Greenhouse Gas Emissions from
Australian Beef Feedlots

Stephanie Kate Muir

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Department of Agriculture and Food Systems
Melbourne School of Land and Environment
The University of Melbourne

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Abstract

Emissions of the greenhouse gases, methane (CH$_4$) and nitrous oxide (N$_2$O) and the indirect greenhouse gas ammonia (NH$_3$) play an increasing role in public concern about the environmental impact of concentrated animal feeding operations, including feedlots. However, there is a lack of emissions measurements under typical commercial conditions and there is high uncertainty in the estimation. The lack of accurate measurements and baseline emissions also makes it difficult to evaluate efficiency of current managemange practices and identify the potential reductions under mitigation options. The objective of this study was to achieve increased understanding of greenhouse gas emissions from Australian beef feedlots, elucidating the biophysical factors controlling emissions from feedlot systems. Specifically, the study utilises measurements of greenhouse gas emissions undertaken at commercial feedlots in Australia using micrometeorological methods and integrates data collected from the feedlot operators into empirical models with the aim to identify and quantify the sources of variation in measured emissions between sites and seasons; test the validity the modelling approach used specifically for feedlots and quantify the link between animal behaviour and diurnal emissions patterns.

This study comprised two detailed modelling exercises. The first utilising the results of published studies to validate a range of equations for predicting enteric methane emissions and for predicting emissions of methane, nitrous oxide and ammonia from manure. The second modelling exercise utilised the results of measurements undertaken in two commercial Australian feedlots to evaluate a range of models under commercial conditions. Finally, the diurnal variation in micrometeorological measurements of CH$_4$ and NH$_3$ were examined in the context of animal feeding behaviour in order to examine implications for measurement accuracy and examine correlations between fluxes and behaviour.

This thesis indicates that the current Australian Inventory methodology for estimating greenhouse gas emissions from feedlots (enteric CH$_4$, manure CH$_4$, N$_2$O and NH$_3$) suffers from considerable inaccuracies. Although more accurate estimates of CH$_4$ emissions appear to be associated with utilising an equation based on ration composition, particularly carbohydrate fractions the current approach over estimates emissions considerably. Inaccuracies in prediction of emissions of N$_2$O and NH$_3$ are related primarily to the use of single “emissions factors” which do not adequately reflect the changes in potential emissions.
associated with changing environmental conditions.

This thesis also explored the contribution of CH$_4$, N$_2$O and NH$_3$ using IPCC default factor of 1.25% deposited NH$_3$ is lost as N$_2$O to total feedlot emissions, represented as CO$_2$-e. Initial estimates suggest that feedlot emissions were dominated by CH$_4$, with minor contributions of direct and indirect N$_2$O. However, based on the measurements nitrogenous greenhouse gases are predicted to contribute up to 52% of total CO$_2$-e. These results indicate that mitigation options to reduce feedlot emissions need to be applied to both enteric CH$_4$ and nitrogenous gas emission, particularly NH$_3$.

These more accurate estimates of greenhouse gas emissions will not only highlight issues with the current emissions inventory but will also assist the feedlot industry to identify mitigation strategies to take the benefit from the incentives for reductions in emissions under the carbon farming initiative (CFI).
Declaration

This is to certify that:

(i) the thesis comprises only my original work towards the PhD except where indicated in the Preface,

(ii) due acknowledgement has been made in the text to all other material used,

(iii) the thesis is fewer than 100,000 words in length, exclusive of tables, maps, bibliographies and appendices

Stephanie Muir

29th September 2011
Preface

The whole system feedlot measurements reported in this study were conducted as part of a larger project (FLOT .331, Greenhouse Gas Emissions from Australian Beef Cattle Feedlots, Meat and Livestock Australia), reported in Chen et al. (2009).

Measurements were undertaken using open-path lasers (University of Melbourne) and open-path FTIR (University of Wollongong). Zoe Loh (formally University of Melbourne), Douglas Rowell (University of Melbourne) and Stephanie Muir were primarily responsible for the operation of the open-path lasers during the reported field campaigns. The open-path FTIR was supplied by The University of Wollongong, and operated by Frances Phillips, Mei Bai and Travis Naylor (University of Wollongong) during the field campaigns. Data analysis for the emissions measurements using the WindTrax software package was handled primarily by Douglas Rowell, Frances Phillips and Mei Bai.

Measurements from these field campaigns are reported in Chen et al. (2009) and details about the open-path FTIR approach were discussed as part of the PhD thesis; Methane Emissions from Livestock Measured by Novel Spectroscopic Techniques, (Mei Bai, School of Chemistry, University of Wollongong, June 2010).

The results of these field campaigns are utilised Chapter 5, as a comparison with modelled results using data collected from the feedlots used in the measurement campaigns and in Chapter 6, with diurnal emissions profile compared with recorded behaviour. Excluding these specific measurements, the remainder of the work reported in this thesis was conducted by the author.
Acknowledgements

Whole system measurements of this kind would not have been possible without the whole of the FLOT.331 project team; Deli Chen, Mei Bai, Tom Denmead, David Griffith, Julian Hill, Sean McGinn, Travis Naylor, Zoe Loh, Frances Phillips and Doug Rowell. Additional thanks also to Dr. Sean McGinn and Mr. Trevor Coates (Agriculture and Agri-Food Canada) and Tom Denmead (CSIRO) for providing expertise and advice in the measurement methodology, micrometeorology and use of the open-path lasers. Extra thanks to Trevor and Zoe for the Melbourne to Queensland road trips and to the UOW team for going above physical chemistry and into the realm of agricultural science when required. Special thanks to Mei for the dumplings and Travis for excellent BBQ cookery and mid field trip morale boosts.

Thanks also to the owners and managers of the two feedlot sites, for opening their sites to our team for measurements, providing all the data I asked for and more, allowing video recording of pens of cattle and access to pens for static chamber measurements.

The FLOT.331 project, which provided the emissions measurement, was funded by Meat and Livestock Australia and the Australian Government (Australian Greenhouse Office). Meat and Livestock Australia provided additional funding in the form of a postgraduate top-up scholarship and operating funds, in conjunction with an Australian Postgraduate Award. Further financial support for conference attendance was provided by the Farrer Memorial Trust.

Huge thanks to my supervisory team, Dr. Julian Hill, Professor Deli Chen, Associate Professor Richard Eckard and Dr. Robert Edis for their support and guidance throughout the last 4 and a bit years; I’ve learnt a lot from you all. Special thank to Julian for dealing with numerous panics, general confusion, many phone calls and multiple drafts with good humour and encouragement. I’m sure it’s been as “good for your soul” as processing pages and pages of feedlot data into a single number was for mine. Thanks also to Dr. Peter Ades for biometrical advice during the revisions phase.

Much appreciation goes to my friends, team mates and Dookie campus work mates for understanding that sometimes I really did just have to stay home and work on my thesis instead of doing the “much more enjoyable than writing” activity you suggested. Thanks also
for continual encouragement in my quest to be a Doctor of cow “emissions”, even though exactly what that was and why I wanted to spend all this time working on it probably wasn’t always clear. Extra special thanks to the fellow PhD candidates among my friends, whether in the next office or across the country, for all the advice, encouragement and understanding.

Finally, many many thanks to my family, for constantly believing I was nearly finished, even when I didn’t think so. Dad, I appreciate any and all attempts to “open the gate” for me, not only in the last few years, but for a lifetime. Mum, thank you for the data entry, but more importantly for looking after me when I didn’t feel like being a grown up and looking after myself. Lucy, thank you for your support even though spending so many years studying cows doesn’t seem normal. I love you all.

