was almost 5% lower when on-farm heifers were retained and 21% lower with increased purchased heifers. Differences in the role of the breeding herd influenced these results; for example at Dongara the breeding herd was only responsible for 12% of total saleable liveweight production and 19% of whole-farm enteric methane production, while the breeding herd at Lancelin was responsible for 72% and 76%, respectively. Thus, a change to the breeding herd will have a greater impact on the EI at Lancelin as opposed to Dongara. The impact was amplified with increased purchased heifers at Lancelin, as emissions associated with raising these heifers and calves prior to entering the farm were excluded. Caution must be made when interpreting the 21% EI decline, as the emissions burden of producing these heifers is entirely transferred to the farm prior instead of being distributed across both accordingly. Data constraints prevented accounting for this distribution. Other studies have made the same assumption and observed similar differences in EI (Bell et al., 2012a; Peters et al., 2010). Peters et al. (2010) observed a 29% EI increase following the transition from a finishing operation for purchased weaners to one which included on-farm breeding activities. Similarly, Lieffering et al. (2016) found that replacing mature cows with once-bred heifers derived from the dairy sector was effective at reducing the national EI of beef production. However, it must be noted that this scenario assumes no difference in meat quality between cows produced by dairy as opposed to beef cattle. Some

studies have included emissions from the production of purchased livestock using national or regional estimates (Wiedemann et al., 2016a), adopted a single EF regardless of the source of the purchased livestock (Jones et al., 2014) or do not provide sufficient information to ascertain the source and detail of this emissions information (Eady et al., 2011). Whilst it is preferable to include these emissions, a streamlined approach is required to ensure accurate and consistent reporting.

Another reason for the observed differences between enterprises was the age of calving/lambing, which influenced the quantity of emissions produced by replacement stock prior to producing offspring. At Dongara for example, heifers calved at 24 months, thus remaining on-farm for two years prior to contributing to saleable liveweight production. By contrast at Bremer Bay maiden ewes first lambed at 18 months. This would have also influenced the difference observed between Dougherty et al. (2018), where maiden ewes were not joined until 18 months of age and/or presented lower weaning rates. Replacement ewes studied by Dougherty et al. (2010) also produced higher emissions than mature ewes, compared to the enterprises in the present study where, on average, emissions were lower for replacement stock. Together, these acted to increase the EI under increased replacement rates.

Changing the herd/flock structure will impact the EI of an enterprise. Whilst increasing the proportion of immature to mature breeding flock has the potential to reduce emissions and increase saleable liveweight production, the net effect on EI will be dependent on the FU employed and farm characteristics such as joining and calving/lambing age along with whether the herd/flock is self-replacing or purchased.

7.6 Increasing reproductive performance

Increasing the reproductive performance of a breeding flock/herd transitions the balance from breeding stock to production stock. This increases the proportion of liveweight on the farm destined for sale and reduces the proportion of non-producing breeding animals producing emissions. The carbon footprint of Bremer Bay was 11% to 38% lower than the other examined enterprises (Chapter Five). A standout characteristic of this enterprise was that it employed an accelerated joining system, with two lambing per year. This meant on an annual basis, the number of lambs to breeding ewes was 1.5:1. This can be compared to Wickepin, where the ratio was 0.7:1, while the number of calves to breeding cows and heifers at Dongara and Lancelin was 0.7:1 and 0.6:1, respectively. It was hypothesised that these differences played a role in the exhibited differences in the carbon footprint between enterprises. This Section explores the impact of increased reproductive performance, specifically

weaning rate, on the carbon footprint of the four enterprises.

7.6.1 Bremer Bay sheep production enterprise

Scenario assumptions

Bremer Bay was a high fecundity enterprise, the accelerated joining system enabled an annual weaning rate of 150% (75% per joining). Previous studies found this weaning rate optimal for Dorpers in accelerated joining systems (Chadwick & Pearce, 2013). As the other examined enterprises could not achieve an accelerated mating system or such weaning rates, the impact of this management strategy on its carbon footprint was examined.

Four scenarios were modelled to analyse the influence of weaning rate at Bremer Bay (Table 7.13). The first two examined the carbon footprint of an annual joining system with a weaning rate equivalent to the baseline (75%); firstly, with the same breeding herd structure as the baseline, “weaning rate 75%”, and secondly, with the breeding herd structure adjusted to produce the same annual saleable liveweight as the baseline (57,975 kg), “weaning rate 75%-match LW”. The rationale behind this second scenario was to match the liveweight sold in the baseline and ascertain the emissions burden of maintaining the same level of productivity with reduced weaning rates. The second two scenarios modelled an annual joining system with a more representative weaning rate. Based on a review of the reproductive performance of Dorper ewes in annual joining systems in southern Australia, the adjusted weaning rate was assumed to be 120% per ewe mated (Chadwick & Pearce, 2013), and the modelled scenarios “weaning rate 120%” and “weaning rated 120%-match LW”.

Scenario findings

The first scenario, “weaning rate 75%”, resulted in a carbon footprint, expressed as EI, of 12.15 kg CO_2-e/kg LW produced for sale (Table 7.14; Figure 7.4), almost double the baseline. This higher EI was attributed to an almost 50% reduction in total liveweight sold; with only one joining and the same number of replacement maiden ewes, the number of lambs sold was 57% lower. Whole-farm emissions were also lower (20.7%), with a reduction in enteric methane (17.1%; Appendix F) responsible for over 60% of this difference.

Table 7.13 - Key assumptions of the weaning rate scenarios at Bremer Bay

Baseline	"Weaning rate 75%" & "weaning rate 75%- match LW" scenarios	"Weaning rate 120%" & "weaning rate 120%- match LW" scenarios
Accelerated joining system for an annual weaning rate of 150% (75% per joining)	Annual joining system with an annual weaning rate of 75%	Annual joining system with an annual weaning rate of 120%
Lambing in March and September	Lambing in July	Lambing in July
Lambs sold at 5.5 months and 45 kg	Per baseline	Per baseline
On-farm lupin production (40 ha)	*"Weaning rate 75%":* Decreased lupin production (20 ha) to account for lower feedlot ration requirement (Table 7.14). Additional 20 ha allocated to pasture *"Weaning rate 75%-match LW":* Per baseline	*"Weaning rate 120%":* Decreased lupin production (35 ha) to account for lower feedlot ration requirement. Additional 5 ha allocated to pasture *"Weaning rate 120%-match LW":* Per baseline
On-farm oat production (90 ha)	*"Weaning rate 75%":* Decreased oat production (35 ha) to account for lower supp. feed requirement. Additional 55 ha allocated to pasture *"Weaning rate 75%-match LW":* Decreased oat production (60 ha) to account for lower supp. feed requirement. Additional 30 ha allocated to pasture	*"Weaning rate 120%":* Decreased oat production (40 ha) to account for lower supp. feed requirement. Additional 50 ha allocated to pasture *"Weaning rate 120%-match LW":* Decreased oat production (50 ha) to account for lower supp. feed requirement. Additional 30 ha allocated to pasture
	"Weaning rate 75%": Saleable liveweight production per 75% weaning rate with same herd structure as baseline *"Weaning rate 75%-match LW":* Herd structure adjusted to achieve saleable liveweight production per baseline	*"Weaning rate 120%":* Saleable liveweight production per 120% weaning rate with same herd structure as baseline *"Weaning rate 120%-match LW":* Herd structure adjusted to achieve saleable liveweight production per baseline

Though a portion of the change in enteric methane emissions was due to less lambs, a large portion was attributed to the change in the average annual physiological status of mature breeding ewes. In the baseline, dry and lactating mature ewes contributed 28% and 33% of total enteric methane emissions, respectively. In the "weaning rate 75%" scenario, dry mature ewes contributed 46% while lactating mature ewes contributed 20%. The transition to a higher proportion of dry ewes saw a net decrease in enteric methane. This was amplified by the removal of the March lambing period. In the Bremer Bay baseline, lactating ewes had to be supplementary fed for the first two months of the March lambing, increasing emissions. With a July lambing this was not required. The second emission source responsible for the lower emissions was nitrous oxide emissions from crop residue, which was 59.4% lower than the baseline and responsible for almost 25% of the difference in whole-farm emissions. This was

driven by lower requirements for supplementary feed on pasture/stubble (65.7% less), as described above, along with lower ration quantities (55% less) in the feedlot.

Table 7.14 - Influence of ewe/cow weaning rate on total liveweight (LW) produced, supplementary feed requirement, total emissions (EQ) and emissions intensity (EI) of the livestock production enterprises

Scenario	LW produced for sale (kg)	Supplementary feed Pasture/crop stubble (kg)	Supplementary feed Feedlot ration (kg)	EQ (kg CO_2-e/ yr)	EI (kg CO_2-e/ kg LW)
Bremer Bay baseline[a]	**57,975**	**81,634**	**40,738**	**474,185**	**8.18**
Bremer Bay "weaning rate 75%"[b]	30,975	28,001	18,347	376,249	12.15
Difference	*-46.6%*	*-65.7%*	*-55.0%*	*-20.7%*	*48.5%*
Bremer Bay "weaning rate 75%- match LW"	57,975	44,667	42,809	730,348	12.60
Difference	*-*	*-45.3%*	*5.1%*	*54%*	*54.0%*
Bremer Bay "weaning rate 120%"[c]	47,175	29,300	33,024	405,106	8.59
Difference	*-18.6%*	*-64.0%*	*-18.9%*	*-14.6%*	*5.0%*
Bremer Bay "weaning rate 120% - match LW"	57,975	34,720	42,809	499,352	8.61
Difference	*-*	*-57.5%*	*5.1%*	*5.3%*	*5.3%*
Wickepin baseline[d]	**108,000**	**404,050**	**-**	**1,145,073**	**10.60**
Wickepin weaning rate Merino	111,478	408,144	-	1,159,779	10.40
Difference	*3.2%*	*1.1%*	*-*	*1.3%*	*-1.9%*
Wickepin weaning rate SAMM	109,935	407,257	-	1,154,832	10.50
Difference	*1.8%*	*0.8%*	*-*	*0.9%*	*-0.9%*
Wickepin weaning rate Merino SAMM	113,413	411,351	-	1,168,460	10.30
Difference	*5.0%*	*1.8%*	*-*	*2.0%*	*-2.8%*
Dongara baseline	***442,075***	***5,200***	***117,455***	***4,051,957***	***9.17***
Dongara weaning rate	444,900	5180	118,603	4,063,044	9.13
Difference	*0.6%*	*-0.4%*	*1.0%*	*0.3%*	*-0.4%*
Lancelin baseline	***150,109***	***179,304***	***-***	***1,981,018***	***13.20***
Lancelin weaning rate	155,378	180,382	-	1,998,849	12.86
Difference	*3.5%*	*0.6%*	*-*	*0.9%*	*-2.6%*

[a] Bremer baseline incorporates an accelerated joining system with two lambings each year for an annual weaning rate of 150% (75% per lambing).
[b] Modelling an annual joining system at the same weaning rate (75% per lambing) as the baseline.
[c] Modelling an annual joining system using a weaning rate (120%) reflective of Dorpers in annual joining systems (Chadwick & Pearce, 2013).
[d] Wickepin values are post-allocation between meat and wool. Allocation factors differ between the scenarios in line with sold LW variations.

If instead, Bremer Bay was modelled to produce of similar amounts of sold liveweight as the baseline, the carbon footprint would increase to 12.60 kg CO_2-e/kg LW. In the

"weaning rate 75%-match LW" scenario, to achieve this at a weaning rate of 75%, the breeding herd would have to double from 800 to 1600, more than doubling whole-farm emissions. More than 94% of this difference can be attributed to the increase in enteric emissions from the adjusted herd structure, specifically the larger breeding herd. Despite the increase in stock numbers, the supplementary feed requirement on pasture/stubble was 45.3% lower than the baseline, attributed to no March lambing. This follows the findings of Chapter Four and Five which found that due to the poorer feedbase quality and higher supplementary feed requirements over the March lambing, the emissions burden was higher than the September lambing. Despite now lambing in the more optimal July period in terms of pasture availability and quality, the low weaning rate makes this strategy far less feasible, both in terms of the practicality of running a larger flock and emissions output.

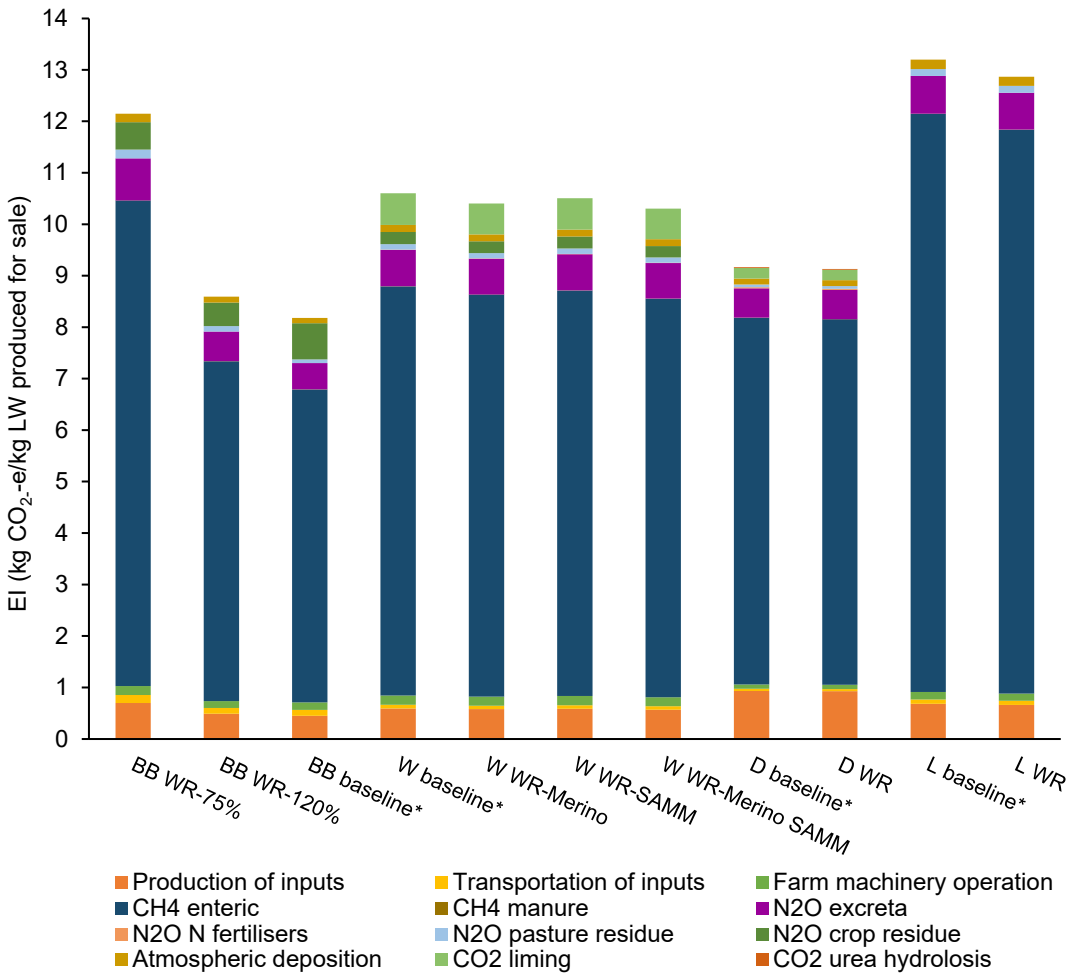


Figure 7.4 - The baseline carbon footprints, expressed as EI, of the four livestock production enterprises, as compared to the respective "increased weaning rate" scenarios.

Where: Wickepin results presented post-allocation of emissions between meat and wool. WR= weaning rate.

The third scenario, “weaning rate 120%” presented a carbon footprint of 8.59 kg CO_2-e/kg LW, only 5% higher than the baseline. However, total saleable liveweight was 18.6% lower, reflecting that although a higher weaning rate of 120% was applied, this fell short of the 150% possible under accelerated joining. Consequently, 810 lambs could be sold compared to 1,050 in the baseline and 450 in the “weaning rate 75%” scenario. The lower weaning rate also had implication for whole-farm emissions, which were 14.6% lower than the baseline. As for the first scenario, this decline resulted primarily from the transition to a higher proportion of dry ewes throughout the production year, along with a July lambing on high quality pasture. In total, enteric methane was 11.7% lower than the baseline, accounting for 60% of the difference in total whole-farm emissions. Excreta nitrous oxide emissions followed a similar, but smaller trend, declining by 8.3% from the baseline, whilst crop residue emissions were 47% lower. This was a smaller difference than in the “weaning rate 75%” scenario, reflecting the higher feedlot ration requirement of the “weaning rate 120%” scenario to finish the higher number of lambs (though still 18.9% lower than the baseline).

The carbon footprint, if Bremer Bay adjusted its practices and flock structure to produce the same saleable liveweight as the baseline, would be 8.61 kg CO_2-e/kg LW, or 5.3% higher. In this “weaning rate 120%-match LW” scenario, the farm would have to increase its breeding herd by 200 under the 120% weaning rate. This increase in stock numbers resulted in a concurrent 5.3% increase in total annual emissions, largely from the increased animal emissions associated with the larger flock. Once again, supplementary feed requirements were lower than the accelerated joining of the baseline, reflective of the change in lambing month.

Overall, these scenarios demonstrate that annual joining at Bremer Bay, even with optimal weaning rates, produces a higher carbon footprint compared to an accelerated joining system. The July lambing, rather than March and September, reduced the supplementary feed requirement of the Dorper flock, reflective of the input-intensive March lambing period, whereby pasture and crop stubble quality was low. The findings highlight the role of time of lambing along with weaning rates. Despite the reduced emissions and supplementary feed requirements associated with the July lambing, this was offset by the requirement for a larger breeding flock to compensate for the lower weaning rates. The net result of the annual joining, whereby the system is adjusted to match liveweight of the baseline, was an increase in the proportion of breeding flock to production flock, increasing the maintenance emissions of the flock and reducing emissions efficiency of the system.

7.6.2 Wickepin sheep production enterprise

Scenario assumptions

In the Wickepin baseline, the weaning rate of the Merino lambs was 85% per ewe mated while for SAMMs the rate was 92.5%.

To model the effect of increased weaning rates on the carbon footprint of Wickepin, three scenarios were modelled. The first, "weaning rate Merino", modelled a 5% increase in the weaning rate of the Merino lambs only. The second scenario, "weaning rate SAMM", modelled a 5% increase in the weaning rate of the SAMM lambs only. The final scenario "weaning rate Merino SAMM" modelled a 5% increase for each breed. This enabled examination of the role of each breed as well as the combined effect on the farm system.

Table 7.15 - Key assumptions of the weaning rate scenarios at Wickepin

Baseline	"Weaning rate Merino" scenario	"Weaning rate SAMM" scenario	"Weaning rate Merino SAMM" scenario
Merino annual joining system with an annual weaning rate of 85%	Merino weaning rate of 90%	Per baseline	Per "weaning rate Merino" scenario
SAMM annual joining system with an annual weaning rate of 92.50%	Per baseline	SAMM weaning rate of 97.5%	Per "weaning rate SAMM" scenario
Merino lambs sold at 8 months at 37 kg	Per baseline	Per baseline	Per baseline
SAMM lambs sold at 6 months at 45 kg	Per baseline	Per baseline	Per baseline
On-farm lupin production (40 ha)	Per baseline as only 1.1% decline in supp. feed requirement, equivalent to less than 5 ha, thus no assumed change	Per baseline as only 0.8% decline in supp. feed requirement, equivalent to less than 5 ha, thus no assumed change	Increased lupin production (45 ha) to account for 1.8% increase in supp. feed requirement. The 5 ha obtained from pasture

Scenario findings

An increase in the weaning rate of Merino lambs reduced the carbon footprint to 10.40 kg CO_2-e/kg LW produced for sale (Table 7.14; Figure 7.4). This was 1.9% lower than the baseline, however it also reflected a protein mass allocation factor of 62% as opposed to the baseline factor of 61.6%. The greater allocation of total emissions to meat resulted from the 3.2% increase in saleable liveweight from the increase in Merino lambs numbers by 94 to 1694 under the higher weaning rate. This increase in liveweight occurred concurrently with a 1.3% increase in whole-farm annual emissions (post-allocation). Over 90% of the change in emissions was attributed to an increase in enteric methane output, which in turn, resulted from an increase in the

proportion of lactating ewes over the production year, along with the resultant increase in unweaned Merino lambs and weaner wethers prior to sale. The increase in enteric methane emissions from unweaned Merino lambs on annual pasture over six months was almost equal to the increase in emissions from the weaner wethers finished on lupin stubble over the final two months prior to sale. This highlights the roles of decisions pertaining to different stock classes and grazing management across the production year, on emissions. This was highlighted in Chapters Four and Five.

Alternatively, when the effect of increasing the weaning rate of SAMM lambs was examined, the reduction in carbon footprint was smaller, less than 1% at 10.50 kg CO_2-e/kg LW. This EI also reflected a revised protein mass allocation factor of 61.9% following the increase in liveweight produced. In this scenario, the increase in weaning rate resulted in an additional 43 lambs, for a total annual production of 843 SAMM lambs. The net effect in terms of saleable liveweight was a 1.8% increase, along with a 0.9% increase in whole-farm annual emissions. As for the "weaning rate Merino" scenario, this increase in emissions was largely explained by the increase in the annual proportion of lactating ewes, along with the additional SAMM lambs.

To compare the respective performance the two scenarios, the increase in total emissions per each additional kg of saleable liveweight was calculated. Using unallocated emissions data, an increase in the weaning rate of SAMM lambs was more efficient, from both productivity and emissions perspectives, as compared to an increase in the weaning rate of Merino lambs. With a 5% increase in weaning rate, the "weaning rate SAMM" scenario produced 3.03 kg CO_2-e/kg additional saleable LW , whilst the "weaning rate Merino" produced an additional 3.44 kg CO_2-e/kg LW. This aligns with the findings of Chapters Four and Five that whilst emissions associated with the production of a SAMM lamb were higher than a Merino lamb, per kg of saleable liveweight produced, SAMMs were more efficient. This is due to the faster turnoff, higher sale weight and lower supplementary feed requirements of SAMMs. However, if instead the post-allocation emissions data of each scenario was applied, the results were reversed, resulting from the respective protein mass allocation factor applied to each scenario. Merino lambs were shorn prior to sale, whereas SAMM lambs were sold prior to shearing. Hence the protein mass allocation factor of the "weaning rate SAMM" scenario was skewed towards the allocation of emissions to meat. This demonstrates the influence of allocation decisions, not only on the results of a carbon footprint, but also choice of mitigation strategy. When allocation has been applied results must be interpreted with caution.

The final, "weaning rate Merino SAMM", scenario combined the improved weaning

rates of both the Merino and SAMM lambs, finding that applied together, the carbon footprint declined by 2.8% to 10.30 kg CO_2-e/kg LW. In this scenario, saleable liveweight production was simply the total of the previous scenarios, whilst the 2% increase in whole-farm emissions was driven by changes to the same emissions sources as the previous scenarios. The key differences were that in this scenario, the combined increased requirement for supplementary feed on pasture and crop stubble by ewes at the beginning of SAMM lambing in April and by weaner Merino wethers on lupin stubble in January, resulted in a change of arable area dedicated to farm-grown supplementary feed.

Overall, all three scenarios demonstrated that an increase in the weaning rate of either Merino or SAMM lambs could reduce the carbon footprint at Wickepin, through an increase in saleable liveweight production. However, the greatest impact was when the strategy was applied across both breeds. This demonstrates that breed and management decisions applied to the production of lambs will influence the emissions efficiency of increasing reproductive performance, whilst allocation factors must be considered carefully when analysing results.

7.6.3 Dongara beef production enterprise

Scenario assumptions

In the Dongara baseline, the weaning rate of the breeding herd was calculated to be 80% calves weaned per cow mated. To examine the influence of increased weaning rates, a “weaning rate” scenario was modelled where weaning rate increased to 85%.

Scenario findings

The carbon footprint of the “weaning rate” scenario was 9.13 kg CO_2-e/kg LW produced for sale, 0.4% lower than the baseline (Table 6.10; Figure 6.6). This small decline reflected the lower contribution of the breeding herd versus the pastoral herd to whole-farm emissions and annual saleable liveweight production. In the baseline, the breeding herd contributed 12% of total saleable liveweight. When the increased weaning rate was applied, the number of calves produced by the breeding herd increased by ten, to 160 calves. All ten calves were assumed to be sold, to maintain the same herd structure as the baseline, resulting in a net increase in annual saleable liveweight of only 0.6%.

Similarly, whole-farm emissions were only 0.3% higher than the baseline. This increase was largely explained by the increased animal emissions associated with the higher proportion of lactating cows and higher calf numbers. The increase in enteric methane and excreta nitrous oxide accounted for 88% and 6% of the

difference, respectively (Appendix F). As described in Chapter Five, enteric methane production was the dominant hotspot at Dongara. Despite this, as the breeding herd only contributed 19% to whole-farm enteric methane emissions, the observed increase in enteric methane production in the scenario only increased this contribution by 0.3%. Once again, despite the changes to total emissions, the small contribution of the breeding herd meant that this had a lesser effect on whole-farm emissions.

Overall, while an increase in weaning rates reduced the carbon footprint of Dongara, the effectiveness of this strategy was lower than other enterprises, a result of the small contribution of the breeding herd to total herd productivity and emissions output.

7.6.4 Lancelin beef production enterprise

Scenario assumptions

In the Lancelin baseline, the weaning rates of calves produced by the mature breeding herd, breeding herd heifers and purchased heifers were calculated using stock counts and sale numbers. The weaning rate of calves produced by the mature breeding herd was calculated to be 92.2% calves per mature cow mated. The rates for breeding herd heifers and purchased heifers were 72.2% and 88.3%, respectively. To model increased reproductive performance at Lancelin a “weaning rate” scenario was developed, whereby the weaning rate of calves produced by all cows and heifers was increased by 5%. This increased the rates to 97.2%, 77.2% and 93.3% for mature breeding cows, on-farm heifers and purchased heifers, respectively.

Scenario findings

The carbon footprint of the “weaning rate” scenario was 12.86 kg CO_2-e/kg LW produced for sale, 2.6% lower than the baseline (Table 6.10; Figure 6.6). In line with the other modelled weaning rate scenarios, this decline was a result of the increased productivity accompanying the strategy, with annual saleable liveweight production increasing by 3.5%. The increased weaning rates across the breeding herd and purchased heifers increased annual calf production by 23 to 415 calves. This increase resulted in a concurrent increase (0.9%) in whole-farm emissions, of which enteric methane and excreta nitrous oxide emissions contributed 90% and 7%, respectively (Appendix F). The increased numbers of farm-produced calves accounted for over 60% of this difference in animal emissions from the baseline, with the increase in calves produced by purchased heifers accounting for a further 15%.

Overall, the modelled scenario demonstrated that increased weaning rates can reduce the carbon footprint at Lancelin. Compared to Dongara, the breeding herd at Lancelin accounted for 65% of total saleable liveweight produced and this increased

contribution is reflected in the greater impact on the modelled carbon footprint.

7.6.5 Implications of findings

Increasing weaning rates successfully reduced the EI of all the enterprises examined. As for the previous strategies examined, the scale of this reduction was dependent on the characteristics of the respective farms. For example, the increased weaning rates afforded by the accelerated joining system at Bremer Bay meant that it was more emissions efficient than even the most optimal weaning rates under annual joining, regardless of the fact that the annual lambing occurred in July under optimal pasture conditions whereas the under the accelerated system, lambing occurred outside of this optimal window and required large quantities of supplementary feed. This highlights that under certain conditions, such as increased weaning rates, the benefits of increased lamb output outweigh the benefits of time of lambing, another recommended mitigation strategy. By contrast at Wickepin, time of lambing and breed were found to influence the effectiveness of increased weaning rates on the carbon footprint. Like the previously examined strategies, the respective contribution of the breeding herd to saleable liveweight production will determine the impact of increased weaning rates, with the EI decline more pronounced at Lancelin with the larger breeding herd.

The primary benefit of increasing weaning rates is the corresponding increase in saleable liveweight production resulting from the production of more livestock destined for sale. From an emissions perspective, though whole-farm emissions increased (0.4 to 14.6% in the examined enterprises), this increase predominantly represented an increase in animal emissions directly attributable to the production of additional saleable liveweight. The ensuing shift in emissions away from maintenance (i.e. unproductive breeding animals) to production (i.e. lactating animals and offspring) is the common outcome of all reproductive strategies found to have mitigation potential, including weaning rate (Alcock & Hegarty, 2011a; Alcock et al., 2015; Beauchemin et al., 2011; Becoña et al., 2014; Cullen et al., 2016; Dick et al., 2015b; Jones et al., 2014; Nieto et al., 2018; Veysset et al., 2014; Wiedemann et al., 2015c), fecundity (Cruickshank et al., 2008; Harrison et al., 2016; Harrison et al., 2014b; Ho et al., 2014; Toro-Mujica et al., 2017) and earlier joining (Alcock et al., 2015; Cruickshank et al., 2008; Cullen et al., 2016; Harrison et al., 2014a; Harrison et al., 2016). Wiedemann et al. (2015c) found that weaning rates explained 37% of the EI variation of beef production in eastern Australia, while the number of lambs per ewe was found to be the most important predictor of the carbon footprint of sheep production in the UK , explaining 27.4% (Jones et al., 2014). Only one study (Cottle

et al., 2016), found no effect of reproductive efficiency on EI, contradicting the findings of other reviewed literature and the present study. Cottle et al. (2016), however, presented the aggregated results of three different farming systems over 28 locations, which likely masked the effects of the individual climatic zones and farming systems. Once again, this highlights the importance of farm-level analyses when undertaking mitigation analyses.

7.7 Conclusion

This Chapter examined the mitigation potential of five identified farming practices. Three of these were associated with improving grazing management; namely the establishment of perennials, feedlot finishing of livestock and stubble grazing. The remaining two were associated with improved reproductive efficiencies, altering the herd structure through replacement rates and increased weaning rates. Each of the examined practices, with the exception of stubble grazing, reduced the EI of the enterprises, but with varying degrees of effectiveness across both strategy type and livestock enterprise, ranging from a reduction of only 0.4% to as much as 20.8%. It was not anticipated that stubble grazing would have mitigation potential, but rather was examined as it is promoted and practiced as a mutually beneficial practice for both livestock production and crop stubble management. The results demonstrated no concurrent emissions benefit of this practice and in some circumstances, requires greater quantities of supplementary feed than grazing senesced annual pasture. The specific findings and interpretations of each examined strategy are discussed in the respective Sections of this Chapter.

There was no one strategy which presented as most or least effective across the considered livestock production systems. Instead, the mitigation potential of a selected strategy was dependent on the characteristics of the farm in question. For example, the effectiveness of the strategies at reducing the EI of the Bremer Bay sheep enterprise, from most to least, were changes to replacement rate, feedlot finishing, increased weaning rate, perennial pasture and crop stubble grazing. By contrast, at the Lancelin beef enterprise the order was perennial pasture, change in replacement rate, weaning rate and feedlot finishing. Such differences highlight the importance of farm-specific analyses when assessing potential mitigation strategies.