Last but not least, thanks to Jack for waggly tail, wet nose and keeping me company while I was working (e.g. sleeping in front of the fire). Some days a walk is exactly what I need even when it’s the last thing I think I want.
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Abbreviations

ABARE- Australian Bureau of Agricultural and Resource Economics

ADF- Acid Detergent Fibre

ADFi- Intake of Acid Detergent Fibre

ADG- Average Daily Gain

AGO- Australian Greenhouse Office

ALFA- Australian Lot Feeders Association

ANOVA- Analysis of Variance

bLS- Backward Lagrangian Stochastic (model)

C- Carbon

CELLi- Intake of Cellulose

CFI- Carbon Farming Initiative

CH₄- Methane

CHO- Carbohydrate

CNCPS- Cornel Net Crude Protein System

CO₂- Carbon Dioxide

CO₂-e- Carbon Dioxide Equivalent

CP- Crude Protein

CPi- Crude Protein Intake

CV- Coefficient of Variation

DCCEE- Department of Climate Change and Energy Efficiency

DM- Dry Matter
DMI- Dry Matter Intake
DOF- Days on Feed
EF- Emissions Factor
FATi- intake of Fat
FTIR- Fourier Transfer Infra-Red
GE- Gross Energy
GEI- Gross Energy Intake
GWP- Global Warming Potential
HEMIi- Intake of Hemicellulose
$H_2$- Hydrogen Gas
IPCC- Intergovernmental Panel on Climate Change
KCL Potassium Chloride
LIGi- Intake of Lignin
LSD- Least Significant Difference
LWG- Liveweight gain
LWT- Liveweight
MAF- Ministry of Agriculture and Forestry
MCF- Methane Conversion Factor
ME- Metabolisable Energy
MEi- Intake of Metabolisable Energy
MLA- Meat and Livestock Australia
MMS- Manure Management System
N- Nitrogen
NDF- Neutral Detergent Fibre

NE- Net Energy

NGGIC- National Greenhouse Gas Inventory Committee (Australia)

NH₃- Ammonia

N₂O- Nitrous Oxide

NOₓ- Nitrogen Oxide

NO₃- Nitrate

NRC- National Research Council

O₂- Oxygen

OM- Organic Matter

OP- Open-path (laser)

RIRDC- Rural Industries Research and Development Corporation

SCA- Standing Committee on Agriculture

SD.- Standard Deviation

SE- Standard Error

SED- Standard Error of Difference

SEM- Standard Error of Mean

SF₆- Sulfar-hexaflouride

USEPA- United States Environmental Protection Agency

UNFCCC- United National Framework Convention on Climate Change

VFA- Volatile Fatty Acid

VS- Volatile Solids
Associated Publications

Measurement


Chapter 4 and 5


Chapter 6

Chapter 1. Introduction

1.1 Global Demand for Beef

Global population growth, particularly in developing nations, and increasing incomes (in developed countries) has increased the global demand for animal products. Between 1962 and 2003 consumption of meat in developing countries increased from 10 to 29 kg/person/year as a result of these trends, whilst milk consumption increased by 20 kg/person/year to 48 kg/person/year (Steinfeld et al. 2006a). Some of the most dramatic increases in consumption have been in China and East Asia, who are likely to continue to import increasing amounts of livestock products in order to meet domestic demand (Steinfeld et al. 2006a). In the period between 1980 and 2002 total protein supplied by livestock products in Asia increased by 130% (Steinfeld et al. 2006a). This trend has resulted in expansion and technological change in the livestock sector. In terms of beef production the major change has been a growth in production intensity, with increased use of feed cereals, advances in genetics and feeding systems, animal health protection and enclosure of animals (Steinfeld et al. 2006b). The global demand for increased meat production is expected to increase faster than can be supported by the development of arable land, meaning that production is likely to continue to intensify (Verge et al. 2008).

Australia is one of the world’s largest beef exporters (Pritchard 2006), although domestic consumption still accounts for more than 30% of Australia beef production (Bindon and Jones 2001). Australia exports both beef and live animals to a variety of countries, with over 600,000 tonnes of beef exported annually (Morgan 2010). The majority of this meat (over 230,000 tonnes) is imported by Japan, with the US and Korea also significant importers of Australian beef (150,000 and 80,000 tonnes respectively; Morgan 2010). According to the Australian Lot Feeders Association (ALFA 2008) the feedlot sector now accounts for 80% of beef sold in domestic supermarkets with increased demand for grain fed product (ALFA 2008).

1.2 Feedlot Systems

The benefit of feedlot finishing is that feed quality and quantity (as delivered to the animals) can be closely controlled, resulting in greater production efficiency, and more consistent quality and quantity of product. Feedlots specialise in feeding cattle on high energy finishing
rations (Ellis et al. 2009), and are one of the most efficient systems in terms of producing meat (Fiala 2008).

Australia’s beef production system has historically been based around pastoral production, particularly in the north, where cattle would be grown and finished on pasture or native forages (ABARE and MAF 2006; Charmley et al. 2008). An increasing proportion of cattle are being finished through feedlots, which allows increased control over feed quality (which can be extremely variable, particularly in Northern Australia) and therefore a more consistent quality and quantity of product. Although lot feeding of beef cattle in Australia developed in the mid 1960’s (BAE 1976), the expansion of the Australian feedlot industry can be linked to rapid increase in Asian beef consumption through the mid 1990s, access to the Japanese and Korean markets encouraged the expansion of the grain fed sector (ABARE and MAF 2006; Bindon and Jones 2001; Pritchard 2006). In 1999 Kurihara et al. (1999) reported that approximately 40% of Australian beef slaughter stock was finished in a feedlot for between 70 and 300 days). In 2002 there were about 600 accredited feedlots in Australia, with capacity for approximately 860,000 cattle (ABARE and MAF 2006).

Typically feedlots consist of multiple pens with watering and feeding sites, each pen holding more than 100 animals for several months (Miller and Berry 2005). The management of nitrogen (N) in excreta has traditionally been considered one of the major issues associated with feedlot systems, with considerable potential for run-off, affecting water quality (Miner et al. 2000) as well as volatilisation of ammonia (NH\textsubscript{3}) and emission of nitrous oxide (N\textsubscript{2}O) (Miller and Berry 2005). Beef cattle feedlots face considerable pressure to improve management of manure to avoid environmental damage and effects on human health (Archibeque et al. 2007; Cole et al. 2005; Miller and Berry 2005; Pandrangi et al. 2003).

Fiala (2008) reports that beef production has the most severe environmental impact of the confined animal production systems (e.g. pork and poultry) due to enteric methane (CH\textsubscript{4}) production. The perception of feedlots as not only environmentally damaging but also large emitters of greenhouse gases will put increasing pressure on individual operators and the industry as a whole, to reduce and manage pollutants- including emissions of CH\textsubscript{4}, N\textsubscript{2}O and NH\textsubscript{3}.

**1.3 The Global Emissions Problem**

Despite increasing demand for beef (and animal products as a whole) “an increasingly environmental consciousness in society requires action by the livestock industry on environmental problems” (Ogino et al. 2004); none more so than emissions of greenhouse
gas. Steinfeld et al. (2006a) reports that at every stage of the livestock production process substances are emitted into the environment, contributing to atmospheric pollution as well as degradation of waterways; a perception which poses a considerable challenge to livestock production worldwide.

The United Nations Framework Convention on Climate Change (UNFCCC) regards the main anthropogenic greenhouse gases as carbon dioxide (CO$_2$), CH$_4$ and N$_2$O (Steinfeld et al. 2006a). However, NH$_3$ is also a significant cause of environmental degradation, and an indirect source of N$_2$O. The direct warming potential of CO$_2$ is the greatest of the three, due to higher atmospheric concentration and larger emitted quantities. Methane is considered the second most important greenhouse gas, with an atmospheric lifetime of 9-15 years. Methane is 21-25 times more effective than CO$_2$ at capturing heat, resulting in a global warming potential (GWP) of 21-25. Nitrous oxide is present in the atmosphere in very small amounts; however it has a GWP of 296-310, and a very long atmospheric lifetime (114 years; Kebreab et al. 2006; Steinfeld et al. 2006a). Once volatilised NH$_3$ can be detrimental to the environment in a number of ways, however its role as a secondary (or indirect) source of N$_2$O once deposited outside the source area is a considerable concern.

Livestock emit large amounts of these three direct greenhouse gases; CO$_2$ from respiratory pathways, CH$_4$ from digestive processes (enteric fermentation in ruminants) and manure, and N$_2$O from manure. In the United States, it is estimated that the production of one kg of beef resulted in emissions of the equivalent of 14.8 kg CO$_2$ (Fiala 2008). For countries such as Australia and New Zealand, which have large agricultural industries, CH$_4$ emissions from sheep and cattle represent a substantial contribution to the total greenhouse gas emissions (Griffith et al. 2008; Peters et al. 2010). McGinn et al. (2009) report that emissions of CH$_4$ from enteric fermentation contribute 12 to 17% of total (combined anthropogenic and natural) global emission, supported by a number of authors including Hegarty et al. (2007) and Lassey (2007). Methane from livestock is reported to be 85- 86 Tg annually (McGinn et al. 2006b; Steinfeld et al. 2006a). Verge et al. (2008) estimated beef cattle to account for 68% of total CH$_4$ from agriculture. Livestock operations are also prominent sources of atmospheric NH$_3$. McGinn et al. (2003) suggests that livestock manure contributes 81% of “industrial” NH$_3$ emissions in Canada.