Farms even within the same region will vary considerably in their characteristics, specific management practices and ensuing productivity (Hyland et al., 2016; Jones et al., 2014; Toro-Mujica et al., 2017; Veysset et al., 2014). Aggregating such data can mask the true impact of a mitigation strategy (Cottle et al., 2016). Similarly, the

findings of specific case studies also encounter difficulties in terms of transferability to other farms with different characteristics. Despite this, the in-depth farm-specific analyses conducted in the present study have demonstrated the mitigation potential of several strategies across a range of diverse production systems. In particular, the mitigation potential of perennial pasture which, while recognised for its potential to improve productivity and sustainability, prior to the present study has not been the subject of such detailed farm- and intra-farm scale analyses previously. Whilst the quantitative results of each farm cannot be directly applied to other systems, they provide the basis for recommendations of strategies with mitigation potential. This is the first study to do so specifically in south-western Australia.

8 CONCLUSION

8.1 Introduction

In recognition of the livestock industry's commitment to reducing its emissions output as Australia inevitably transitions to a carbon-constrained economy, this study sought to build on existing knowledge regarding the carbon footprint of Australian livestock production and potential pathways to mitigate this footprint. The goal of this study was therefore, to investigate the carbon footprint of broadacre livestock production in south-western Australia under different pasture systems and identify regionally appropriate mitigation strategies for adoption by producers. This final Chapter concludes the key research findings and opportunities this presents for sustainable livestock production.

8.2 Research outcomes

This section outlines the outcomes of the three research objectives and the implications of these findings in a broader context.

8.2.1 Objective one: Develop a comprehensive tool that allows the calculation of the carbon footprint of sheep and beef cattle enterprises and examination of ensuing mitigation strategies

Two carbon footprint Frameworks were developed; one for beef cattle production and one for sheep production, in accordance with the four steps of LCA; goal and scope definition, inventory analysis, impact assessment and interpretation. In line with the "cradle to farm gate" scope, the Frameworks accounted for all pre- and on-farm inputs and associated emissions. The functional unit applied to assess the carbon footprint was "kg CO_2-e / kg LW produced for sale".

The Frameworks were designed with the capacity to integrate multiple methodological approaches and data outputs; from NGGI and IPCC methodologies, to detailed farm records, to GrassGro and SimaPro output. Not only did the Frameworks enable whole-farm carbon footprint analyses, but also intra-farm analyses, through the breakdown of feedbases, stock classes and months throughout the production year. The in-depth analyses enabled investigation into the impact of different pasture systems and farm practices, along with the targeted analysis of potential mitigation strategies.

A primary challenge of the Framework development was to balance a model

positioned between the highly complex non-carbon footprint whole-farm analyses which require an in-depth knowledge of the operation of biophysical models and their outputs, and the broader scale carbon footprint studies which, whilst requiring large quantities of information, typically rely on regional or state averages for many animal and pasture attributes. Though this study integrated GrassGro output, with detailed farm and pasture management records, the Frameworks do not require biophysical model output and thus have scope to be utilised by a greater range of users, particularly in farm-specific carbon footprint analyses.

8.2.2 Objective two: Quantify the carbon footprint of livestock production systems in south-western Australia, with focus on perennial versus annual pasture systems

Two beef cattle and two sheep production enterprises, located in the Great Southern, Wheatbelt and Northern Agricultural regions of south-western Australia, participated in the study. Farm records, supported by GrassGro and SimaPro output, data from state government and industry research trials, and literature were applied to the Frameworks.

Carbon footprints, expressed as EI, of the Dongara and Lancelin beef enterprises were 9.17 and 13.20 kg CO_2-e/kg LW produced for sale, whilst the footprints for the Bremer Bay and Wickepin sheep production enterprises were 8.18 and 10.60 kg CO_2-e/kg LW. The primary hotspot at each enterprise was enteric methane production, ranging from 74.4% to 85.1%, whilst the second hotspot was either nitrous oxide emissions from crop residue (8.6%; Bremer Bay), excreta nitrous oxide (6.7%, Wickepin; 5.6%, Lancelin) or emissions from the production of inputs (10.2%; Dongara). Neither enterprise type nor the size of the enterprise was indicative of the carbon footprint of a system. For example, Bremer Bay, a sheep enterprise, was the smallest in terms of arable area and production whilst Dongara, a beef enterprise, was the largest in area and production, yet they produced the two smallest carbon footprints.

Subsequent breakdowns of the whole-farm carbon footprints across feedbases, stock classes and months of the production year revealed the characteristics and practices influencing these differences. Perennial pasture systems were typically the most emissions efficient of the grazed feedbases, ranging from 6.32 to 15.38 kg CO_2-e/ kg LW (Bremer Bay; Wickepin, respectively). This was followed by annual pasture, ranging from 8.10 to 14.21 kg CO_2-e/kg LW (Bremer Bay; Lancelin) and then crop stubble, which if present, ranged from 15.05 to 38.57 kg CO_2-e/kg LW (Wickepin; Bremer Bay). Feedlots, where present, were more emissions efficient, or equivalent,

to the perennials (2.05–6.69 kg CO_2-e/kg LW), reflecting the high feed conversion efficiency by stock whilst being finished for sale. In terms of whole-farm contributions from livestock, emissions from the breeding flock/herd were predominant (70.7-91.1%), with the exception of Dongara where the backgrounding of pastoral cattle was the primary focus of the enterprise and the breeding herd contributed only 19.2%.

The EI of a feedbase was determined by reproductive and grazing management practices, which were in turn, determined by the attributes of the feedbase. For example, the lower EI of annual pasture at some enterprises did not reflect a more efficient pasture system, rather that livestock only grazed the months where the pasture was of high quality. Similarly, inefficiencies were not reflected where the EI of the perennial pasture was higher, rather that the perennial as a grazing source was a secondary role or that the farmer was taking advantage of the year-round productivity to graze breeding stock.

The role of the perennial pasture was demonstrated through the conducted scenario analyses, which found that to maintain the same levels of productivity as the baselines, an annual pasture only system would result in whole-farm EI increases of 3.3 to 4.9%. However, to maintain productivity significant increases in supplementary feed would be required, particularly at the beef enterprises which did not require supplementary feed with a perennial component. Without perennial pasture, the viability of a number of farm management practices observed in the baselines, including accelerated joining, breeding herd expansion, and the backgrounding and agistment of livestock, came into question and in some cases, was no longer viable. The exception to this was at Wickepin whereby the removal of saltbush reduced the whole-farm EI by 0.4%. Where the potential impact of C sequestration was also considered, carbon footprint was partially offset or, in other cases, the enterprise became a net sink of carbon.

8.2.3 Objective three: Examine the mitigation potential of identified strategies on the carbon footprint of livestock production systems and provide regionally appropriate recommendations for application.

Identified practices with mitigation potential fell into two categories; grazing management and reproductive management. These practices influenced the primary hotspots of the enterprises, animal emissions. Alongside the examination of pasture type, which fell under grazing management but also influenced reproductive management, additional farm practices were investigated. Two practices from each category were selected; feedlot finishing and stubble grazing under grazing

management, and altering the herd age distribution through replacement rates and weaning rates under reproductive management. The criteria for selection included evidenced mitigation potential in prior research, on-farm accessibility and potential for short-term implementation. The exception to this was stubble grazing, which was selected to explore whether there were carbon footprint benefits accompanying this commonly practiced management strategy. For each practice, the impact of its removal (where the farm employed the practice), or its implementation (where the farm did not employ the practice), was simulated.

The mitigation potential of each practice varied across both the practices and the livestock enterprise under consideration. No one strategy presented as most or least effective across all livestock enterprises; the impact was dependent on the characteristics of the farm in question. Despite this, feedlot finishing, increased replacement rates and weaning rates reduced the EI of all the enterprises considered, with declines of 0.4 to 8.3%, 1.8 to 20.8% and 0.4 to 5.3%, respectively. Grazing stubble yielded no net emissions benefits, increasing EI by between 0.4 and 0.7%, accompanied by additional supplementary feed requirements.

The findings of the mitigation analyses demonstrated that improved grazing and reproductive management can have both productivity and emissions benefits in livestock systems in southern Australia, but that the magnitude of such benefits will be farm specific. Furthermore, there is often a trade-off between the productivity, emissions, economic and other benefits (i.e. in the case of stubble grazing, simultaneous benefits to the cropping system), from which the ultimate outcome will be determined by farmer motivators.

8.2.4 Significance of study findings

This study contributes significantly to the existing body of knowledge in the field of carbon footprint analysis and sustainable livestock production, both in Australia and internationally. This research was the first of its kind to consider the carbon footprint of beef cattle production and the second to consider sheep meat production in south-western Australia. Within Australia, it was the first study to conduct an intra- and inter-farm carbon footprint comparison of perennial and annual pasture systems. The study makes a significant contribution to the current knowledge gap in Australia with regards to whole-farm carbon footprint mitigation analysis.

Rather than just quantifying the whole-farm carbon footprint and hotspots, as is the case in many carbon footprint studies, the in-depth analysis enabled by the Frameworks permitted a greater understanding as to the particular characteristics and

practices which influence emissions output and productivity of a livestock system. Ultimately, regardless of research findings and policies, the decision to adopt carbon-mitigating practices is made by the farmer. As such, analysis of the carbon footprint and influencing factors at both the farm and intra-farm scale is important, being at the scale at which the farmer operates. The Frameworks enable such analyses and represent the level of detail and understanding to be produced by carbon footprint and LCAs, acting as valuable tools in the decision-making processes that occur at the farm-scale. This will be important in the inevitable transition from quantifying the benchmarking of emissions to direct on-farm action.

A key finding of this research is that perennial pasture systems have the potential to reduce the carbon footprint of livestock production systems. This occurs not through the direct reduction of emissions, but rather through improved farm productivity. The provision of out-of-season feed by perennial pasture enabled the examined farming systems to employ productivity-driven farm practices such as accelerated joining and the backgrounding and agistment of stock in addition to a farm breeding herd. The improved pasture management can also reduce supplementary feeding, which has additional emissions benefits, along with economic benefits not explored directly in this study. The potential for C sequestration, particularly under perennial shrubs, presents a further opportunity to offset farm emissions.

This research demonstrated that there are reproductive and grazing management practices with both on-farm productivity and emissions benefits for southern Australian livestock systems. Importantly, these are practices that have demonstrated implementation in farms, driven by other motivators, and have overcome the barriers to adoption commonly encountered by other mitigation strategies. Whilst such practices alone will not achieve the levels of reduction required to meet emission reduction targets in the long-term, they are accessible to producers immediately and available for short-term implementation. Consequently, they represent the pathways to initiating the transition to the more carbon-efficient production that the industry has been trying to achieve.

8.3 Limitations of the study

The limitations of this study relate to the adopted methodological approach and the scope and time constraints encountered, forming the basis of recommendations for future research.

- Despite extensive and considered reviews of literature and methodologies, the use of case study information and farm-specific modelled output, along with

the development of tailored carbon footprint Frameworks, the results of this study cannot be taken as quantitative duplications of the systems modelled. This is because no whole-farm modelling exercise can directly replicate real-world interactions and outputs. However, the key messages in the findings are relevant and an important contribution to existing research.

- GrassGro parameter sets were not available to model the perennial grass and forage species considered in the study. Previous studies have used temperate perennial grass parameter sets as substitutes for subtropical perennial grasses, however differences between these pasture species were considered too large and entirely unsuitable for the forage shrubs, saltbush and tagasaste. As such whilst annual pasture components were modelled in GrassGro, perennial pasture components were modelled separately using regionally specific data. Divisions in the level of detail in outputs will no doubt accompany with these two approaches.
- GrassGro does not model crops. The biophysical modelling software, APSIM, can do so and the inclusion of the software could be targeted in a future study rather than crop-related information modelled from farm-specific information and attribute values in literature.
- Emissions associated with the pre-farm production of purchased or backgrounded cattle were excluded from the analysis. This could have implications for the findings of the enterprises in question, particularly when examining the impact of strategies with mitigation potential. In line with other literature, these emissions were excluded, recognising the difficulties of collating such detailed pre-farm information given the time and scope restraints of the study.
- Due to a lack of streamlined and consistent methodology, along with data availability issues, emissions associated with LU and dLUC were excluded. Provisional estimates of C sequestration provided in Chapter Six and comparisons with existing studies demonstrated the lack of a universally accepted approach for calculating these emissions and the considerable uncertainty associated with the quantified outputs.
- Difficulties with examining the uncertainty of such complex biological systems were encountered. Monte Carlo analyses can be conducted in SimaPro software to quantify the uncertainty associated with LCA or carbon footprint studies. However, the complex nature of the Frameworks, with data inputs and outputs numbering in the thousands, meant that such integration into Simapro

was a significant task and could not be achieved within the constraints of the study.

- As a carbon footprint study, this research did not consider other impact categories typical of LCAs, such as energy and water demand, eutrophication and acidification potential. Given the diversity of the livestock production systems examined and ensuing impacts of farming practices with mitigation potential, there may be unknown trade-offs between GHG emissions and other categories.
- The cost-effectiveness of the farming practices in Chapter Seven, whilst considered in the selection process, was not directly measured. As alluded to in the examination of the feedlot scenarios, for example, whilst carbon footprint reductions were achieved, it was only possible with large quantities of supplementary feed. The cost-effectiveness of these additional inputs, along with potential infrastructure requirements, was not considered and was likely to be considerable, particularly where the production of on-farm supplementary feed crop was not possible.

8.4 Recommendations for future research

Additional questions and presented opportunities for further research arising from the study are suggested as follows.

8.4.1 Recommendations directly related to the present study

- The carbon footprints Frameworks were developed in Excel. The replication of these calculators in a coded program with a user-friendly interface would improvise their usability. Provided access to sufficient data, most of which can be sourced from comprehensive farm records, this could encourage their use by researchers, industry and farmers. In doing so the Frameworks could become a valuable tool employed in decision-making processes.
- APSIM could be incorporated to simulate the production of crops within the livestock production systems, particularly supplementary feed crops. This would generate daily time-step data which could be aggregated in the same manner as GrassGro, allowing for estimates of improved accuracy of soil and animal emissions pertaining to the crop component of the system.
- A broadening of scope from a carbon footprint to an LCA with multiple impact categories would be valuable to identify interactions and trade-offs which may not be evident in a carbon footprint. The Frameworks could be enhanced and

integrated with relevant tools to achieve this.

- The review of other LCA and carbon footprint studies conducted as part of this study found that chosen method of co-product allocation can have considerable impact on EI and ensuing interpretations. While the research presented in this thesis adopted PMA as the preferred and most appropriate allocation method, it would be of interest to compare these results to those obtained using other approaches, such as economic or biophysical allocation.
- The on-farm adoption of any potential mitigation strategy is dependent on its respective impact on farm profitability. The Frameworks could be enhanced by incorporating economic analyses in the assessment of potential mitigation strategies.

8.4.2 Recommendations building on the findings of this research

- In line with the calls of many other studies, a streamlined methodological approach is needed when conducting carbon footprints or LCAs. Differences in FUs, system boundaries, allocation methods, emissions calculations and EFs, for example, mean that the findings of different studies cannot be directly compared. Yet studies continue to do so, at risk of drawing misleading findings from these comparisons.
- There needs to be greater data quality control when conducting carbon footprint and LCA studies. The quality of carbon footprint results is only as good as the data used to calculate them. Conducting farm-level analyses with state or national averages can mask or skew results, particularly when calculating of hotspots, such as pasture and animal data. This is particularly important when analysing mitigation strategies for on-farm adoption.
- In line with the difficulties outlined earlier regarding the quantification of the emissions contribution by purchased or agisted livestock, there needs to be a universally adopted approach with regards to the calculation of these pre-farm emissions, given the difficulties of collating such detailed study-specific data. There also need to be consistency regarding the allocation of these emissions between the farm under consideration and the farm from which the livestock were transferred.
- Given the role of perennial pasture systems in this study, more attention needs to be directed to the development of perennial pasture parameter sets in biophysical models. In GrassGro, for example, there are currently only parameter sets for a small number of temperate perennial grasses. Expanding

the capacity of software to include subtropical grasses or even shrubs such as tagasaste, would be an invaluable addition.