In Australia, agriculture accounts for 16% of total anthropogenic greenhouse gas emissions (Charmley et al. 2008), with livestock responsible for about 70% of agriculture sector emissions (Peters et al. 2010). The intensive nature of feedlot operations makes them a significant point source of emissions, and more likely than a grazing system to face scrutiny
regarding environmental pollution, although feedlots are estimated to contribute 3.5% of
Australian livestock CH\textsubscript{4} emissions (compared with 58% from grass-fed beef; ALFA 2008).

1.4 Issues for the Feedlot Sector

Major issues for the Australian feedlot sector include grain price, market access, animal
welfare and disease, climate change and increased government support for grain based bio-
fuel production (ALFA 2008). Gurian-Sherman (2008) reports that the primary issue related
to the management of animals in a concentrated environment is the production of large
amounts of manure. However, energy use and greenhouse gas emissions are likely to become
more significant issues in the move towards a carbon constrained future. The nature of
intensive beef production results in a considerable amount of “energy” (fossil fuel/ carbon)
expenditure in feed crop production, and in the preparation and delivery of feeds to the
half the energy expenditure for livestock production is for feed production, increasing to
nearly all for intensive beef production. Howden and Reyega (1999) suggest that emissions
associated with the production of grain consumed by feedlot cattle could be over four times
those emitted directly by the animal (when emissions from grain harvesting, processing and
transport of grains are included). Peters \textit{et al.} (2010) suggest that increasing proportion of lot
fed beef in the Australian red meat industry is favourable since the additional greenhouse
gases emitted produced in producing and transporting grain feed is offset by the increased
efficiency of meat production.

Despite the volatile political environment which surrounds proposals aimed at reducing
greenhouse gas emissions it is likely that there will be constraints on carbon pollution in the
future. Although it is likely that agriculture will be excluded from these schemes (at least
initially), it will be increasingly important to benchmark and monitor emissions. However,
current methodologies for estimating emissions from enteric fermentation and manure
decomposition lack validation, particularly for feedlot environments (Stackhouse \textit{et al.} 2011).

1.5 Feedlot Emissions Balance

As reported by Fiala (2008) it is commonly thought that beef production systems are more
environmentally damaging than monogastric production systems (pigs and poultry) due to the
contribution of enteric CH\textsubscript{4}. Methane is thought to be the most important greenhouse gas from
feedlot systems (ALFA 2008), however, this stems from the perception of ruminant emissions
being dominated by CH\textsubscript{4}. Little consideration is often given to direct and secondary (from
NH\textsubscript{3}) emissions of N\textsubscript{2}O.
Chapter 1. Introduction

Based on National Inventory Methodology (AGO 2006) estimates, a 600 kg feedlot steer, growing at 1.5 kg/day, consuming a ration (at approximately 10 kg DM/day) comprising a minimum of 70% grain emits 176 g CH$_4$ from enteric fermentation, (109 g CH$_4$ using the IPCC Tier II approach), 3.4 g CH$_4$ from manure, 7.4 g N$_2$O and 70 g NH$_3$ daily, supporting the observation of Fiala (2008), regarding the dominance of CH$_4$ from livestock operation emissions. However, the accuracy of models has not been evaluated in detail from feedlot systems. Early measurements of NH$_3$ from Australian feedlot systems have been reported at up to 253 g/head/day (Loh et al. 2008) and 324 g/head/day (Chen et al. 2009). Whilst emissions of CH$_4$ are reported at 146 to 166 g/head/day (Loh et al. 2008) and 63.8 to 138 g/head/day (Chen et al. 2009). Although predicted CH$_4$ emissions are within the range reported, predicted NH$_3$ is significantly lower than measured values- suggesting that N may contribute to feedlot emissions to a greater extent than could be assumed using methodology estimates.

1.6 Summary

The Australian beef industry is heavily reliant on exports earnings, particularly from sales to the Asian markets, which encourages finishing increasing number of cattle on grain in a feedlot. Ruminant production systems are significant sources of greenhouse gas emissions, the production of which is likely to come under increased scrutiny as other industries face regulations to reduce emissions. The ability to reduce emissions, and examine management impacts on emissions is limited by lack of benchmark data and models which have not been adequately validated for feedlot environments.

1.7 Objectives

The objective of this study was to achieve increased understanding of greenhouse gas emissions from Australian beef feedlots, elucidating the biophysical factors controlling emissions from feedlot systems. Specifically, the study utilises the first measurements of greenhouse gas emissions undertaken at commercial feedlots in Australia using micrometeorological methods and integrates data collected from the feedlot operators into empirical models with the aim to;

- Identify and quantify the sources of variation in measured emissions between sites and seasons.
- Test the validity the modelling approach used specifically for feedlots, and compare predicted emissions to measured emissions
- Quantify the link between animal behaviour and diurnal emissions patterns
Chapter 2. Literature Review

2.1 Introduction

In all animal production systems, the most important sources of the greenhouse gases (CH$_4$ and N$_2$O) and indirect greenhouse gases/pollutants (NH$_3$) are the animal themselves, animal housing, storage and treatment areas for manure and waste water and spreading of manure and chemical fertilizers (Hartung and Monteny 2000). The concentrated nature of feedlot operations makes them a significant point source of emissions of CH$_4$, N$_2$O and NH$_3$. These gases arise from C and N cycling through the system, CH$_4$ as a product of anaerobic bacterial fermentation of carbohydrates in feeds and excreta, whilst N$_2$O is formed from denitrification and nitrification (Hartung and Monteny 2000) directly from N in the excreta, and indirectly from deposited NH$_3$.

This literature review aims to outline the mechanisms of the main greenhouse gas emissions from feedlot systems, and discuss methods used for both measurement and modelling these emissions. In general, the perceptions of feedlot systems is of a CH$_4$ dominant emissions profile, however the following review outlines the significant contribution to emissions from NH$_3$ and potential indirect N$_2$O. This emissions balance needs to be considered in the light of the current policy initiatives surrounding agricultural emissions (The Carbon Farming Initiative, Australian Government 2010a) where abatement and mitigation can be used to generate carbon credits.

2.2 Emissions Sources in the Feedlot

Feedlot production systems specialise in feeding cattle on high energy finishing rations (Ellis et al. 2009), and are one of the most efficient systems in terms of producing meat (Fiala 2008). The benefit of feedlot finishing is that feed quality and quantity (as delivered to the animals) can be closely controlled, resulting in greater production efficiency, and more consistent quality and quantity of product. Typically, a feedlot consists of multiple pens, with watering and feeding sites, each pen holding more than 100 animals for several months (Miller and Berry 2005).

Livestock emit large amounts of these three direct greenhouse gases (Figure 2.1); CO$_2$ from respiratory pathways, CH$_4$ from digestive processes (enteric fermentation in ruminants) and manure, and N$_2$O from manure. Therefore, within a feedlot- CH$_4$ is emitted from enteric fermentation and manure and N$_2$O directly from manure. Ammonia (although not strictly a
greenhouse gas) volatilises from feedlot manure, and following deposition, becomes a secondary (indirect) source of N₂O.

Manure storage areas (solid, composting stacks and effluent systems), and the spreading of manure in the field are further sources of greenhouse gas emissions from feedlot systems (including NH₃) - however this study considers only direct emissions from the animals and manure deposited in the pen (Figure 2.1).

![Figure 2.1 Emissions sources in a feedlot. This study focuses on the direct emissions from livestock and emissions from manure.](image)

2.2 Enteric CH₄ Emissions Process

The biological advantage of ruminants is that they can convert poor quality fibrous feeds, such as poor quality grass, straw and waste products to utilisable energy for growth, milk, meat and fibre production. This review does not intend to be an exhaustive description of rumen fermentation processes; however an overview of the processes which enable the conversion of the poor quality feeds to a usable energy source is necessary to describe the production of enteric CH₄. The processes which enable the utilisation of these feeds are mastication, rumination and fermentation (Pinares-Patino et al. 2000), made possible by the gastrointestinal structure of the ruminant. Mastication during feeding reduces particle size of
the feed stuff, which is further reduced by rumination; whereby digesta is regurgitated, the liquids swallowed, the solids re-chewed and returned to the rumen (Hungate 1975).

The rumen (often described as the first “stomach” of ruminant animals) is effectively a large fermentation chamber, where plant compounds are fermented with the actions of bacteria, protozoa and fungi (Immig 1996). Methane emissions from ruminants come from enteric fermentation in the rumen, the breakdown of carbohydrates by microbes (Equation 2.1) (Saggar et al. 2004b).

**Equation 2.1** Methane production from carbohydrate (Saggar et al. 2004b)

\[
C_6H_{12}O_6 \rightarrow 3CO_2 + 3CH_4
\]

Rumen fermentation produces volatile fatty acids (VFA), CH$_4$ and CO$_2$ (Miller 1995). The formation of VFA depends on the individual fermentations of different microbial species, which also produce hydrogen (H$_2$). Methane producing organisms use H$_2$ to reduce CO$_2$ to CH$_4$ (Miller 1995; Equation 2.2). This process requires successive action of four different classes of micro-organisms which degrade complex molecules into simple compounds (Moss 1993). Microbial activity in the rumen and through the action of enzymes in saliva hydrolyses dietary organic matter to amino acids and simple sugars. These products are then fermented to volatile fatty acids, hydrogen and CO$_2$. Carbohydrate fermenting bacteria generally do not produce CH$_4$. However, these bacteria produce formate, hydrogen (H$_2$) and carbon dioxide (CO$_2$) which act as substrates for Methanogenic bacteria (Moss 1993). Hydrogen is the principal substrate for methanogenesis in the rumen, with formate the substrate for approximately 18% of rumen CH$_4$ production (O’Mara 2004).

**Equation 2.2** Reduction of CO$_2$ to form CH$_4$ (O’Mara 2004)

\[
CO_2 + 4H_2 \rightarrow CH_4 + 2H_2O
\]

The process of CH$_4$ formation in the rumen is vital as it has an important role in removing hydrogen from the rumen, which can become toxic to microbial growth (Hegarty et al. 2007; Johnson and Johnson 1995). Methanogenic bacteria play an important part in maintaining low partial pressure of H$_2$ in the rumen, removing H$_2$ and reducing inhibition of fermentation associated with a build up of H$_2$ (Immig 1996). Partial pressure of H$_2$ significantly influences rate of rumen methanogenesis and the range of VFA produced (Hegarty and Gerdes 1998). The concentration of dissolved H$_2$ (partial pressure of H$_2$) influences fermentation pathways, which use of produce H$_2$ (Janssen 2010). Janssen (2010) reported that at high concentrations
of H₂, formation of H₂ becomes thermodynamically unfavourable, such that the free energy change of substrate transformation via H₂ producing pathways is less favourable than fermentation to other products. Microorganisms that can change fermentation pathways can do so in response to even small differences in free energy change, to switch to pathways which are more thermodynamically favourable. This effectively reduces fermentation to H₂, at high H₂ concentrations.

The end products or fermentation are primarily CO₂, CH₄, short chain fatty acids, acetate, propionate and butyrate (Immig 1996). Relative proportions of VFA produced from fermentation are influenced primarily by diet type, and ratio of fermentable structural carbohydrate (CHO) to starch and quickly degradable CHO (Lassey 2007). If rumen fermentation patterns are shifted from acetate to propionate dominant processes hydrogen and CH₄ production will be reduced (Takahashi 2001). An inverse relationship exists between propionate and CH₄ production in the rumen (Tamminga 1992). Diets which are high in fibre and low in structural carbohydrate (CHO) promote acetate formation and increase CH₄ production (Yan et al. 2000).

The rumen, rather than the hindgut is the primary site of fermentation in the ruminant digestive system. However hindgut fermentation can increase in ruminants when substrates decrease in rumen degradability (high starch and fat). Immig (1996) suggest (based on stoichiometry) that between 28 and 592 mmol of CH₄ will be produced in the hindgut of steers fed diets high in corn starch, but that the majority of this is absorbed into the blood stream from the hindgut and released via the lungs.

A large amount of fermentative gas is produced in the rumen almost constantly, with an estimated 2 l/min in a 400 kg animal (Doughtey 1968); a 500 kg cow produces around 800 litres of H₂ daily (Immig 1996). These gases are principally CH₄ and CO₂; CO₂ comprises 60% and CH₄ 35% of total gas in the reticulorumen. This rate of gas production requires an effective elimination method (Doughtey 1968). Methane is released primarily through eructation (via the mouth and nose), although small amounts are absorbed by the digestive system into the blood and realised via the lungs (Doughtey 1968, Immig 1996) or released via the anus. Eructation is a reflex action, stimulated by gaseous distortion of the reticulorumen, provided the gas formation in the rumen allows the formation of pockets of gas. If the rumen contents are in the form of a foamy mass, where the gas forms small bubbles throughout the substrate inhibition of eructation can occur (Doughtey 1968). There has been little research into flatus mediated release of CH₄, however it has been observed as approximately 2% in sheep (Saggar et al. 2004b).
2.2.1 Enteric CH$_4$ Emissions from Lot Fed Cattle

Methane is produced as a result of microbial fermentation in the rumen which breaks down cellulose and other large molecules (Jentsch et al. 2007; Johnson and Ward 1996; Johnson and Johnson 1995; Van Nevel and Demeyer 1996). Methane is also produced by the anaerobic fermentation of manure. The amount of CH$_4$ lost (as a percentage of gross energy, and as an absolute value, from feedlot cattle is lower than that from grazing beef and dairy cattle. High yielding dairy cows are reported to produce more than 100 kg CH$_4$/year from enteric fermentation alone (equating to around 270 g CH$_4$/day; Saggar et al. 2004b), this is more than twice the value assigned to other (non-lactating) cattle. Cattle are estimated to lose 2-15% of gross dietary energy as eructated CH$_4$, although this figure varies with diet (Kurihara et al. 1997). There is considerable variation in the amount of CH$_4$ emitted by feedlot cattle dependent primarily on level of grain in the ration. Beauchemin and McGinn (2005) examined corn and barley based diets, at different levels of grain (low grain backgrounding and high grain finishing) measuring emissions of 170 g/day (corn based backgrounding diet), 129.7 g/day (barley based backgrounding diet), 62.1 g/day (corn based finishing diet) and 80.4 g/day for the barley based finishing diet. The lower emissions observed with the higher grain diets are inconsistent with those observed by Boadi et al. (2002) with a high grain diet (64.3g/day), however the lower grain ration measured by Boadi et al. (2002) demonstrated lower emissions (91.4 g/day) than those reported by Beauchemin and McGinn (2005). Johnson and Johnson (1995) reports that cattle lose approximately 6% of dietary gross energy as CH$_4$. Methane production from feedlot cattle as a percentage of gross energy intake (GEi) has been reported as 0.9 to 6.9% GEi on a low forage:grain diet and 0.7 to 4.9% GEi (Boadi et al. 2004b) and 2.8-7.3% GEi (Beauchemin and McGinn 2005).

Predominantly, research focussing on CH$_4$ emissions from lot fed or growing beef cattle originate from North America or Canada, where lot feeding makes up a significant proportion of the beef industry (Stackhouse et al. 2011; Verge et al. 2008). In the United States, between 11 and 17 million steers are housed in feedlots depending on the time of year (Stackhouse et al. 2011). McGinn et al. (2007) and Loh et al. (2008) report higher emissions from both Canadian (214 g/head/day) than Australian (166 g/head/day) feedlots, similarly Hegarty et al. (2007) measured emissions of 142 and 190 g/head/day from feedlot cattle. Van Haarlem et al. (2008) measured CH$_4$ emissions of 118.1 g/head/day, consistent with the results of Beauchemin and McGinn (2005) as an estimate for feedlot cattle (at the higher forage proportion diet).

There are also limited studies focusing on CH$_4$ production from tropical type (Bos Indicus) cattle, particularly under feedlot conditions. However, Kurihara et al. (1999) measured
emissions of 134 g/day for *B. Indicus* cattle fed high grain rations (Hunter 2007). As *B. Indicus* cattle make up large proportion of feedlot cattle in Australia, particularly in the north, more extensive investigation will be required to determine differences in emissions between tropical and temperate breeds fed high grain diets.

Differences in enteric CH$_4$ emissions between lot fed and grazing cattle are due, in the most part, to differences in ration composition. However, animal characteristics, such as breed, size, intakes and growth rates can contribute to differences in emissions, both between individual animals and between lot fed and grazing cattle.

### 2.2.2 Animal Factors Influencing Enteric CH$_4$ Emissions

In broad terms, CH$_4$ production is determined by the quality and quantity of feed consumed (Hegarty *et al.* 2007); however other animal characteristics moderate intake, fermentation and energy utilisation impacting on CH$_4$ emissions. Hegarty (2001) suggest that the key attributes of the animal population which may influence enteric fermentation include size (and selection for smaller size), feed efficiency (feed conversion efficiency or residual feed intake) and digesta kinetics.