- The Australian NGGI methodology contains several shortcomings with regards to emission calculation of livestock systems. Examples include; not considering the influence of DMA on cattle intake, restricting analyses to a set range of livestock classes, not making allowances for supplementary feed (particularly relevant in southern Australia where supplementary feeding is common practice), not distinguishing between pasture types in attribute values (i.e. legumes vs grass, annual vs pasture), providing only two options for the amortisation of pasture residue (i.e. 10 or 30 years). The NGGI methodologies undergo a continuous process of revision and improvement and these should be focus of such revisions. Furthermore, given the deferral of so many whole-farm system studies to the state average default values which accompany the NGGI, greater focus should be made to improving the quality of this information to account of the diversity of farming systems throughout Australia.
- There are currently no universally accepted methodologies or data quality requirements for the calculation of LU and dLUC. As such, there are considerable variations between studies which consider these emissions and significant uncertainty within the estimates themselves. As these sources/sinks are recognised for their potential to mitigate emissions from livestock systems, the development of a standard methodological approach and improved data availability is required. This will improve farm-specific estimates, along with the state- or nation-wide estimates which are included in emissions targets and programs such as the ERF.
- An opportunity exists to conduct further carbon footprint analyses of beef and sheep production throughout southern Australia. This would also include examination of potential mitigation strategies, both the impact on other farms and the assessment of strategies not included in the present study. Further analyses would serve to further build on the body of knowledge in the field, particularly the whole-farm impacts of mitigation strategies.
- Accompanying the expectation to reduce carbon footprints, ruminant production systems will also have to adapt to projected climate change. Whilst some limited-scope whole-farm analyses have considered the impact of projected climate change, no reviewed LCA or carbon footprint study has done so. This is likely because most LCA and carbon footprint studies do not integrate biophysical modelling, instead using static data. It is possible to

model climate change in biophysical models using climate projections data. The effect of projected climate change on the annual pasture systems in the present study was modelled in GrassGro, however, this stage was excluded prior to integration in the Frameworks due to scope and time constraints. Considering climate change in LCA and carbon footprint studies is an essential contribution to the body of existing research and focus should be directed to such analyses. This will be invaluable to not only quantify the carbon footprint and productivity changes of specific farming systems under projected climate change but could also influence decisions regarding mitigation strategy implementation. The effectiveness of strategies may differ under projected climate and such knowledge would be particularly important when considering strategies which would be implemented in medium to longer timeframes.

- The advent of digital technologies presents additional opportunities to enhance the quality of carbon footprint analyses. Precision technologies such as remote sensing to measure biomass, probes to measure pasture quality, variable rate technology in farm machinery and GPS tracking of cattle, can all enhance the spatial and temporal data available to LCA and carbon footprint practitioners. Such technology could enable daily or weekly calculations of emissions. Impacts could also be quantified not only at the whole-farm scale (per most studies) or the feedbase scale (per the present study), but also at a paddock scale, greatly enhancing analyses and potentially enabling targeted application of mitigation strategies within the farm.

8.5 Summary

If Australia is to achieve its committed emissions reduction target by 2030, significant and sustained changes need to be implemented now. As a key emitter, the Australian livestock industry shares responsibility, and has indeed committed, to achieve these reductions. Changes in the industry have thus far been insufficient to meet such commitments, hindered by inconsistent policies, insufficient incentives and barriers to adoption of mitigation strategies.

This research set out to fill multiple identified gaps in existing research and build on the current body of knowledge regarding the carbon footprint of livestock production in Australia and ensuing mitigation strategies. In achieving this, a methodological approach was also developed which could potentially improve the calculation of the carbon footprint of livestock enterprises by researchers, industry and national programs such as the ERF.

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APPENDIX A - PARTICIPANT FARM CONSENT FORM

CARBON FOOTPRINT ANALYSIS OF BROADACRE LIVESTOCK GRAZING SYSTEMS IN SOUTHERN AUSTRALIA

Principal Investigator: Danielle Gale, HDR student, Curtin University
Supervisors: Wahidul Biswas & Deborah Pritchard, Curtin University

CONSENT FORM

I …….. agree to participate in the study conducted by Danielle Gale, a Higher Degree by Research (HDR) student from Curtin University. I have been given the opportunity to ask questions and I am now sufficiently informed and understand the purposes of the study. I understand that the information I provide to the investigators will only be used for purposes outlined to me and to those for which I provide my prior consent. I understand that any confidential information will be handled sensitively and in accordance with my requests. I understand that I can withdraw my participation from this study at any time, and if required, any information supplied will be subsequently destroyed or returned to me without prejudice.

Name: ……..

Signature: ……..

Date: ……..

APPENDIX B - CASE STUDY FARM QUESTIONNAIRES

Sheep enterprise questionnaire

QUESTIONNAIRE

GENERAL INFORMATION

Name				
Contact No.				
Email				
Property Location				
Soil Type(s)				
Annual Rainfall	Average	*Year*	*Year*	*Year*

Total property area (ha)			
Current land-use breakdown	**Species** *Please detail whether grown in mix or alone. If grown in a mix what percentage does each species make up?*	**Area (ha)**	**Year of initial establishment**
Crop			
Annual Pasture			
Perennial Pasture			
Other (*please list*)			
Do you employ a rotational cropping/pasture system? If so, please list the relevant crops and pastures, and phase duration for each.			

LIVESTOCK INFORMATION

Please complete using information based on your last complete production year i.e. 2013. Please use average values where detailed information for the year is not available.

Total flock count (head)	
Breed(s)	
Primary product (i.e. prime lamb)	

Breeding stock	**Liveweight (kg)**	**Count (Head)**
Ewes		
Replacement ewes		
Rams		
Replacement rams		

Ewes first bred at _____ months and lamb an average of _____ times before being culled.

Rams are first joined at _____ months and are joined an average of _____ times before being culled.

Lambs are _____kg at birth, are weaned at _____ months and/or at a liveweight of _____ kg.

Sheep production stock (please list applicable)	**Liveweight (kg)**	**Age (months)**	**Count (head)**
Cull ewes			
Cull rams			
Lambs etc			

Average reproductive rate (%)	
Average marking rate (%)	

Were stock bought in?		
If Yes:		
What was bought in?		
Count (head)		
Average liveweight at purchase (kg)		
Average liveweight at sale (kg)		
Age at purchase (months)		
Age at sale (months)		
Please list the key reason(s) for buying in stock		

Sheep Production Calendar

Please list the key tasks carried out each month during a typical production year (this may be different to the calendar year detailed below).

For example; joining, lambing, shearing, drenching, marking/vaccination, weaning, buying/selling stock, supplementary feeding, time on pasture (perennial/annual), crop stubble grazing, sowing pasture, fertiliser application...

JAN	
FEB	
MAR	
APR	
MAY	
JUN	
JUL	
AUG	
SEP	
OCT	
NOV	
DEC	

PASTURE INFORMATION

Given the size of the below forms, this section includes forms for only one annual and one perennial pasture type. Should you require more, please copy and paste the forms as needed.

ANNUAL PASTURE	
Species (if grown as a mix please list all)	
Total pasture area (ha)	
Number of paddocks	
Pasture yield (kg/ha/yr)	
Pasture setup *(soil preparation and seeding)*	
Months sowed/fertilised/first grazed	
Quantity of seed sown (kg/ha) *(If it is a mixed pasture then kg/ha per species)*	
Cost of seed ($/kg, $/t) Name & location of supplier Method of transport to farm	

Chemicals/soil amendments applied *This includes all chemicals & soil amendments applied during soil preparation (i.e. lime, clay dolomite, wetting agents, other soil primers) and during seeding (i.e. fertilisers, pesticides)*	Chemical/ soil amendment type	Month applied	Amount applied (L/ha or kg/ha)	Brand	Cost ($/kg or $/L)	Name of business purchased from/supplier and location	Distance chemical transported from supplier to farm (km)	How was chemical transported to farm?	Mode of transport (make, model of vehicle)	Other details…

Machinery used *This includes all machinery used in the soil preparation and seeding stages (i.e. clayer, seeder, sprayer)*	Machinery type	Frequency of use (i.e. annual)	Brand & model	Source (hired, on-farm)	Cost ($ to hire p/day or to purchase outright)	If hired: Name of business hired from and location	Distance supplier travelled to transport machinery to farm	How was machinery transported to farm?	Period of use (days, weeks)	Machinery usage			
										Fuel consumption (L/hr)	Width (m)	No. passes	Speed (km/hr)

Ongoing *(during and after pasture establishment, and to end of pasture life)*

Chemicals applied *This includes any fungicides, pesticides, fertilisers, lime, wetting agents applied to pasture after establishment*	Chemical type	Month/Yearly interval applied	Amount applied (L/ha or kg/ha)	Brand	Cost ($/kg or $/L)	Name of business purchased from/supplier and location	Distance chemical transported from supplier to farm (km)	How was chemical transported to farm?	Mode of transport (make, model of vehicle)	Other details…

Machinery used *This includes any machinery used on the pasture after establishment and through to preparation for next establishment (i.e. sprayer, topdressor, plough)*	Machinery type	Frequency of use (i.e. annual)	Brand & model	Source (hired, on-farm)	Cost ($ to hire p/day or to purchase outright)	If hired: Name of business hired from and location	Distance supplier travelled to transport machinery to farm	How was machinery transported to farm?	Period of use (days, weeks)	Machinery usage			
										Fuel consumption (L/hr)	Width (m)	No. passes	Speed (km/hr)

PERENNIAL PASTURE	
Species (if grown as a mix please list all)	
Total pasture area (ha)	
Number of paddocks	
Pasture yield (kg/ha/yr)	
Pasture setup *(soil preparation and seeding)*	
Months sowed/fertilised/first grazed	
Quantity of seed sown (kg/ha) *(If it is a mixed pasture then kg/ha per species)*	
Cost of seed ($/kg, $/t) Name & location of supplier Method of transport to farm	

Chemicals/soil amendments applied *This includes all chemicals & soil amendments applied during soil preparation (i.e. lime, clay dolomite, wetting agents, other soil*	Chemical/ soil amendment type	Month applied	Amount applied (L/ha or kg/ha)	Brand	Cost ($/kg or $/L)	Name of business purchased from/supplier and location	Distance chemical transported from supplier to farm (km)	How was chemical transported to farm?	Mode of transport (make, model of vehicle)	Other details…

primers) and during seeding (i.e. fertilisers, pesticides)													
Machinery used *This includes all machinery used in the soil preparation and seeding stages (i.e. clayer, seeder, sprayer)*	Machinery type	Frequency of use (i.e. annual)	Brand & model	Source (hired, on-farm)	Cost ($ to hire p/day or to purchase outright)	If hired: Name of business hired from and location	Distance supplier travelled to transport machinery to farm	How was machinery transported to farm?	Period of use (days, weeks)	Machinery usage			
										Fuel consumption (L/hr)	Width (m)	No. passes	Speed (km/hr)

Ongoing *(during and after pasture establishment, and to end of pasture life)*										
Chemicals applied *This includes any fungicides, pesticides, fertilisers, lime, wetting agents applied to pasture after establishment*	Chemical type	Month/Yearly interval applied	Amount applied (L/ha or kg/ha)	Brand	Cost ($/kg or $/L)	Name of business purchased from/supplier and location	Distance chemical transported from supplier to farm (km)	How was chemical transported to farm?	Mode of transport (make, model of vehicle)	Other details…

Machinery used *This includes any machinery used on the pasture after establishment and through to preparation for next establishment (i.e. sprayer, topdressor, plough)*	Machinery type	Frequency of use (i.e. annual)	Brand & model	Source (hired, on-farm)	Cost ($ to hire p/day or to purchase outright)	If hired: Name of business hired from and location	Distance supplier travelled to transport machinery to farm	How was machinery transported to farm?	Period of use (days, weeks)	Machinery usage			
										Fuel consumption (L/hr)	Width (m)	No. passes	Speed (km/hr)

CROP INFORMATION

CROP										
Species (if grown as a mix please list all)										
Total pasture area (ha)										
Number of paddocks										
Crop yield (kg/ha/yr)										
Crop setup *(soil preparation and seeding)*										
Months sowed/fertilised/harvested										
Quantity of seed sown (kg/ha) *(If it is a mixed pasture then kg/ha per species)*										
Cost of seed ($/kg, $/t) Name & location of supplier Method of transport to farm										
Chemicals/soil amendments applied *This includes all chemicals & soil amendments applied during soil preparation (i.e. lime, clay*	Chemical/ soil amendment type	Month applied	Amount applied (L/ha or kg/ha)	Brand	Cost ($/kg or $/L)	Name of business purchased from/supplier and location	Distance chemical transported from supplier to farm (km)	How was chemical transported to farm?	Mode of transport (make, model of vehicle)	Other details…

dolomite, wetting agents, other soil primers) and during seeding (i.e. fertilisers, pesticides)										

Machinery used *This includes all machinery used in the soil preparation and seeding stages (i.e. clayer, seeder, sprayer)*	Machinery type	Frequency of use (i.e. annual)	Brand & model	Source (hired, on-farm)	Cost ($ to hire p/day or to purchase outright)	If hired: Name of business hired from and location	Distance supplier travelled to transport machinery to farm	How was machinery transported to farm?	Period of use (days, weeks)	Machinery usage			
										Fuel consumption (L/hr)	Width (m)	No. passes	Speed (km/hr)

Ongoing										
Chemicals applied *This includes any fungicides, pesticides, fertilisers,*	Chemical type	Month/Yearly interval applied	Amount applied (L/ha or kg/ha)	Brand	Cost ($/kg or $/L)	Name of business purchased from/supplier and location	Distance chemical transported from supplier to farm (km)	How was chemical transported to farm?	Mode of transport (make, model of vehicle)	Other details…

lime, wetting agents applied to crop over greater than annual intervals											

Machinery used *This includes any machinery used on the crop after establishment and through to preparation for next establishment (i.e. sprayer, topdressor, harvester, baler)*	Machinery type	Frequency of use (i.e. annual)	Brand & model	Source (hired, on-farm)	Cost ($ to hire p/day or to purchase outright)	If hired: Name of business hired from and location	Distance supplier travelled to transport machinery to farm	How was machinery transported to farm?	Period of use (days, weeks)	Machinery usage			
										Fuel consumption (L/hr)	Width (m)	No. passes	Speed (km/hr)

Pasture grazing management							
			If continual:	If rotational:			
Pasture species	Class of stock grazed *Does this vary during the year?*	Rotational or continual grazed?	Stocking rate (Does this vary considerably during the year? How?)	What rotated with? Same pasture different paddocks, different pasture, crop stubble?	Average stocking rate (DSE/ha)	Months of year grazed	Duration of grazing (days)

Crop stubble grazed? Y / N *Please circle*				
If yes:				
Crop	Class of stock grazed	Calendar months grazed	Duration (days)	Stocking rate

Supplementary Feed? *This includes grains, protein meals, urea, molasses, mineral licks, and any other supplements fed to livestock on pasture, crop stubble, feedlots/yards.*									
						If bought in:			
Type (lupin, oats, mineral licks, hay etc)	Calendar months supplied to stock	Duration (days)	Class of stock supplied to	Amount fed (kg/head/day)	Source (i.e. farm, bought in)	Cost ($ per kg/per t)	Brand (if applicable)	Name of business feed purchased from & location	Mode of transport (type & capacity of truck/ute

Drenching						
Class of stock	Frequency (one-off, annual)	When does this take place (month)?	Brand name	Dosage	Cost	Place of purchase

Vaccinations						
Class of stock	Frequency (one-off, annual, booster)	When does this take place (month)?	Brand name	Dosage	Cost	Place of purchase

Supply of drinking water for stock			
Pump Brand & Model	Number	Year(s) of purchase	Operating hours (p/day)
Other (i.e. dams):			

Have your paddocks with pasture been tested for soil carbon? Y / N *Please circle*

Thank you for your participation!