In extensive examination of experiments measuring CH$_4$ production in ruminants, (based on Blaxter and Clapperton 1965); Pinares-Patino *et al.* (2003) determined significant variation in the amount of CH$_4$ produced by sheep consuming the same diet. From this, they suggested that these differences may be due to differences in size or weight or breed of the animal. Differences in metabolism and rates of fermentation may be related to breed differences; *Bos Taurus* and *B. Indicus* cattle have been observed to exhibit differences in metabolism (Kurihara *et al.* 1999). This can lead to over estimation of energy requirement by algorithms used for predicting CH$_4$ output (Hegarty 2004). *B. Indicus* cattle were found to have a faster fermentation rate and shorter retention time in the rumen than *B. Taurus* cattle (Hegarty 2004; Hunter and Siebert 1985). However, it is clear that these differences in metabolism may influence potential enteric emissions, particularly when animals are fed the same ration.

### 2.2.3 Ration Composition Effects

Altering ration composition has a significant effect on enteric CH$_4$ output. In short, altering the diet in a way which influences hydrogen production (e.g. reduced fermentable CHO), or which influences hydrogen capture (e.g. reduced pH) will reduce CH$_4$ emissions. More broadly, a diet which favours propionate (high starch/ high grain) production from fermentation over acetate (high fibre/ high forage) tends to result in decreased CH$_4$. 
The high grain, high digestibility rations utilised in feedlot operations, result in a fermentation pattern favouring propionate production and reduced emissions compared with grass fed cattle. This is the primary reason for reduced emissions (per kg of dry matter intake (DMI)) in feedlot or high grain rations. This effect is highlighted by the results of Beauchemin and McGinn (2005) comparing “backgrouding” (high forage) with “finishing” (high grain) rations, observing emissions of 129 to 170 g/head/day for the high forage (70 % silage) rations, compared with 62 to 80 g/head/day for the high grain (9% silage) rations. Similarly, Lovett et al. (2003) observed maximal CH₄ emissions on a ration comprising 40% forage: 60% concentrate (270 L CH₄/head/day) compared with a ration comprising 10% forage: 90% concentrate (170 L CH₄/head/day). Boadi et al. (2004b) observed higher emissions on a ration with high grain (83.5% barley grain) compared with high forage (41% barley silage, 41% barley grain), with measured emissions of 127.9 g/head/day on the high grain ration, and 90 g/head/day on the low grain ration, however the low grain ration also produced a lower DMI and had a higher fat content (in the form of sunflower seeds) than the high grain diet, contributing to lower observed emissions.

At a smaller scale, individual grains/ silage types e.g. maize/ corn compared with wheat or barley) with differential fermentation characteristics can also alter CH₄ output; due to starch influences on acetate: propionate ratio. Beauchemin and McGinn (2005) compared maize (corn) and barley grains and silage, at both low (30%) and high (91%) grain concentrations. The maize grain contained 66% starch, compared with 54.5% in the barley grain. In the high grain rations measured CH₄ emissions of 62.1 g/head/day (corn) and 80.4 g/head/day (barley) were reported. However, they attribute this to numerical differences in DMI. This does indicate potential changes in emissions moderated by grain type. Beauchemin and McGinn (2005) suggests that the difference between barley and corn diets in the finishing phase may be related to a change in the site of digestion from the rumen to the small intestine (in the corn diets), or the result of a change in rumen pH.

Kurihara et al. (1999) and Hunter (2007) reported the results of a comparison between tropical grasses and grain fed to B. Indicus beef cattle, determining emissions of 94.5 g/head/day for Angelton grass, 215 g/head/day for Rhodes grass and 134 g/head/day for high grain. Similarly to the Rhodes grass CH₄ emission observed by Kurihara et al. (1999), McCaughey et al. (1997) determined emissions between 242.2 and 306.7 L/head/day for steers grazing pasture clearly demonstrating the impact of poorer quality feeds on emissions from growing beef cattle. Chaves et al. (2006) compared alfalfa (lucerne), a more digestible, lower neutral detergent fibre (NDF) and higher crude protein (CP) forage, with pasture (grass). They determined higher absolute CH₄ emissions and higher emissions as a percentage
of GEI, and of DMI from the alfalfa sward. This is contrary to expected, given the higher quality of the alfalfa forage, although similar to the results of Benchaar et al. (2001). These differences were attributed by the authors to compositional and maturity differences in forage types. Increased assessment of emissions on native and tropical forages is necessary particularly for the Australian beef industry, where a large proportion of cattle are grazed on extensive pasture prior to feedlot finishing. Further, differences in the types of forages used as supplements in feedlots (although at a relatively low rate), may influence rate of emissions.

2.3 Methane Emissions from Manure

The second source of CH$_4$ from animal systems is derived from fermentation of organic carbon (C) sources in excreta (also described as volatile solids; Kreuzer and Hinderichsen 2006).

2.3.1 Methane Emissions Process

Methane is formed in manure by the anaerobic decomposition of organic matter (Steed and Hashimoto 1994). Methanogenic fermentation occurs under anaerobic conditions, with low sulphate and NO$_3$ concentrations (Saggar et al. 2004b). Organic wastes are biologically degraded (under anaerobic conditions) to CO$_2$, CH$_4$, N$_2$, NH$_4$ and H$_2$. CH$_4$ will only be produced under anaerobic conditions with low redox potential (Equation 2.1; Saggar et al. 2004b). A 600 kg feedlot animal produces between 1200 and 1800 kg of solid manure each year, containing 1105 to 1600 kg of volatile solids (van Sliedregt et al. 2000), this equates to approximately 1.3-2.6 kg volatile solids/head/day (Kissinger et al. 2007). Lodman et al. (1993) estimates the average beef cow produces 24 kg of wet manure per day, which contains on average 2.8 kg of organic matter, which can be anaerobically fermented to CH$_4$ gas (Equation 2.3).

Methane is produced in manure from the degradation of soluble lipids, carbohydrates, organic acids and proteins (Chadwick 2005). Effectively the amount of CH$_4$ produced by manure is controlled by the amount of degradable organic matter (known as volatile solids) in the substrate, but the potential CH$_4$ production in manure is influenced by a number of factors including amount, physical form of dung or urine patch, climatic and soil conditions, and the length of time a single dung patch remains intact before desiccation and decomposition (Saggar et al. 2004b). Emissions from a single dung patch on pasture are often considered negligible, but emissions from manure stored under anaerobic conditions can produce 7 to 20% of total CH$_4$ emissions in ruminants (Kreuzer and Hinderichsen 2006).
Manure CH$_4$ may not be released immediately after defecation, however CH$_4$ emission rate slows as aerobic decomposition begins (Lassey 2007; Saggar et al. 2004a). Yamulki et al. (1999) report emissions from grazing dung pats to continue for 10-20 days. Miller and Berry (2005) used incubations to measure emissions of greenhouse gases from feedlot soil and manure and observed CH$_4$ emissions only after 14 days of incubation. Methane in these incubations was found to account for only 3% CO$_2$ equivalents (CO$_2$-e) emitted over the measurement period (Miller and Berry 2005). Similarly, Jarvis et al. (1995) observed emissions in laboratory and field chambers to be stimulated by the addition of manure, but emissions declined to background concentration over roughly 10 days. Differences in the duration of emissions appear to be related to how long the substrate remains aerobic (Jarvis et al. 1995). Williams and Haynes (1990) report significant emissions of CH$_4$ from manure pats under Australian conditions for three days after deposition in winter, and two days in summer.

The feedlot pen surface provides a considerably different substrate than a single dung patch or slurry, with considerable spatial variation in manure composition (moisture, thickness, fresh/older deposits) and correspondingly, considerably difference in emissions potential. Boadi et al. (2004b) used a diagonal transect across the pen for static chamber measurements, to attempt to encompass differences in manure depth and composition across the pen. Manure pack temperature was higher in deeper sections of the pen surface than shallower sections-with a temperature range from -2°C to 8°C, and manure pack depth from 0.2 to 0.4 m.

### 2.3.2 Emissions Potential

Kreuzer and Hinderichsen (2006) reports that CH$_4$ produced by enteric fermentation is approximately 1000 times more than that produced from the manure of the cow. Furthermore, CH$_4$ produced from fermentation in a single dung pat (deposited at pasture) is very low. Under anaerobic storage manure can produce 7 to 27% of enteric CH$_4$ emissions (Kreuzer and Hinderichsen 2006). Jarvis et al. (1995) measured emissions from rectangular artificial manure pads in the field (created using manure collected primarily from dairy cows under a range of management conditions) and in laboratory scale incubations (with or without soil) emissions from dung in this study ranged from 716 to 2040 mg/ m$^2$, or an average of 73.6 mg of CH$_4$/ dung pat. Steed and Hashimoto (1994) compared different manure management systems, including feedlot, to determine ultimate CH$_4$ yield and CH$_4$ conversion factors (CH$_4$ production from organic components in the manure). Emissions from drylot management systems were considerably lower than those for slurry, although when solid manure was stored under anaerobic conditions CH$_4$ conversion factor was more similar to a slurry system.
over a range of temperatures. Solid manure tends to decompose more aerobically earlier and therefore produces less CH$_4$ (Lassey 2007).

In terms of a feedlot system, the primary focus of research focusing on emissions of CH$_4$ from manure has been in the composting phase, rather than directly from the pen surface (Chadwick 2005; Hao et al. 2004; Hao et al. 2005; Larney et al. 2006), despite the fact that composting is a relatively recent addition to manure management in feedlot systems (Larney et al. 2006). However, Boadi et al. (2004b) report on emissions of CH$_4$ from the manure pack (from small pens, containing 14 head for approximately 20 days during each measurement period) using small static chambers. In this study, emissions ranged from 11.0 g/pen/day to 17.7 g/pen/day, equating to between 0.79 g/head/day and 1.3 g/head/day. Similarly, CH$_4$ emissions from grazing dairy cows in Denmark were observed to range between 0.613 and 2.8 g CH$_4$/cow/day (Holter 1997).

### 2.3.3 Factors Influencing CH$_4$ emissions from Manure

It is possible only small amounts of the potential emissions of CH$_4$ are released under field conditions, due to the influence of environmental factors (Jarvis et al. 1995). From manure, the main factors controlling CH$_4$ production is the amount of fermentable organic matter present in the product, however the potential amount of CH$_4$ produced from manure depends on the amount of material excreted, the physical form of the deposit, excretal form (slurry, solid or effluent), climatic and soil conditions and the amount of time the deposit remains intact before being decomposed (Lassey 2007). Lodman et al. (1993) identified temperature, moisture and diet as the main factors controlling CH$_4$ emissions. Whilst Boadi et al. (2004b) suggest that under field conditions variation in temperature, moisture and time of exposure to air can contribute to large differences in potential emissions from manure of lot fed cattle.

Methane emissions from both enteric fermentation and from dung are affected by the diet. Gonzalez-Avalos and Ruiz-Suarez (2007) report that type of diet has the largest role in controlling differences in emissions when different production systems were compared. Specifically, changes in the diet affect the C:N ratio in the manure, which will influence the amount of CH$_4$ emitted (Boadi et al. 2004b). Feeding a concentrate based diet has been found to increase CH$_4$ emissions from manure (per unit of volatile solids in manure; Kreuzer and Hinderichsen 2006), possibly due to greater CHO degradation (by fermentation) in the manure. A doubling in CH$_4$ emissions from manure was observed by Hinderichsen et al. (2006) from manure of cows offered mixed forage and concentrate diet (1:1 ratio) compared with cows offered a forage only ration. However the reverse was observed by (Jarvis et al. 1995) who observed higher emissions from forage only dung pats. Boadi et al. (2004b)
observed insignificant differences between isocaloric diets varying in forage content on manure greenhouse gas emissions. Lodman et al. (1993) reports increased emissions from grain based rations (0.30 to 1.22 l CH₄/kg OM/day) compared with forage based rations (0.08 to 0.1 l CH₄/kg OM/day), attributed to more readily fermentable CHO in the manure of grain fed animals.

More broadly, ration affects the amount of fermentable substrates available in manure (Figure 2.2). Jarvis et al. (1995) reported that cattle consuming high forage diets produce manure with a higher content of partially degraded cell wall material, which is more resilient to microbial degradation than manure of cattle fed high grain diets. However, Hashimoto et al. (1981) observed a decline in volatile solids with increasing corn silage content in the ration; compared beef cattle manure from cattle fed diets ranging from 7 to 91.5% corn silage using anaerobic fermenters determining 0.173, 0.232 and 0.290 L CH₄/g of volatile solids.

Methane production from dung has been observed to have strong temperature dependence (Figure 2.2) (Jarvis et al. 1995). Gonzalez-Avalos and Ruiz-Suarez (2001) suggests that temperature plays the greatest role in controlling emissions within the same production system. It appears that temperature affects the rate at which CH₄ is produced (possibly due to effect on microbial activity) but does not increase the amount of CH₄ that can be produced from a unit of substrate (Hashimoto et al. 1981). Jarvis et al. (1995) observed that significant emissions can occur at as low as 6°C, consistent with the observations of Boadi et al. (2004b) observed greater emissions from chambers located in the deepest parts of a pen, which were also the warmest temperature (range from -2 to 8 °C). Steed and Hashimoto (1994) used an artificial dry lot manure management system (open chamber) demonstrated CH₄ conversion factors ranging between 0 and 2.0% from 10°C to 30°C. In contrast CH₄ conversion factors from a liquid slurry system (closed chamber) ranged from 0.2 to 75.6% from 10°C to 30°C. Furthermore, Hashimoto et al. (1981) observed faster fermentation at 45°C than at 35°C.
Figure 2.2 Stylised representation of three significant factors influencing emissions of methane from feedlot manure: ration forage and concentrate contents (A), moisture (B) and temperature (C).
Moisture content (Figure 2.2) has also been found to significantly influence CH\textsubscript{4} emissions by Miller and Berry (2005) and Gonzalez-Avalos (2001). For example, Miller and Berry (2005) determined greenhouse gas fluxes to differ with soil manure and moisture content, with the largest fluxes at moderate to high moisture content, depending on manure content. Jarvis \textit{et al.} (1995) suggests that rainfall increases the anaerobic status of individual dung pats, but that high rainfall may decrease emissions by washing substrates available for methanogenic activity out of the dung patch into the soil. Miller \textit{et al.} (2003) determined a negative relationship between the water content of manure and bacterial content. However, an anaerobic surface would be expected to result in increased CH\textsubscript{4} formation (Miller \textit{et al.} 2003).

Miller and Berry (2005) observed greatest fluxes of greenhouse gas at moderate moisture contents (0.25-0.43 g/g FM) when the substrate contained moderate manure levels. Temperature and moisture status interacts significantly in controlling emissions (Jarvis \textit{et al.} 1995). Higher temperature stimulates microbial activity and CH\textsubscript{4} production, however higher temperatures also results in crust formation (Jarvis \textit{et al.} 1995). The formation of a crust helps maintain the anaerobic status of the pat; however it also influences the exchange of CH\textsubscript{4} between the pat and the atmosphere (de Klein \textit{et al.} 2003; Saggar \textit{et al.} 2004b). Lodman \textit{et al.} (1993) suggest that temperature (while it is between 20 and 30°C) has little effect on CH\textsubscript{4} production where manure dries rapidly. Further Steed and Hashimoto (1994) determined CH\textsubscript{4} conversion factors to be different depending on temperature; they suggest that lower CH\textsubscript{4} conversion factors observed at high temperatures was related to drying and crusting of the manure.

Along with temperature, oxygen plays a role in controlling CH\textsubscript{4} emissions from stored solid manure, despite methane production being the result of anaerobic fermentation. Microbial activity consuming oxygen in stored manure can result in increased temperature (Tiquia \textit{et al.} 1996), which, as discussed above increases CH\textsubscript{4} production. Hao \textit{et al.} (2001b) observed emission rates of CH\textsubscript{4} to be higher when O\textsubscript{2} consumption rates were high in composting feedlot cattle manure. This effect may produce hot spots of CH\textsubscript{4} emissions in the manure pad, as microbial activity decreases O\textsubscript{2}, increasing temperature and creating an anaerobic environment. Overall, Hao \textit{et al.} (2001b) observed higher greenhouse gas emissions from a composting stack which was actively aerated- which they attribute to effect on biological activity. The effect of this process has some implications for emissions directly from the manure pad, where animal hoof activity would regularly mix and aerate the substrate.
2.4 Nitrogenous Gases

Built up manure in the feedlot provides a significant source of nitrogenous gases; both volatile NH₃, and N₂O produced from transactions in the soil surface. NH₃ is lost from the feedlot manure pad via volatilisation, which depends on complex interactions between soil (manure pad), climatic and management conditions. The primary factors controlling NH₃ emissions are NH₄ and NO₃ concentrations, aeration, soil water, availability of degradable organic matter (OM), pH and temperature (Saggar et al. 2004b). Mosier et al. (1998) report that there are three potential sources of N₂O from animal production, the animals themselves, wastes from confined animals and the urine and dung deposited by grazing animals. However, volatilised NH₃, deposited outside the feedlot can itself be a source of N₂O (Denmead et al. 2008) although further quantification of the magnitude of these emissions are required.