Beef cattle enterprise questionnaire

QUESTIONNAIRE

GENERAL INFORMATION

Name				
Contact No.				
Email				
Property Location				
Soil Type(s)				
Annual Rainfall	Average	*Year*	*Year*	*Year*

Total property area (ha)			
Current land-use breakdown	**Species** *Please detail whether grown in mix or alone. If grown in a mix what is the species ratio?*	**Area (ha)**	**Year of initial establishment**
Crop			
Annual Pasture			
Perennial Pasture			
Other (*please list*)			
Do you employ a rotational cropping/pasture system? If so, please list the relevant crops and pastures, and phase duration for each.			

LIVESTOCK INFORMATION

	Year	*Year*	*Year*
Total herd count (head)			
Breed(s)			
Primary product (i.e. weaners, finished steers)			

Breeding stock	**Liveweight (kg)**	**Count (Head)**		
		Year	*Year*	*Year*
Cows				
Replacement heifers				
Bulls				
Replacement bulls				

Cows first bred at _____ months and calve an average of _____ times before being culled.

Bulls are first joined at _____ months and are joined an average of _____ times before being culled.

Calves are weaned at _____ months and/or at a liveweight of _____ kg.

Beef production stock	**Liveweight (kg)**	**Age (months)**	**Count (head)**		
			Year	*Year*	*Year*
Weaner bulls					
Weaner surplus heifers					
Finished bulls					
Finished surplus heifers					
Cull bulls					
Cull cows					
Other?					

Average reproductive rate (%)	
Average weaning rate (%)	
Average finishing rate (%)	

Were stock bought in? Y / N *Please circle*			
If Yes:	*Year*	*Year*	*Year*
What was bought in? Average liveweight at purchase (kg) Average liveweight at sale (kg) Age at purchase (months) Age at sale (months) Count (head)			
Please list the key reason(s) for buying in stock			

Beef Production Calendar

Please list the key tasks carried out each month during a typical production year (this may be different to the calendar year detailed below).

For example; joining, calving, marking/vaccination, weaning, buying in stock, selling finished stock, supplementary feeding, time on pasture (perennial/annual), crop stubble grazing, sowing pasture, fertiliser application...

JAN	
FEB	
MAR	
APR	
MAY	
JUN	
JUL	
AUG	
SEP	
OCT	
NOV	
DEC	

PASTURE INFORMATION

Should any of this information vary between observed years please detail the year for which it relates. If this represents an average of the observed years please state so.

Given the size of the below forms, this section includes forms for only one annual and one perennial pasture type. Should you require more, please copy and paste the forms as needed.

ANNUAL PASTURE	
Species (if grown as a mix please list all)	
Total pasture area (ha)	
Number of paddocks	
Pasture yield (kg/ha/yr)	
Pasture setup *(soil preparation and seeding)*	
Months sowed/fertilised/first grazed	
Quantity of seed sown (kg/ha) *(If it is a mixed pasture then kg/ha per species)*	
Cost of seed ($/kg, $/t) Name & location of supplier Method of transport to farm	

Chemicals/soil amendments applied *This includes all chemicals & soil amendments applied during soil preparation (i.e. lime, clay dolomite, wetting agents, other soil primers) and during seeding (i.e. fertilisers, pesticides)*	Chemical/ soil amendment type	Month applied	Amount applied (L/ha or kg/ha)	Brand	Cost ($/kg or $/L)	Name of business purchased from/supplier and location	Distance chemical transported from supplier to farm (km)	How was chemical transported to farm?	Mode of transport (make, model of vehicle)	Other details…

Machinery used *This includes all machinery used in the soil preparation and seeding stages (i.e. clayer, seeder, sprayer)*	Machinery type	Frequency of use (i.e. annual)	Brand & model	Source (hired, on-farm)	Cost ($ to hire p/day or to purchase outright)	If hired: Name of business hired from and location	Distance supplier travelled to transport machinery to farm	How was machinery transported to farm?	Period of use (days, weeks)	Machinery usage			
										Fuel consumption (L/hr)	Width (m)	No. passes	Speed (km/hr)

Ongoing *(during and after pasture establishment, and to end of pasture life)*

Chemicals applied *This includes any fungicides, pesticides, fertilisers, lime, wetting agents applied to pasture after establishment*	Chemical type	Month/Yearly interval applied	Amount applied (L/ha or kg/ha)	Brand	Cost ($/kg or $/L)	Name of business purchased from/supplier and location	Distance chemical transported from supplier to farm (km)	How was chemical transported to farm?	Mode of transport (make, model of vehicle)	Other details…

Machinery used *This includes any machinery used on the pasture after establishment and through to preparation for next establishment (i.e. sprayer, topdressor, plough)*	Machinery type	Frequency of use (i.e. annual)	Brand & model	Source (hired, on-farm)	Cost ($ to hire p/day or to purchase outright)	If hired: Name of business hired from and location	Distance supplier travelled to transport machinery to farm	How was machinery transported to farm?	Period of use (days, weeks)	Machinery usage			
										Fuel consumption (L/hr)	Width (m)	No. passes	Speed (km/hr)

PERENNIAL PASTURE	
Species (if grown as a mix please list all)	
Total pasture area (ha)	
Number of paddocks	
Pasture yield (kg/ha/yr)	
Pasture setup *(soil preparation and seeding)*	
Months sowed/fertilised/first grazed	
Quantity of seed sown (kg/ha) *(If it is a mixed pasture then kg/ha per species)*	
Cost of seed ($/kg, $/t) Name & location of supplier Method of transport to farm	

Chemicals/soil amendments applied *This includes all chemicals & soil amendments applied during soil preparation (i.e. lime, clay dolomite, wetting agents, other soil*	Chemical/ soil amendment type	Month applied	Amount applied (L/ha or kg/ha)	Brand	Cost ($/kg or $/L)	Name of business purchased from/supplier and location	Distance chemical transported from supplier to farm (km)	How was chemical transported to farm?	Mode of transport (make, model of vehicle)	Other details…

primers) and during seeding (i.e. fertilisers, pesticides)													
Machinery used *This includes all machinery used in the soil preparation and seeding stages (i.e. clayer, seeder, sprayer)*	Machinery type	Frequency of use (i.e. annual)	Brand & model	Source (hired, on-farm)	Cost ($ to hire p/day or to purchase outright)	If hired: Name of business hired from and location	Distance supplier travelled to transport machinery to farm	How was machinery transported to farm?	Period of use (days, weeks)	Machinery usage			
										Fuel consumption (L/hr)	Width (m)	No. passes	Speed (km/hr)

Ongoing *(during and after pasture establishment, and to end of pasture life)*										
Chemicals applied *This includes any fungicides, pesticides, fertilisers, lime, wetting agents applied to pasture after establishment*	Chemical type	Month/Yearly interval applied	Amount applied (L/ha or kg/ha)	Brand	Cost ($/kg or $/L)	Name of business purchased from/supplier and location	Distance chemical transported from supplier to farm (km)	How was chemical transported to farm?	Mode of transport (make, model of vehicle)	Other details…

Machinery used *This includes any machinery used on the pasture after establishment and through to preparation for next establishment (i.e. sprayer, topdressor, plough)*	Machinery type	Frequency of use (i.e. annual)	Brand & model	Source (hired, on-farm)	Cost ($ to hire p/day or to purchase outright)	If hired: Name of business hired from and location	Distance supplier travelled to transport machinery to farm	How was machinery transported to farm?	Period of use (days, weeks)	Machinery usage			
										Fuel consumption (L/hr)	Width (m)	No. passes	Speed (km/hr)

Pasture grazing management							
			If continual:	If rotational:			
Pasture species	Type of stock grazed *Does this vary during the year?*	Rotational or continual grazed?	Stocking rate (Does this vary considerably during the year?)	What rotated with? Same pasture different paddocks or with different pasture?	Average stocking rate (DSE/ha)	Months of year grazed	Duration of grazing (days)

Stubble grazed? Y / N *Please circle*				
If yes:				
Year	Type of crop stubble	Calendar months grazed	Duration (days)	Stocking rate
Year				
Year				

Supplementary Feed?

This includes grains, protein meals, urea, molasses, mineral licks, and any other supplements fed to livestock

Y / N *Please circle*

If Yes:

Please complete the below section – if the type/amount/duration of supplement varies with different pastures, please also complete according to each pasture type.

					If bought in:				
Type (lupin, oats etc)	Calendar months fed	Duration (days)	Amount fed (kg/head/day)	Source (i.e. farm, bought in)	Cost (per kg/per t)	Brand (if applicable)	Name of business feed purchased from & location	Distance feed transported from place of purchase to farm (km)	Mode of transport (type & capacity of truck/ute

Drenching

Class of stock	Frequency (once off, annual)\	When does this take place (month)?	Brand name	Dosage	Cost	Place of purchase

Vaccinations						
Class of stock	Frequency (once off, annual, booster)	When does this take place (month)?	Brand name	Dosage	Cost	Place of purchase

Supply of drinking water for stock			
Pump Brand & Model	Number	Year(s) of purchase	Operating hours (p/day)

Have your paddocks with pasture been tested for soil carbon? Y / N *Please circle*

If yes, can you please provide the data, or a contact name?

_____

Thank you for your participation!

APPENDIX C - FRAMEWORK INPUT PARAMETERS

The following tables list the parameters requiring input within the carbon footprint Frameworks to calculate the carbon footprint. Some fields require manual entry by the user using sourced data, others automatically populate from calculations on various worksheets within the Framework.

Table C.1 - Livestock parameters in the carbon footprint Frameworks

Input	Unit	Source	Manual or automatic link
Stock class		Farm information	Manual
Monthly liveweight (average) per stock class	kg	Farm information or secondary source	Manual (if maintenance weight) and automatic (if growing)
Monthly count (average) per stock class		Farm information	Manual
Months of joining, lambing, weaning	Month	Farm information	Manual
Annual count per stock class		Farm information	Manual
Sale liveweight (average)	kg	Farm information	Manual
Lambing/Calving rate	%	Farm information	Manual
Marking rate	%	Farm information	Manual
Weaning rate	%	Farm information	Manual
Lamb/calf birthweight	kg	Farm information or secondary source	Manual
Lamb/calf weight first on pasture	kg	Farm information	Manual
Lamb/calf age first on pasture	Weeks	Farm information	Manual
Lamb/calf weight at weaning	kg	Farm information	Manual
Lamb/calf age at weaning	Weeks	Farm information	Manual
Replacement stock starting weight	kg	Farm information	Manual
Replacement stock starting age	Weeks	Farm information	Manual
Replacement stock weight at maturity	kg	Farm information	Manual

Replacement stock age at maturity	Weeks	Farm information or secondary source	Manual
Monthly proportion of breeding ewes/cows lactating	%	Farm information	Manual
Monthly feed adjustment (FE)[b]	1.1 or 1.3 depending on calving month	Secondary source	Manual
Monthly milk intake (MC)[b]	kg/day	Secondary source	Manual
Monthly milk production (MP)[b]	kg/day	Secondary source	Manual
Monthly greasy wool production	kg/head	Farm information or secondary source	Manual
Clean yield percentage[a]	%	Farm information or secondary source	Manual
Monthly liveweight gain (LWG)	kg	Farm information	Manual (maintenance stock) and automatic (growing stock)
Standard reference weight (SRW)	kg	Farm information or secondary source	Manual
Dry Standard Equivalent (DSE)	DSE	Secondary source	Manual
Energy requirement for maintenance	MJ ME/head/day	Secondary source	Manual
Energy required for growth	MJ ME/head/day	Secondary source	Manual

[a] Sheep framework only

[b] Cattle framework only

Table C.2 - Feedbase parameters in the carbon footprint Frameworks

Input	**Unit**	**Source**	**Manual or automatic link**
Area	Ha	Farm information	Manual
Years since establishment	Yr	Farm information	Manual
Species		Farm information	Manual
Crop yield	(kg/hay/yr)	Farm information	Manual
DM content of grain	%	Secondary source	Manual
DM content of stubble	%	Secondary source	Manual
Harvest index (HI)	%	Secondary source	Manual
Pasture/stubble grazed for each month	Y or N	Farm information	Manual
Monthly DMA of each feedbase type	t DM/ha	GrassGro or other secondary source	Manual (pasture) automatic (crop, from other section of “Farm specifications”)
Monthly ME of each	MJ/kg DM	GrassGro or other	Converted from DMD

feedbase type		secondary source	values
Monthly DMD of each feedbase type	%	GrassGro or other secondary source	Manual
Monthly CP content of each feedbase type	%	GrassGro or other secondary source	Manual
Monthly intake of crop stubble per stock class	kg DM/head/day		Automatic (from "CH_4 from enteric emissions")

Table C.3 - Feedlot parameters in the carbon footprint Frameworks

Input	**Unit**	**Source**	**Manual or automatic link**
Ration component		Farmer information	Manual
ME	MJ/kg DM	Secondary source	Manual
DMD	%	Secondary source	Converted from ME values
DM content	%	Secondary source	Manual
CP content	%	Secondary source	Manual
Amount per t of grain[a]	kg or l/t	Farmer information	Manual
Density[b]	kg/t	Secondary source	Manual
Grain proportion of ration	%	Farmer information	Manual
Hay proportion of ration	%	Farmer information	Manual

[a] Applicable to grain mixes with multiple ingredients only

[b] Liquid components of ration only

Table C.4 - Supplementary feed parameters in the carbon footprint Frameworks

Input	**Unit**	**Source**	**Manual or automatic link**
Monthly supplementary feeding	1, 2, or 3 depending on type of feed	Farm information	Manual
Supplementary feed type		Farmer information	Manual
ME	MJ/kg DM	Secondary source	Manual
DMD	%	Secondary source	Manual
DM content	%	Secondary source	Manual
CP content	%	Secondary source	Manual
Proportion of total supplementary feed provided	%	Farmer information	Manual

Table C.5 - Parameters for enteric methane calculations in the carbon footprint Frameworks

Input	Unit	Source	Manual or automatic link
Pasture & crop type & feedlot			Automatic (from “Farm specifications”)
Stock class			Automatic (from “Farm specifications”)
Monthly digestibility (DMD) of pasture/stubble/feedlot feed	%		Automatic (from “Farm specifications”)
Monthly liveweight (W)	kg		Automatic (from “Farm specifications”)
Average daily liveweight gain (LWG) for each month[a]	kg/head/day		Automatic (“Farm specifications”)
Monthly dry matter availability (DMA) of pasture or stubble	t/ha		Automatic (from “Farm specifications”)
Monthly proportion of breeding ewes/cows lactating	%		Automatic (from Farm specifications”)
Monthly feed adjustment for MA	1.0 or 1.3		Automatic (from “Farm specifications”)
Monthly metabolisable energy (ME) content of feedbase	MJ/kg DM		Automatic (from “Farm specifications”)
Monthly animal ME requirement	MJ/day	Secondary source	Automatic (from “Farm specifications”) and manual (when inter-monthly changes)
Milk consumption	kg/head/day		Automatic (from “Farm specifications”)
ME content of milk	MJ/kg		Automatic (from “N_2O emissions from excreta)
ME content of each supplementary feed type	MJ/kg DM		Automatic (from “Farm specifications”)
DMD of each supplementary feed type[a]	%		Automatic (from “Farm specifications”)
Type of supplementary feed fed each month			Automatic (from “Farm specifications”)
Proportion each supplementary feed contributes to total	%		Automatic (from “Farm specifications”)
Monthly number of stock in each class	Head		Automatic (from “Farm specifications”)