2.5 Nitrous Oxide Emissions Process

Nitrous oxide is produced (mainly in soils) by denitrifying and nitrifying microorganisms (Saggar et al. 2004b). Denitrifiers reduced N-oxides to N-gas under anaerobic conditions, and when there is sufficient NO₃ and available C. Nitrifying microbes convert soil NH₄ to NO₃ under aerobic conditions; Incomplete conversion results in the formation of N₂O (de Klein et al. 2003). Nitrous oxide is the gaseous intermediate of the reaction sequences of both processes which leaks from microbial cells into the soil atmosphere (Mosier et al. 1998).

The production of N₂O from manure can be considered a 2 step process (Equations 2.3 and 2.4), dissolved NH₄ is oxidised by nitrifying bacteria in aerobic zones, while the products of nitrification are denitrified in anoxic areas. Nitrification occurs in the presence of oxygen (O₂), denitrification, which follows nitrification occurs when oxygen levels are depleted and NO₃ becomes the primary source of O₂ for the microbial populations. When the bacteria break apart the NO₃ to produce oxygen N₂O is produced. Denitrification occurs only under anaerobic or anoxic conditions (Saggar et al. 2004b; Uchida et al. 2008). Both nitrification and denitrification can occur in a single dung pat, compost stack, or within a feedlot pen, associated with differences in aeration of the manure (Chadwick 2005).

Equation 2.3 Nitrification (Saggar et al. 2004b)

\[
\begin{align*}
\text{Ammonium Mono-oxygenase} & : & \text{Hyroxylamine Oxidoreductase} & : & \text{N₂O} \\
\text{NH₄⁺} & \rightarrow & \text{NH₂OH} & \rightarrow & [\text{HNO}] & \rightarrow & \text{NO₂⁻} & \rightarrow & \text{NO₃⁻} \\
\text{O₂} & & \frac{1}{2} \text{O₂} & & \text{H₂O} & & \text{H₂O}
\end{align*}
\]
Equation 2.4 Denitrification (Saggar et al. 2004b)

\[
\begin{align*}
\text{Nitrate Reductase} & \quad \text{Nitrite Reductase} & \quad \text{NO- Reductase} & \quad \text{N2O Reductase} \\
\text{NO}_3^- & \quad \rightarrow & \quad \text{NO}_2^- & \quad \rightarrow & \quad [\text{NO}] & \quad \rightarrow & \quad \text{N}_2\text{O} & \quad \rightarrow & \quad \text{N}_2 \\
\uparrow & \quad \uparrow & \quad \uparrow & \quad \uparrow & \\
\text{ATP} & \quad \text{ATP} & \quad \text{ATP} & \quad \text{ATP}
\end{align*}
\]

2.5.1 N\textsubscript{2}O Emissions Potential

Solid manure stores can be a significant source of N\textsubscript{2}O emissions (Chadwick 2005; Chadwick et al. 2001), however actual emissions from a source may be significantly lower than potential due to the influence of substrate and climatic conditions. Chadwick (2005) measured emissions of N\textsubscript{2}O between 0.1 and 2.3% of total inorganic N in the non compacted manure storage heap, whilst others (Amon et al. 2001, Sommer et al. 2004) report emission of less than 1% of the initial N content. Van Groenigen et al. (2005) report on the results of a range of studies with artificial, bovine and ovine urine, reporting potential N\textsubscript{2}O emissions from 0.07 to 15.5% (of applied urine N) from bovine urine. Very few studies report N\textsubscript{2}O emissions from lot fed systems, which research primarily focusing on N\textsubscript{2}O emitted from urine or dung patches, slurries or grazing dairy systems (Chadwick 2005; Kulling et al. 2001; Kulling et al. 2003). Boadi et al. (2004b); measured N\textsubscript{2}O emissions from pens containing 14 head, determining emissions of 1.9 to 2.7 g/pen/day, equating to 0.14 g N\textsubscript{2}O/head/day to 0.19 g N\textsubscript{2}O/head/day.

2.5.2 Factors Influencing Emissions of N\textsubscript{2}O

The production and transport of N within the soil and manure media is primarily related to factors which influence the soils physical condition, specifically aeration and soil water content (Figure 2.3) (Saggar et al. 2004b). Uchida et al. (2008) identified key factors influencing microbial N processes as soil moisture status, soil respiration rates, soil aggregation and degree of soil compaction- which all contribute to the formation of anaerobic conditions- promoting N\textsubscript{2}O emission. Van Groenigen et al. (2005) suggest the combined effects of compaction, urine and dung in the field control N\textsubscript{2}O emissions from pasture. It is likely that these factors will effectively control emissions from feedlot manure pads; however the emissions process is likely to be more complex due to the continual deposition of dung and urine, resulting in considerable variation in manure age, compaction, moisture, depth and temperature even within a single pen (Cole et al. 2009).
Primarily, the availability of substrate N (NH$_4$ and NO$_3$) affects N$_2$O gas formation. Nitrogen in urine is a major source of N for transformations in soil, with deposition of urine N creating very high concentrations in localised areas. Combined with readily available C, urine patches create ideal conditions for N$_2$O emissions (Denmead et al. 2008). Decreasing N concentration in urine, or total amount of urine N applied can significantly decrease potential N$_2$O emission (Van Groenigen et al. 2005). Van Groenigen et al. (2005) observed the highest emissions in their incubation which contained dung and urine (two sources of N), compared with urine alone, and urine plus dry soil. However, other authors observed a similar effect only in winter, with higher percentage losses from dung in spring and summer (Saggar et al. 2004b). Oenema et al. (1998) observed emissions of N$_2$O from dung which were equal to those from urine, however almost three times as much N was applied in the dung.

The microbial agents of N$_2$O production, nitrifiers and denitrifiers differ in their requirements for oxygen. Denitrifiers require anaerobic conditions, while nitrifiers operate in aerobic environments (Uchida et al. 2008). Mosier et al. (1998) report that aeration initiates the nitrification/ denitrification reactions, making the release of N$_2$O possible. Whilst, Uchida et al. (2008) determined that increasing compaction increased the proportion of water filled pore space, increasing anaerobic conditions and resulting in relatively high fluxes from denitrification, supported by Van Groenigen et al. (2005) and Ball et al. (1999). Animal treading triggers denitrification, increasing anaerobic conditions (Saggar et al. 2004b) and is reported by Van Groenigen et al. (2005) to create local hotspots of N$_2$O emissions. Hot spots of N$_2$O emissions may also be the result of a combination of aerobic and anaerobic spots within the substrate. Availability of O$_2$ results in nitrification, which coupled with anaerobic denitrification, will result in increased emissions. Hao et al. (2001b) observed increased rates of N$_2$O emission from areas of a composting manure stack which also demonstrated high O$_2$ consumption rates.

A significant reduction in aeration (associated with compaction, or high rates of O$_2$ consumption) (Figure 2.3) will decrease nitrification, eventually slowing denitrification as substrate availability declines. Van Groenigen et al. (2005) determined the highest N$_2$O emissions from incubations containing both urine and dung, and those which had applied urine and compaction; however when soil was compacted peak in N$_2$O emissions was delayed by approximately 15 days. This may be consistent with the observations of Chadwick (2005), who suggest that compaction of manure (in storage and in the field) can reduce N$_2$O emissions, through the creation of anaerobic conditions, which inhibit nitrification. Inhibition of nitrification can prevent the formation of NO$_3$, limiting loss of N$_2$O via denitrification (Chadwick 2005).
Figure 2.3 Stylised representations of three key factors controlling N$_2$O. (A) Dietary nitrogen, (B) water filled pore space (through effect on aeration) and (C) Soil compaction (through effect on aeration).
Chadwick (2005) reports significantly lower emissions from compacted storage heaps (0.59 compared with 0.17 kg/heap). Similarly, Denmead et al. (2008) supports the observations of Hutchinson et al. (1982) that nitrification in the feedlot occurs under dry aerobic conditions, whereas under a wet feedlot pen, with an anaerobic surface reduced nitrification occurs, combined with increased NH$_3$ volatilisation.

It could be assumed that the high levels of compaction observed in a feedlot pen, combined with the continual addition of N, and constant treading would create ideal conditions for the transformations producing N$_2$O; however, this does not appear to be supported by the observations of (Boadi et al. 2004b) or the lack of research attention this subject has received.

2.6 Ammonia Emissions Process

Once N is excreted it undergoes a number of transformations which result in release of gaseous N compounds, including emission of N$_2$O and volatilisation of NH$_3$. The hydrolysis of macro-molecules of N (e.g. proteins) is undertaken by heterotrophic microorganisms, resulting in the release of amino acids and amines, which can be metabolized enzymatically to produce NH$_3$ (Saggar et al. 2004b). Urea N is broken into bicarbonate and NH$_4$ by the action of urease, a process known as ammonification or urea hydrolysis (Equation 2.6; Saggar et al. 2004b; Satter et al. 2002).