[a] Beef framework only

Table C.6 - Parameters for manure methane emission calculations in the carbon footprint Frameworks

Input	Unit	Source	Manual or automatic link
Monthly weighted DMD	%	GrassGro and secondary source	Automatic (from "N_2O emission from manure")
Daily feed intake per month	kg DM/head/day		Automatic (from "CH_4 emissions from enteric fermentation")
Monthly number of stock in each class	Head		Automatic (from "Farm specifications")

Table C.7 - Farm machinery parameters in the carbon footprint Frameworks

Input	Unit	Source	Manual or automatic input
Machinery type		Farm information	Manual
Feedbase area	Ha		Automatic (from "Farm specifications")
Years since establishment of feedbase type	Years		Automatic (from "Farm specifications")
Number of passes at establishment		Farm information	Manual
Number of passes following establishment		Farm information	Manual
Machinery width (where applicable)	m	Farm information, or Secondary source	Manual
Speed	km/hr	Farm information, or Secondary source	Manual
Fuel consumption of machine	l/hr	Farm information, or Secondary source	Manual
Cost of machinery	AUD	Farm information, or Secondary source	Manual
Lifetime of machinery	Years	(ATO, 2018)	Manual

Table C.8 - Feedbase chemical parameters in the carbon footprint Frameworks

Input	Unit	Source	Manual or Automatic link
Chemical name & brand		Farm information	Manual
Active ingredient		Secondary source (MSDS)	Manual
Mass of active ingredient (AI)	g/l or g/kg	Secondary source (MSDS)	Manual
Density of chemical	kg/l	Secondary source (MSDS)	Manual
Nitrogen content	%	Secondary source	Manual
Age of feedbase type	Yr		Automatic (from "Farm specifications")
Feedbase area	Ha		Automatic (from "Farm specifications")
Application rate	l/ha/yr or kg/ha/yr	Farm information	Manual

Table C.9 - Veterinary product parameters in the carbon footprint Frameworks

Input	Unit	Source	Manual or Automatic link
Chemical name & brand		Farm information	Manual
Active ingredient		Secondary source (MSDS)	Manual
Mass of active ingredient (AI)	g/l	Secondary source (MSDS)	Manual
Density of chemical	kg/l	Secondary source (MSDS)	Manual
Application rate	ml/head/yr	Farmer information or secondary source	Manual
Stock numbers per livestock class	No. head		Automatic (from "Farm Specifications")

Table C.10 - Feedlot ration and supplementary feed parameters in the carbon footprint Frameworks

Input	Unit	Source	Manual or Automatic link
Feed type		Farm information	Manual
Brand of feed		Farm information	Manual
DM content	%	Secondary source	Manual
Wastage	%		Manual
Feed intake	kg DM/day		Automatic (from "CH_4 from enteric emissions)
Stock numbers per month	Head		Automatic (from "Farm specifications")

Table C.11 - Seed parameters in the carbon footprint Frameworks

Input	Unit	Source	Manual or Automatic link
Seed type		Farm information	Manual
Application rate	kg/ha	Farm information	Manual
Years since establishment of feedbase type	Yr		Automatic (from “Farm specifications”)
Feedbase area	Ha		Automatic (from “Farm specifications”)

Table C.12 - Transport parameters in the carbon footprint Frameworks

Input	Unit	Source	Manual or automatic link
Input type			Automatic (linked from respective worksheet, i.e. “veterinary products”, “plant chemicals”)
Journey 1 (manufacturer to distributor 1) - Method of transport - Distance travelled - Details	km	Farm information, or secondary source	Manual
Journey 2 (distributor 1 to distributor 2) - Method of transport - Distance travelled - Details	km	Farm information, or secondary source	Manual
Journey 3 (distributor 2 to distributor 3) - Method of transport - Distance travelled - Details	km	Farm information, or secondary source	Manual
Journey 4 (distributor 3 to farm) - Method of transport - Distance travelled - Details	km	Farm information, or secondary source	Manual
Input weight	kg		Automatic (linked from respective worksheet, i.e. “veterinary products”, “plant chemicals”)

Table C.13 - Parameters for the N fertiliser nitrous oxide emission calculations

Input	Unit	Source	Manual or automatic link
Type of fertiliser applied		Farmer information	Manual
N content of fertiliser	%	Secondary source	Manual
Mass of fertiliser applied to each feedbase type	kg/yr	Farmer information	Manual
EF (N fertiliser)= 0.002	kg N_2O-N/kg N	NIR (2017)	Automatic (from “EF” worksheet”)

Table C.14 - Parameters for the dung and urine nitrous oxide emission calculations

Input	Unit	Source	Manual or automatic link
Monthly mass of faecal N deposited	kg		Automatic (from “N excretion onto paddock”)
Monthly mass of urinary N deposited	kg		Automatic (from “N excretion onto paddock”)
Mass of fertiliser applied to each feedbase type	kg/yr	Farmer information	Manual
EF (N dung and urine deposited)= 0.004	kg N_2O-N/kg N	NIR (2017)	Automatic (from “EF” worksheet”)

Table C.15 - Parameters for pasture & crop residue nitrous oxide emission calculations

Input	Unit	Source	Manual or automatic link
Area	Ha		Automatic (from “Farm specifications”)
Pasture or crop residue remaining after grazing	kg DM/ha/yr		Automatic (from “Farm specifications”) or manual from GrassGro output
Below-ground:above-ground residue ratio		NIR (2017)	Manual
Above-ground N content	kg/ha	NIR (2017) or GrassGro	Manual
Below-ground N content	kg/ha	NIR (2017)	Manual
EF (N from pasture or crop residue)= 0.01	kg N_2O-N/kg N	NIR (2017)	Automatic (from “EF” worksheet)

Table C.16 - Parameters for nitrous oxide emissions from leaching & runoff calculations

Input	Unit	Source	Manual or automatic link
Annual evapotranspiration		GrassGro	Manual
Annual rainfall	ml	Secondary source	Manual
Fertiliser N applied to each feedbase type	kg/yr		Automatic (from other section of “soil emissions”)
Dung and urine N applied to each feedbase type	kg/yr		Automatic (from other section of “soil emissions”)
Pasture and crop residue N applied to each feedbase type	kg/yr		Automatic (from other section of “soil emissions”)
EF (leaching and runoff) = 0.0075	kg N_2O-N/kg N	NIR (2017)	Automatic (from “EF” worksheet”)

Table C.17 - Parameters for nitrous oxide emissions from atmospheric deposition calculations

Input	Unit	Source	Manual or automatic link
Type of fertiliser applied		Farmer information	Manual
N-content of fertiliser	%	Secondary source	Manual
Mass of fertiliser applied on each feedbase type	kg/yr		Automatic (from other section of "soil emissions" worksheet)
Faecal N excreted for each stock class on each feedbase type	kg/yr		Automatic (from "N excretion on paddock" worksheet)
Urinary N excreted for each stock class on each feedbase type	kg/yr		Automatic (from "N excretion on paddock" worksheet)
EF (N fertiliser atmospheric deposition) = 0.002	kg N_2O-N/kg N	NIR (2017)	Automatic (from "EF" worksheet)
EF (urine & dung N atmospheric deposition) = 0.004	kg N_2O-N/kg N	NIR (2017)	Automatic (from "EF" worksheet)

Table C.18 - Parameters for nitrous oxide from urea hydrolysis calculations

Input	Unit	Source	Manual or automatic link
Urea applied to feedbase type	kg/yr		Automatic (from "Plant chemicals")
EF (urea) = 0.2	CO_2-C/kg lime	NIR (2017)	Automatic (from "EF" worksheet)

Table C.19 - Input fields for carbon dioxide emissions from liming calculations

Input	Unit	Source	Manual or automatic link
Mass of lime/dolomite applied to feedbase type	kg/yr		Automatic (from "Plant chemicals")
EF (lime) = 0.12	CO_2-C/kg lime	IPCC 2006	Automatic (from "EF" worksheet)
EF (dolomite) = 0.13	CO_2-C/kg dolomite	IPCC 2006	Automatic (from "EF" worksheet)

APPENDIX D - CARBON FOOTPRINT FRAMEWORKS

Please contact the author for further information regarding the Frameworks developed for this research.

APPENDIX E - FARM MACHINERY INVENTORY AND EMISSIONS CALCULATIONS

The steps followed in the two carbon footprint Frameworks to calculate the inventory outputs and emissions associated with the manufacture of farm machinery and farm machinery fuel consumption are outlined in this Appendix. These emissions sources are explained in Chapter Three, Section 3.3.4.4.

Step 1: Calculation of distance travelled per hectare (D_m, km/ha/yr) by a machinery type on a feedbase

$$D_m = \frac{\left[\frac{10,0000}{w}\right]}{1,000} \times p_m \qquad (0.1)$$

Where:

D_m = distance travelled per hectare by the machinery on feedbase (km/ha/yr)

w = width of machinery/implement used (m)

p_m = number of passes the machinery makes annually

For each feedbase, distance travelled was calculated separately for passes made at establishment of the feedbase (i.e. pre-seeding and seeding) and passes made later in the production year (i.e. annual fertilising of both annuals and perennials) or on a reoccurring basis (i.e. liming every seven years). This enabled the distance travelled by machinery in the establishment year to be annualised for perennial pastures.

Step 2: Calculation of total distance travelled (TD_m; km/yr) by a machinery type on a feedbase either at establishment or ongoing

$$TD_m = D_m \times Area \qquad (0.2)$$

Where:

TD_m = total distance travelled by the machinery on feedbase (km/yr)

D_m = as calculated using equation 3.26 (km/ha/yr)

Area = total area of feedbase (ha)

This was also calculated separately for distance travelled at establishment (annualised establishment for perennials) and distance travelled ongoing through the

remainder of the production year.

Step 3: Calculation of total distance (TD_{mf}; km/yr) travelled by a machinery type on a feedbase in a production year

In this step, the annual distance travelled at establishment (TD_m establishment) and the annual distance travelled through the production year (TD_m ongoing) were totalled to obtain total distance travelled by a machinery type on a feedbase (TD_{mf}; km/yr). This value was calculated separately for each machinery type and each feedbase.

The obtained value was then used to calculate the cost of machinery and total fuel consumption of each machinery type. These values were then used to calculate emissions arising from the manufacture of each machinery type and those arising from the production of fuel consumed on-farm. These steps are outlined below.

Step 4: Calculation of the annual operational time of a machinery type (O_m; hr/yr) on a feedbase

$$O_m = \frac{TD_{mf}}{v_m} \quad (0.3)$$

Where:
O_m = annual operational time of the machinery on feedbase (hr/yr)
TD_{mf} = as calculated in equation 3.27 (km/yr)
v_m = operational speed of machinery (km/hr)

Where an implement was used, the calculated operational time applied to both the implement and the tractor used to operate that implement. The value obtained in this step is then used to calculate the on-farm operational lifetime of each machinery type and the ensuing cost of the machinery on a per annum basis. This will differ for a tractor, which is the sum of all implements it operates, as opposed to the implement which only accounts for the time it is utilised.

Step 5: Calculation of the on-farm lifetime (L_m; hr) of a machinery type

$$L_m = \Sigma(O_{m\ annual} + O_{m\ perennial} + \ldots) \times Life \quad (0.4)$$

Where:
L_m = on-farm lifetime of the machinery (hr)
O_m = as calculated in equation 3.28 (hr/yr)
Life = lifetime of the machinery (yr), sourced from ATO (2018)

Step 6: Calculation of the cost of a machinery type per hour ($Cost_{m\ hr}$; AUD/hr) across the farm

$$Cost_{m\ hr} = \frac{Price_m}{L_m} \quad (0.5)$$

Where:

$Cost_{m\ hr}$ = cost of the machinery per hour (AUD/hr)

$Price_m$ = original purchase price of the machinery (AUD)

L_m = as calculated in equation 3.29 (yr)

Step 7: Calculation of the cost of a machinery type per hectare ($Cost_{m\ ha}$; AUD/ha) operated on a feedbase

The EF employed to calculate emissions associated with the manufacture of machinery was sourced from the USA Input/Output Simapro database and was based on the value of machinery. This EF was presented as emissions produced per USD (1998 equivalent) equivalent of farm machinery, so the cost of farm machinery had be deflated to the 1998 price in AUD and converted to USD before the EF could be applied.

$$Cost_{m\ ha} = Cost_{m\ hr} \times O_m \quad (0.6)$$

Where:

$Cost_{m\ ha}$ = cost of machinery per hectare, allocating according to use on feedbase (AUD/ha)

$Cost_{m\ hr}$ = as calculated using equation 3.30 (AUD/hr)

O_m = as calculated in equation 3.28 (hr/yr)

For implements such as boomsprays, or sole-purpose vehicles such as harvesters, this value represents the total hours on each hectare that the implement was in use on a feedbase. For tractors, this value represents the total hours on each hectare the tractor was in use with all implements it operates in a production year.

Step 8: Calculation of the cost of a machinery type operated on a feedbase in 1998 AUD equivalent ($Cost_{AUD1998}$; AUD_{1998}/ha)

$$Cost_{AUD1998} = \frac{Cost_{m\ ha}}{(1+\pi)^{(t-x)}} \quad (0.7)$$

Where:

$Cost_{AUD1998}$= cost of the machinery per hectare in 1998 AUD equivalent, allocated according to use on feedbase (AUD_{1998}/ha)

$Cost_{m\ ha}$= as calculated in equation 3.31 (AUD/ha)

π= average annual inflation rate between 1998-2013, 2.90% pa (RBA, 2015b)

t= year deflation calculation commences

x= target year for deflation, 1998

Step 9: Calculation of the cost of a machinery type operated on a feedbase in 1998 USD equivalent ($Cost_{USD1998}$; USD_{1998}/ha)

$$Cost_{USD1998} = Cost_{AUD1998} \times ER_{USD:AUD} \quad (0.8)$$

Where:

$Cost_{USD1998}$= cost of the machinery per hectare in 1998 USD equivalent, allocated according to use on feedbase (USD_{1998}/ha)

$Cost_{AUD1998}$= as calculated in equation 3.32 (AUD_{1998}/ha)

$ER_{USD:AUD}$= USD:AUD exchange rate in 1998; 0.629 (RBA, 2015a)

Step 10: Calculation of the total annual cost of a machinery type operated on a feedbase in 1998 USD equivalent (Total $Cost_{USD1998}$; USD_{1998}/yr)

$$\text{Total } Cost_{USD1998} = Cost_{USD1998} \times \text{Area} \quad (0.9)$$

Where:

Total $Cost_{USD1998}$= total annual cost of the machinery in 1998 USD equivalent, allocated according to use on feedbase (USD_{1998}/yr)

$Cost_{USD1998}$= as calculated in equation 3.33 (USD_{1998}/ha)

Area= area of feedbase (ha)

Step 11: Calculation of the annual GHG emissions (kg CO_2-e/yr) associated with the manufacture of a machinery type

Emissions arising from the manufacture of a machinery type, as allocated to according to use on the feedbase considered, was converted directly to CO_2-e in the “Impact assessment” component using an EF obtained from a Simapro database.

$$E_m = \text{Total } Cost_{USD1998} \times EF_m \quad (0.10)$$

Where:

E_m= annual emissions resulting from the production of machinery, allocated according to use on feedbase (kg CO_2-e/yr)

Total $Cost_{USD1998}$= as calculated in equation 3.34 (USD/yr)

EF_m= machinery manufacture EF (kg CO_2-e/USD 1998)

Step 12: Calculation of the annual fuel consumption of a machinery type (F_m, l/yr) on a feedbase

a) $F_m = O_m \times r_f$ (0.11)

Where:

F_m= annual fuel consumption of the machinery on feedbase (l/yr)

O_m= as obtained in equation 3.28 (hr/yr)

r_f= rate of fuel use by the machinery (l/hr)

For stand-alone machinery, such as harvesters, this calculated value represented fuel consumption during its operation. For machinery implements, this value represented fuel consumption by the tractor whilst pulling the implement.

b) In the case where a tractor operates multiple implements over the production year, fuel consumption (l/yr), as calculated in equation 3.36, whilst operating each machinery implement on a feedbase is totalled. For example;

$F_{tractor} = F_{seeder} + F_{sprayer} + F_{spreader} + \ldots$

Step 13: Conversion of the annual fuel consumption from l/yr to kg/yr

Per plant chemicals, the EF for the production of diesel required the inventory value in kilograms. As such, the annual application of diesel, as derived in equation 3.36, was converted to kg/yr using the density (kg/l) of diesel.

This step was also required when calculating emissions associated with the transportation of diesel to the farm.

Step 14: Calculation of the annual GHG emissions (kg CO_2-e/yr) associated with the production of diesel consumed during on-farm machinery operation for a feedbase

Diesel consumed on each feedbase was converted directly to CO_2-e (E_{fuel}; kg CO_2-e/yr) in the "Impact assessment" component using an EF obtained from Simapro databases.

$E_{fuel} = F_m \times EF_{fuel}$ (0.12)

Where:

E_{fuel}= annual emissions resulting from the production of diesel consumed by the machinery on feedbase (kg CO_2-e/yr)

F_m= as calculated in equation 3.36 (l/yr)

EF_{fuel}= diesel EF (kg CO_2-e/kg diesel consumed)

APPENDIX F - CARBON FOOTPRINT ANALYSES

Table F.1 - Baseline carbon footprints, expressed as emissions intensity (EI) and total emissions quantity (EQ), of the four livestock production enterprises

Case study farm	Production of inputs	Transport-ation of inputs	Farm machinery operation	CH_4 enteric	CH_4 manure	N_2O excreta	N_2O N fertilisers	N_2O crop residue	N_2O pasture residue	N_2O leaching/ run off	Atmospheric deposition	CO_2 liming	CO_2 urea hydrolysis	TOTAL
BREMER BAY														
EI (kg CO_2-e/kg LW)	0.45	0.12	0.14	6.08	0.00	0.51	-	0.70	0.07	-	0.10	-	-	**8.18**
EQ (kg CO_2-e/ yr)	25977	6763	8367	352600	77	29535	-	40655	4304	-	5907	-	-	**474185**
WICKEPIN[a]														
EI (kg CO_2-e/kg LW)	0.59	0.07	0.18	7.95	0.00	0.71	0.01	0.24	0.11	-	0.13	0.62	-	**10.60**
EQ (kg CO_2-e/ yr)	63977	7533	19635	858392	209	76438	630	25520	11549	-	14579	66610	-	**1145073**
DONGARA														
EI (kg CO_2-e/ kg LW)	0.93	0.04	0.08	7.13	0.00	0.57	0.02	-	0.05	-	0.12	0.20	0.02	**9.17**
EQ (kg CO_2-e/ yr)	412063	18027	37532	3150200	732	252173	9363	-	22890	-	51371	90567	7040	**4051957**
LANCELIN														
EI (kg CO_2-e/ kg LW)	0.68	0.08	0.15	11.23	0.00	0.73	-	-	0.14	-	0.18	-	-	**13.20**
EQ (kg CO_2-e/ yr)	102257	12594	22054	1686074	287	110130	-	-	20521	-	27100	-	-	**1981018**

[a] Post protein mass allocation of emissions between meat and wool production

Table F.2 - Contribution of the pre-farm emission sources to the baseline carbon footprints of the four livestock production enterprises

Case study farm	Herbicides	Pesticides	Fertilizers	Lime	Urea (pasture/ crop)	Veterinary products	Feed & Supplements	Farm machinery	Seed	Diesel	TOTAL
BREMER BAY											
Production of inputs											
EI (kg CO_2-e/kg LW)	0.04	-	0.14	-	-	0.00	0.11	0.11	0.03	0.01	**0.45**
EQ (kg CO_2-e/ yr)	2422	-	8211	-	-	271	6363	6358	1499	853	**25977**
Transportation of inputs											
EI (kg CO_2-e/kg LW)	0.00	-	0.01	-	-	0.00	0.10	-	0.00	0.00	**0.12**
EQ (kg CO_2-e/ yr)	190	-	748	-	-	15	5590	-	212	8	**6763**
WICKEPIN [a]											
Production of inputs											
EI (kg CO_2-e/kg LW)	0.10	0.00	0.12	0.04	-	0.01	-	0.22	0.08	0.02	**0.59**
EQ (kg CO_2-e/ yr)	10585	35	13251	4693	-	1159	-	23505	8748	2001	**63977**
Transportation of inputs											
EI (kg CO_2-e/kg LW)	0.00	0.00	0.01	0.06	-	0.00	-	-	0.00	0.00	**0.07**
EQ (kg CO_2-e/ yr)	422	2	571	6029	-	55	-	-	435	18	**7533**
DONGARA											
Production of inputs											
EI (kg CO_2-e/kg LW)	0.01	0.00	0.82	0.02	-	0.00	0.04	0.02	0.02	0.01	**0.93**
EQ (kg CO_2-e/ yr)	4000	67	360899	8820	-	6	16747	7630	10070	3824	**412063**
Transportation of inputs											
EI (kg CO_2-e/kg LW)	0.00	0.00	0.02	0.02	-	0.00	0.01	-	0.00	0.00	**0.04**
EQ (kg CO_2-e/ yr)	437	22	7063	7686	-	60	2294	-	401	64	**18027**
LANCELIN											
Production of inputs											
EI (kg CO_2-e/kg LW)	0.01	0.00	0.34	-	-	0.00	0.25	0.06	0.00	0.01	**0.68**
EQ (kg CO_2-e/ yr)	1300	189	50649	-	-	10	37771	9664	427	2247	**102257**
Transportation of inputs											
EI (kg CO_2-e/kg LW)	0.00	0.00	0.05	-	-	0.00	0.03	-	0.00	0.00	**0.08**
EQ (kg CO_2-e/ yr)	74	18	7747	-	-	22	4591	-	82	61	**12594**

[a] Post-allocation of emissions between meat and wool production

Table F.3 - Emissions breakdown, expressed as emissions intensity (EI) and total emissions quantity (EQ), across the different feedbase types at the Bremer Bay sheep enterprise

Feedbase type	Production of inputs	Transport-ation of inputs	Farm machinery operation	CH_4 enteric	CH_4 manure	N_2O excreta	N_2O N fertilisers	N_2O crop residue	N_2O pasture residue	N_2O leaching/ run off	Atmospheric deposition	CO_2 liming	CO_2 urea hydrolysis	TOTAL
EI (kg CO_2-e/kg LW)														
Annual pasture	0.30	0.09	0.05	6.69	0.00	0.67	0.00	0.00	0.16	0.00	0.13	0.00	0.00	**8.10**
Subtropical grass pasture	0.29	0.11	0.05	7.06	0.00	0.59	0.00	0.00	0.06	0.00	0.12	0.00	0.00	**8.28**
Lupins	4.47	0.41	2.22	9.20	0.00	0.34	0.00	9.11	0.00	0.00	0.07	0.00	0.00	**25.82**
Oats	4.67	0.41	2.63	9.50	0.00	0.27	0.00	21.05	0.00	0.00	0.05	0.00	0.00	**38.57**
Feedlot	0.17	0.11	0.00	1.64	0.00	0.11	0.00	0.00	0.00	0.00	0.02	0.00	0.00	**2.05**
EQ (kg CO_2-e/yr)														
Annual pasture	4592	1330	744	101261	22	10137	0	0	2409	0	2027	0	0	**122522**
Subtropical grass pasture	8533	3270	1618	211407	45	17554	0	0	1895	0	3511	0	0	**247833**
Lupins	3835	351	1903	7905	2	293	0	7824	0	0	59	0	0	**22172**
Oats	7278	641	4101	14816	5	414	0	32831	0	0	83	0	0	**60168**
Feedlot	1740	1170	0	17212	3	1138	0	0	0	0	228	0	0	**21490**

Table F.4 - Emissions breakdown, expressed as emissions intensity (EI) and total emissions quantity (EQ), across the different feedbase types at the Wickepin sheep enterprise

Feedbase type	Production of inputs	Transport-ation of inputs	Farm machinery operation	CH_4 enteric	CH_4 manure	N_2O excreta	N_2O N fertilisers	N_2O crop residue	N_2O pasture residue	N_2O leaching/ run off	Atmospheric deposition	CO_2 liming	CO_2 urea hydrolysis	TOTAL
EI (kg CO_2-e/kg LW)														
Annual pasture	0.29	0.06	0.07	6.95	0.00	0.69	0.00	0.00	0.12	0.00	0.13	0.62	0.00	**8.95**
Saltbush pasture	0.41	0.13	0.16	12.46	0.00	1.01	0.00	0.00	0.13	0.00	0.19	0.90	0.00	**15.38**
Lupins	11.36	0.41	4.10	10.32	0.00	0.77	0.20	7.98	0.00	0.00	0.17	2.29	0.00	**37.59**
Income crop stubble	0.00	0.00	0.00	14.14	0.00	0.76	0.00	0.00	0.00	0.00	0.14	0.00	0.00	**15.05**
EQ (kg CO_2-e/yr)														
Annual pasture	26410	5843	6060	630266	143	62461	0	0	11168	0	11906	56606	0	**810863**
Saltbush pasture	1217	373	462	37134	11	3015	0	0	382	0	569	2684	0	**45847**
Lupins	36350	1317	13112	33025	7	2465	630	25520	0	0	529	7320	0	**120276**
Income crop stubble	0	0	0	157966	48	8497	0	0	0	0	1575	0	0	**168087**

[a] Post-allocation of emissions between meat and wool production

Table F.5 - Emissions breakdown, expressed as emissions intensity (EI) and total emissions quantity (EQ), across the different feedbase types at the Dongara beef enterprise

Feedbase type	Production of inputs	Transport-ation of inputs	Farm machinery operation	CH_4 enteric	CH_4 manure	N_2O excreta	N_2O N fertilisers	N_2O crop residue	N_2O pasture residue	N_2O leaching/ run off	Atmospheric deposition	CO_2 liming	CO_2 urea hydrolysis	TOTAL
EI (kg CO_2-e/kg LW)														
Annual pasture	1.17	0.06	0.10	9.46	0.00	0.79	0.00	0.00	0.07	0.00	0.16	0.31	0.00	**12.12**
Tagasaste pasture	0.57	0.03	0.06	4.95	0.00	0.44	0.00	0.00	0.03	0.00	0.09	0.15	0.00	**6.32**
Subtropical grass pasture	0.97	0.04	0.09	7.21	0.00	0.53	0.04	0.00	0.06	0.00	0.11	0.20	0.03	**9.27**
Feedlot	1.21	0.12	0.00	4.46	0.00	0.75	0.00	0.00	0.00	0.00	0.15	0.00	0.00	**6.69**
EQ (kg CO_2-e/yr)														
Annual pasture	106476	5024	9455	861696	198	72147	0	0	6165	0	14429	28679	0	**1104269**
Tagasaste pasture	53508	2426	5961	461899	97	40858	337	0	2655	0	8205	13585	0	**589531**
Subtropical grass pasture	238208	9218	22116	1775287	423	130544	9026	0	14070	0	27011	48302	7040	**2281246**
Feedlot	13871	1359	0	51318	13	8625	0	0	0	0	1725	0	0	**76911**

Table F.6 - Emissions breakdown, expressed as emissions intensity (EI) and total emissions quantity (EQ), across the different feedbase types at the Lancelin beef enterprise

Feedbase type	Production of inputs	Transport-ation of inputs	Farm machinery operation	CH_4 enteric	CH_4 manure	N_2O excreta	N_2O N fertilisers	N_2O crop residue	N_2O pasture residue	N_2O leaching/ run off	Atmospheric deposition	CO_2 liming	CO_2 urea hydrolysis	TOTAL
EI (kg CO_2-e/kg LW)														
Annual pasture	1.10	0.15	0.09	11.75	0.00	0.76	0.00	0.00	0.16	0.00	0.19	0.00	0.00	**14.21**
Tagasaste pasture	0.53	0.06	0.18	12.60	0.00	0.98	0.00	0.00	0.09	0.00	0.22	0.00	0.00	**14.66**
Subtropical grass pasture	0.71	0.08	0.13	8.35	0.00	0.25	0.00	0.00	0.22	0.00	0.10	0.00	0.00	**9.84**
Feedlot	0.81	0.23	0.00	3.99	0.00	0.10	0.00	0.00	0.00	0.00	0.02	0.00	0.00	**5.15**
EQ (kg CO_2-e/yr)														
Annual pasture	29105	3841	2499	309943	51	19942	0	0	4321	0	5071	0	0	**374774**
Tagasaste pasture	43046	5207	14323	1028135	197	79626	0	0	7212	0	17865	0	0	**1195611**
Subtropical grass pasture	29207	3297	5232	343591	37	10454	0	0	8988	0	4142	0	0	**404946**
Feedlot	899	250	0	4405	1	109	0	0	0	0	22	0	0	**5686**

Table F.7 - Carbon footprints, expressed as emissions intensity (EI) and total emissions quantity (EQ), of the modelled no perennial scenarios, along with the EQ difference from the baseline scenario

Scenario	Production of inputs	Transp-ortation of inputs	Farm machinery operation	CH4 enteric	CH4 manure	N2O dung & urine	N2O N fertilisers	N2O crop residue	N2O pasture residue	N2O leaching/ runoff	Atmosphe-ric deposition	CO2 liming	CO2 urea hydrolysis	TOTAL
BREMER BAY - no perennial														
EI (kg CO_2-e/kg LW)	0.47	0.12	0.16	6.13	0.00	0.54	-	0.95	0.09	-	0.11	-	-	**8.58**
EQ (kg CO_2-e/ yr)	27059	7074	9446	355445	82	31468	-	55244	5330	-	6294	-	-	**497443**
Difference between baseline & mitigation scenario (%)	4.2%	4.6%	12.9%	0.8%	5.7%	6.5%	-	35.9%	23.8%	-	6.5%	-	-	**4.9%**
WICKEPIN - no perennial[a]														
EI (kg CO_2-e/kg LW)	0.59	0.07	0.18	7.95	0.00	0.70	0.01	0.24	0.10	-	0.13	0.59	-	**10.56**
EQ (kg CO_2-e/ yr)	63307	7163	19172	858530	209	76039	630	25520	11168	-	14501	66610	-	**1142850**
Difference between baseline & mitigation scenario (%)	-1.0%	-4.9	-2.4%	0.0%	-0.1%	-0.5%	0.0%	0.0%	-3.3%	-	-0.5%	0.0%	-	**-0.2%**
DONGARA – no perennial														
EI (kg CO_2-e/kg LW)	1.20	0.10	0.07	7.13	0.00	0.60	0.00	-	0.04	-	0.12	0.20	-	**9.46**
EQ (kg CO_2-e/ yr)	530059	45969	29858	3150360	713	263691	0	-	19757	-	52738	90567	-	**4183711**
Difference between baseline & mitigation scenario (%)	28.6%	155.0%	-20.4%	0.0%	-2.6%	4.6%	-100.0%	-	-13.7%	-	2.7%	0.0%	-	**3.3%**
LANCELIN – no perennial														
EI (kg CO_2-e/kg LW)	1.72	0.22	0.07	10.95	0.00	0.62	-	-	0.12	-	0.15	-	-	**13.84**
EQ (kg CO_2-e/ yr)	257438	33416	10274	1642992	303	93704	-	-	17638	-	22376	-	-	**2078141**
Difference between baseline & mitigation scenario (%)	151.8%	165.3%	-53.4%	-2.6%	5.5%	-14.9%	-	-	-14.1%	-	-17.4%	-	-	**4.9%**

[a] Post protein mass allocation of emissions between meat and wool production

Table F.8 - Carbon footprints, expressed as emissions intensity (EI) and total emissions quantity (EQ), of the feedlot scenarios, along with the EQ difference from the baseline scenario

Scenario	Production of inputs	Transp-ortation of inputs	Farm machinery operation	CH4 enteric	CH4 manure	N2O dung & urine	N2O N fertilisers	N2O crop residue	N2O pasture residue	N2O leaching/ runoff	Atmosphe-ric deposition	CO2 liming	CO2 urea hydrolysis	TOTAL
BREMER BAY - no feedlot														
EI (kg CO_2-e/kg LW)	0.43	0.11	0.13	6.77	0.00	0.55	-	0.68	0.08	-	0.11	-	-	**8.86**
EQ (kg CO_2-e/ yr)	24879	6549	7486	392374	87	31740	-	39626	4392	-	6348	-	-	**513481**
Difference between baseline & mitigation scenario (%)	-4.2%	-3.3%	-10.5%	11.3%	12.9%	7.5%	-	-2.5%	2.0%	-	7.5%	-	-	**8.3%**
WICKEPIN - feedlot Merino [a]														
EI (kg CO_2-e/kg LW)	0.59	0.07	0.18	7.82	0.00	0.69	0.01	0.23	0.11	-	0.13	0.62	-	**10.45**
EQ (kg CO_2-e/ yr)	63655	7569	19224	845017	207	74131	609	25233	11805	-	14132	66610	-	**1128192**
Difference between baseline & mitigation scenario (%)	-0.5%	0.5%	-2.1%	-1.6%	-0.9%	-3.0%	-3.3%	-1.1%	2.2%	-	-3.1%	0.0%	-	**-1.5%**
WICKEPIN - feedlot Merino SAMM [a]														
EI (kg CO_2-e/kg LW)	0.60	0.07	0.18	7.70	0.00	0.68	0.01	0.24	0.11	-	0.13	0.62	-	**10.34**
EQ (kg CO_2-e/ yr)	64514	7649	19635	831752	204	73087	630	26165	12126	-	13937	66610	-	**1116308**
Difference between baseline & mitigation scenario (%)	0.8%	1.5%	0.0%	-3.1%	-2.3%	-4.4%	3.3%	2.5%	5.0%	-	-4.4%	0.0%	-	**-2.5%**
DONGARA - feedlot pellet														
EI (kg CO_2-e/kg LW)	1.48	0.13	0.08	6.36	0.00	0.53	0.02	-	0.06	-	0.11	0.20	0.02	**8.99**
EQ (kg CO_2-e/ yr)	654216	58854	37532	2811647	653	233754	9363	-	24567	-	47687	90567	7040	**3975880**
Difference between baseline & mitigation scenario (%)	58.8%	226.5%	0.0%	-10.7%	-10.8%	-7.3%	0.0%	-	7.3%	-	-7.2%	0.0%	0.0%	**-1.9%**

Table cont…														
DONGARA - feedlot lupin														
EI (kg CO_2-e/kg LW)	1.24	0.04	0.08	6.36	0.00	0.59	0.02	-	0.06	-	0.12	0.20	0.02	**8.74**
EQ (kg CO_2-e/ yr)	550043	19581	37532	2811647	652	261474	9363	-	24567	-	53231	90567	7040	**3865696**
Difference between baseline & mitigation scenario (%)	33.5%	8.6%	0.0%	-10.7%	-10.9%	3.7%	0.0%	-	7.3%	-	3.6%	0.0%	0.0%	**-4.6%**
LANCELIN - feedlot lupin														
EI (kg CO_2-e/kg LW)	0.73	0.09	0.15	11.15	0.00	0.74	-	-	0.11	-	0.18	-	-	**13.14**
EQ (kg CO_2-e/ yr)	109533	13328	22054	1673478	286	111050	-	-	16199	-	27243	-	-	**1973170**
Difference between baseline & mitigation scenario (%)	7.1%	5.8%	0.0%	-0.7%	-0.7%	0.8%	-	-	-21.1%	-	0.5%	-	-	**-0.4%**

[a] Post protein mass allocation of emissions between meat and wool production

Table F.9 - Carbon footprints, expressed as emissions intensity (EI) and total emissions quantity (EQ), of the stubble grazing scenarios, along with the EQ difference from the baseline scenario

Scenario	Production of inputs	Transp-ortation of inputs	Farm machinery operation	CH4 enteric	CH4 manure	N2O dung & urine	N2O N fertilisers	N2O crop residue	N2O pasture residue	N2O leaching/ runoff	Atmosphe-ric deposition	CO2 liming	CO2 urea hydrolysis	TOTAL
BREMER BAY - no stubble grazed														
EI (kg CO_2-e/kg LW)	0.44	0.11	0.14	6.02	0.00	0.52	-	0.70	0.08	-	0.10	-	-	**8.12**
EQ (kg CO_2-e/ yr)	25738	6659	8167	348975	75	30104	-	40587	4394	-	6020	-	-	**470723**
Difference between baseline & mitigation scenario (%)	-0.9%	-1.5%	-2.4%	-1.0%	-2.2%	1.9%	-	-0.2%	2.1%	-	1.9%	-	-	**-0.7%**
WICKEPIN - no stubble grazed [a]														
EI (kg CO_2-e/kg LW)	0.58	0.07	0.17	7.91	0.00	0.73	0.01	0.23	0.11	-	0.14	0.62	-	**10.56**
EQ (kg CO_2-e/ yr)	62253	7477	18402	853786	207	79298	567	25162	11993	-	15140	66610	-	**1140896**
Difference between baseline & mitigation scenario (%)	-2.7%	-0.8%	-6.3%	-0.5%	-0.9%	3.7%	-10.0%	-1.4%	3.8%	-	3.8%	0.0%	-	**-0.4%**
WICKEPIN – no stubble grazed with added saltbush pasture [a]														
EI (kg CO_2-e/ kg LW)	0.57	0.07	0.17	7.85	0.00	0.73	0.01	0.23	0.11	-	0.14	0.62	-	**10.50**
EQ (kg CO_2-e/ yr)	62025	7573	18577	847345	206	79246	567	25162	11841	-	15133	66610	-	**1134285**
Difference between baseline & mitigation scenario (%)	-3.1%	0.5%	-5.4%	-1.3%	-1.8%	3.7%	-10.0%	-1.4%	2.5%	-	3.8%	0.0%	-	**-0.9%**

[a] Post protein mass allocation of emissions between meat and wool production

Table F.10 - Carbon footprints, expressed as emissions intensity (EI) and total emissions quantity (EQ), of the replacement rate scenarios, along with the EQ difference from the baseline scenario

Scenario	Production of inputs	Transport-ation of inputs	Farm machinery operation	CH4 enteric	CH4 manure	N2O dung & urine	N2O N fertilisers	N2O crop residue	N2O pasture residue	N2O leaching/ runoff	Atmosphe-ric deposition	CO2 liming	CO2 urea hydrolysis	TOTAL
BREMER BAY - increased maiden ewe														
EI (kg CO_2-e/kg LW)	0.41	0.10	0.13	5.35	0.00	0.45	-	0.62	0.07	-	0.09	-	-	**7.40**
EQ (kg CO_2-e/ yr)	25625	6631	8162	338522	74	28367	-	39076	4517	-	5673	-	-	**456648**
Difference between baseline & mitigation scenario (%)	-1.4%	-2.0%	-2.4%	-4.0%	-4.4%	-4.0%	-	-3.9%	4.9%	-	-4.0%	-	-	**-3.7%**
DONGARA - increased heifer														
EI (kg CO_2-e/kg LW)	0.91	0.04	0.08	7.01	0.00	0.56	0.02	-	0.05	-	0.11	0.20	0.02	**9.00**
EQ (kg CO_2-e/ yr)	412091	18036	37532	3187554	741	255003	9363	-	22832	-	51937	90567	7040	**4092696**
Difference between baseline & mitigation scenario (%)	0.0%	0.0%	0.0%	1.2%	1.2%	1.1%	0.0%	-	-0.2%	-	1.1%	0.0%	0.0%	**1.0%**
LANCELIN - increased on-farm heifer														
EI (kg CO_2-e/ kg LW)	0.65	0.08	0.14	10.68	0.00	0.69	-	-	0.13	-	0.17	-	-	**12.55**
EQ (kg CO_2-e/ yr)	102220	12585	22054	1671260	284	108718	-	-	20524	-	26817	-	-	**1964461**
Difference between baseline & mitigation scenario (%)	0.0%	-0.1%	0.0%	-0.9%	-1.1%	-1.3%	-	-	0.0%	-	-1.0%	-	-	**-0.8%**
LANCELIN - increased purchased heifer														
EI (kg CO_2-e/ kg LW)	0.54	0.07	0.12	8.89	0.00	0.58	-	-	0.11	-	0.14	-	-	**10.46**
EQ (kg CO_2-e/ yr)	102383	12630	22054	1670937	284	109178	-	-	20531	-	26909	-	-	**1964906**
Difference between baseline & mitigation scenario (%)	0.1%	0.3%	0.0%	-0.9%	-1.3%	-0.9%	-	-	0.0%	-	-0.7%	-	-	**-0.8%**

Table F.11 - Carbon footprints, expressed as emissions intensity (EI) and total emissions quantity (EQ), of the weaning rate scenarios, along with the EQ difference from the baseline scenario

Scenario	Production of inputs	Transport-ation of inputs	Farm machinery operation	CH4 enteric	CH4 manure	N2O dung & urine	N2O N fertilisers	N2O crop residue	N2O pasture residue	N2O leaching/ runoff	Atmosphe-ric deposition	CO2 liming	CO2 urea hydrolysis	TOTAL
BREMER BAY – weaning rate 75%														
EI (kg CO_2-e/kg LW)	0.70	0.15	0.17	9.43	0.00	0.82	-	0.53	0.17	-	0.16	-	-	**12.15**
EQ (kg CO_2-e/ yr)	21622	4770	5414	292228	63	25289	-	16520	5286	-	5058	-	-	**376249**
Difference between baseline & mitigation scenario (%)	-16.8%	-29.5%	-35.3%	-17.1%	-19.1%	-14.4%	-	-59.4%	22.8%	-	-14.4%	-	-	**-20.7%**
BREMER BAY – weaning rate 75% – match LW														
EI (kg CO_2-e/kg LW)	0.46	0.15	0.12	10.22	0.00	0.89	-	0.51	0.07	-	0.18	-	-	**12.60**
EQ (kg CO_2-e/ yr)	26845	8704	7202	592686	126	51350	-	29393	3772	-	10270	-	-	**730348**
Difference between baseline & mitigation scenario (%)	3.3%	28.7%	-13.9%	68.1%	63.2%	73.9%	-	-27.7%	-12.4%	-	73.9%	-	-	**54.0%**
BREMER BAY – weaning rate 120%														
EI (kg CO_2-e/ kg LW)	0.49	0.11	0.13	6.60	0.00	0.57	-	0.46	0.11	-	0.11	-	-	**8.59**
EQ (kg CO_2-e/ yr)	22990	5404	6221	311285	66	27086	-	21564	5072	-	5417	-	-	**405106**
Difference between baseline & mitigation scenario (%)	-11.5	-20.1	-25.6%	-11.7%	-14.2%	-8.3%	-	-47.0%	17.8%	-	-8.3%	-	-	**-14.6%**
BREMER BAY – weaning rate 120% - match LW														
EI (kg CO_2-e/ kg LW)	0.42	0.11	0.12	6.73	0.00	0.58	-	0.45	0.08	-	0.12	-	-	**8.61**
EQ (kg CO_2-e/ yr)	24572	6538	6813	390076	83	33830	-	25986	4686	-	6766	-	-	**499352**
Difference between baseline & mitigation scenario (%)	-5.4%	-3.3%	-18.6%	10.6%	7.4%	14.5%	-	-36.1%	8.9%	-	14.5%	-	-	**5.3%**

Table cont…														
WICKEPIN – weaning rate Merino[a]														
EI (kg CO_2-e/ kg LW)	0.58	0.07	0.18	7.81	0.00	0.70	0.01	0.23	0.10	-	0.13	0.60	-	**10.40**
EQ (kg CO_2-e/ yr)	64403	7582	19759	870607	212	77532	634	25606	11623	-	14786	67033	-	**1159779**
Difference between baseline & mitigation scenario (%)	0.7%	0.6%	0.6%	1.4%	1.3%	1.4%	0.6%	0.3%	0.6%	-	1.4%	0.6%	-	**1.3%**
WICKEPIN – weaning rate SAMM[a]														
EI (kg CO_2-e/ kg LW)	0.59	0.07	0.18	7.88	0.00	0.70	0.01	0.23	0.11	-	0.13	0.61	-	**10.50**
EQ (kg CO_2-e/ yr)	64320	7574	19740	866192	211	77202	634	25657	11611	-	14725	66967	-	**1154832**
Difference between baseline & mitigation scenario (%)	0.5%	0.5%	0.5%	0.9%	0.8%	1.0%	0.5%	0.5%	0.5%	-	1.0%	0.5%	-	**0.9%**
WICKEPIN –weaning rate Merino SAMM[a]														
EI (kg CO_2-e/ kg LW)	0.57	0.07	0.17	7.74	0.00	0.69	0.01	0.22	0.10	-	0.13	0.59	-	**10.30**
EQ (kg CO_2-e/ yr)	64446	7612	19654	878332	214	78291	627	25267	11707	-	14931	67380	-	**1168460**
Difference between baseline & mitigation scenario (%)	0.7%	1.0%	0.1%	2.3%	2.1%	2.4%	-0.5%	-1.0%	1.4%	-	2.4%	1.2%	-	**2.0%**
DONGARA – weaning rate														
EI (kg CO_2-e/ kg LW)	0.93	0.04	0.08	7.10	0.00	0.57	0.02	-	0.05	-	0.12	0.20	0.02	**9.13**
EQ (kg CO_2-e/ yr)	412334	18072	37532	3159977	734	253010	9363	-	22878	-	51538	90567	7040	**4063044**
Difference between baseline & mitigation scenario (%)	0.1%	0.3%	0.0%	0.3%	0.3%	0.3%	0.0%	-	-0.1%	-	0.3%	0.0%	0.0%	**0.3%**
LANCELIN –weaning rate														
EI (kg CO_2-e/ kg LW)	0.66	0.08	0.14	10.96	0.00	0.72	-	-	0.13	-	0.18	-	-	**12.86**
EQ (kg CO_2-e/ yr)	102353	12621	22054	1702184	292	111464	-	-	20515	-	27366	-	-	**1998849**
Difference between baseline & mitigation scenario (%)	0.1%	0.2%	0.0%	1.0%	1.4%	1.2%	-	-	0.0%	-	1.0%	-	-	**0.9%**

[a] Post protein mass allocation of emissions between meat and wool production