**Equation 2.6** The urea hydrolysis reaction produces ammonium from urea (Saggar et al. 2004b)

$$\text{CO(NH}_2\text{)}_2 \rightarrow 2\text{NH}_4^+ + 2\text{OH}^- + \text{CO}_2$$

Urea hydrolysis (Equation 2.6) releases hydroxyl ions, increasing the pH of urine spots and other substrates (Saggar et al. 2004b). Under these alkaline conditions NH$_4$ dissociates to gaseous NH$_3$ (Saggar et al. 2004b; Satter et al. 2002). NH$_3$ is lost from the soil/manure media by volatilisation (Equation 2.7); a chemical process which occurs only under alkaline conditions (pH >7.5).

**Equation 2.7** Ammonia volatilisation (Saggar et al. 2004b)

$$\text{NH}_4^+ + \text{OH}^- \rightarrow \text{NH}_3 + \text{H}_2\text{O}$$

Under optimal conditions (pH, moisture and temperature) all the urea in excreta can be hydrolysed within hours (Satter et al. 2002). In the feedlot, processes in the manure and soil act to produce a large pool of NH$_4$ from urine and faeces. The high concentration of NH$_3$ in
the soil/manure generally results in volatilisation simply due to the concentration difference between the manure “solution” and the air (Satter et al. 2002).

2.6.1 Potential NH$_3$ Emissions from Feedlots

Livestock operations are a major source of atmospheric NH$_3$ (McGinn et al. 2003). Compared with the emissions of the greenhouse gases N$_2$O and CH$_4$, emissions of NH$_3$ from feedlot environments has received considerably more research, primarily due to the association of NH$_3$ with malodorous volatiles, and contribution of potential health hazards (McGinn et al. 2003). McGinn et al. (2003) observed emissions of NH$_3$ approaching or exceeding odour thresholds up to 200m from individual feedlots. Ammonia emissions from feedlots are significantly influenced by climatic conditions, including temperature, rainfall and substrate composition. McGinn et al. (2007) report NH$_3$ fluxes of 140 g/head/day similar to the observations of Flesch et al. (2007) of 150 g/head/day. Todd et al. (2008) measured emissions between 51 and 131 g/head/day, with emissions tending to be higher in summer than in winter. Similarly Todd et al. (2005) observed fluxes during winter of approximately 50 g/head/day (2140 kg/day for the measured feedlot) to be half those in summer (approximately 98 g/head/day, 4650 kg/day). Baek et al. (2006) observed the same trends, with substantially lower emissions in winter (approximately 6.5 g/head/day) compared to summer (75.9 g/head/day), which they attribute to the extremely low manure temperature during winter.

Australian studies investigating NH$_3$ volatilisation are more limited, probably due to the lack of regulations (in contrast to North American and Canada) surrounding air quality in and around feedlot sites. Denmead et al. (2008) reports emissions of 69 g/head/day and 24 g/head/day, although these are lower than those reported by both Loh et al. (2008) and Chen et al. (2009) for the same sites (94 g/head/day to 324 g/head/day).

2.6.2 Factors Influencing NH$_3$ Emissions

The majority of volatile N emitted from feedlot environments is volatilised from urine spots, rather than from dung (Rhoades et al. 2008). Up to 80% of the urea in urine can be volatilised within two hours of deposition, depending on conditions (Rhoades et al. 2008). The major factors which control the emission of NH$_3$ from animal feeding operations are number, age and type of animal, housing design and management, type of manure storage, and application technique, excretion rates per animal and environmental conditions (Arogo et al. 2006). High urinary N per cow and per day gives the potential of a high NH$_3$ release, however it is physical and biochemical factors within the urine and in the surrounds will determine how much is volatilised (Figure 2.4; Cole et al. 2005; Swensson 2003). Cole et al. (2003) reports
that factors other than urinary N concentration accounted for nearly 40% of total variation in NH$_3$ emissions. However, concentration of urea (urinary N) is the major factor controlling volatilisation of NH$_3$ (Smits et al. 1995). Thirty to 70% of N fed is commonly excreted in typical feedlot diets; however this value increases when fed N exceeds requirements. Cole et al. (2005) suggests that increasing urinary N will increase NH$_3$ emissions (and conversely decreasing crude protein and urinary N excretion will decrease potential volatilisation). Under *in vitro* conditions Cole et al. (2005) determined an increase in NH$_3$-N loss from 3.15 to 4.32% of applied N with an increase in dietary CP from 11.5 to 14.5%, although this is a extremely small proportion of applied N, compared with those measured *in situ*. Erickson et al (2000.) observed similar results, with decreased N excretion accompanied by decreased NH$_3$ volatilisation. NH$_3$ volatilisation averaged 108 to 158 g/head/day in yearling steers, and 62 to 73 g/head/day for calves.

This effect is observed under field conditions, although moderated by environmental impacts. Todd et al. (2008) report emissions equivalent to 53% of the N fed to cattle, with an increase in emissions between 10-64% when dietary N increased by 15 to 26%, annual emission rate was 53% of N fed to cattle, or 4430 kg NH$_3$/day (over the source feedlot). However summer emissions rates were considerably higher than winter, 7420 kg NH$_3$/day and 3330 kg NH$_3$/day. Similarly, Rhoades et al. (2008) reported emissions of between 1387 and 2955 kg NH$_3$/day, equating to 34 to 70% of fed N, whilst Erickson et al. (2000) report emissions during summer of 60% of N intake compared with 40 to 50% during winter.

The variability in emission rates reported from feedlots (Loh et al. 2008; McGinn et al. 2007; Rhoades et al. 2008; Todd et al. 2005) represent the impact of environmental conditions on emissions to a greater extent than they do differences in crude protein offered. The key factors that determine the rate of volatilisation of NH$_3$ from soils/manure media are those which affect the rate of conversion between NH$_4$ and NH$_3$ and the transfer of NH$_3$ gas between the manure media and the atmosphere (Saggar et al. 2004b). The influence of the animal factors discussed above is primarily related to the concentrations of N and urea in the manure.

Environmental factors such as media pH, moisture, temperature and wind velocity influence the both conversion and transfer (Petersen et al. 1998, Rhoades et al. 2008, Saggar et al. 2004b). High microbial activity, warm temperature, large emission surface, high pH and air velocity may result in increased NH$_3$ volatilisation (Arogo et al. 2006). Acidity is considered by many to be the most important property controlling NH$_3$ volatilisation, through its influence on the equilibrium between NH$_4$ and NH$_3$. Dissociation of NH$_3$ from NH$_4$ requires
alkaline conditions (Panetta et al. 2005). More acidic surfaces/acid forming compounds will result in negligible release of NH$_3$ (Saggar et al. 2004b).

Acidity influences the transformation of urea to NH$_3$, at higher pH the reaction favours increased release of NH$_3$; when pH > 8 the urea molecule hydrolyses into two molecules of NH$_3$. At lower pH (<6.5) the majority of N is found as NH$_4$ (Equation 2.8 ;Rhoades et al. 2008). Whitehead and Raistrick (1993) observed that pH measured after 24 hours was more strongly correlated with NH$_3$ volatilised than initial pH, which was poorly correlated with volatile NH$_3$.

Equation 2.8 Hydrolysis of urea in the presence of water, to NH$_3$, at an alkaline pH. This is a bidirectional process, with formation of urea as pH decreases (Rhoades et al. 2008).

\[
\text{CO(NH}_3\text{)}_2 + \text{H}_2\text{O} \rightarrow 2\text{NH}_3 + \text{CO}_2 \quad [\text{pH } > 8]
\]
\[
2\text{NH}_3 + \text{CO}_2 \rightarrow \text{CO(NH}_3\text{)}_2 + \text{H}_2\text{O} \quad [\text{pH } < 6.5]
\]

Ammonia volatilisation increases with increasing temperature and wind speed. Release of NH$_3$ from the soil surface is influenced primarily by the partial pressure of NH$_3$ in the atmosphere. Temperature and wind speed have significant effects on release of NH$_3$ through their effects on the concentration of NH$_3$ at the media surface (Saggar et al. 2004b). Saggar et al. (2004b) reports that increases in temperature decreases the solubility of NH$_3$ in water, increases the proportion of NH$_3$ (compared with NH$_4$) in the media and increases diffusion away from the surface. Very low temperatures (e.g. below minus four) can result in the cessation of microbial activity, and therefore urea hydrolysis (Saggar et al. 2004b).
Figure 2.4 Stylised representation of three factors influencing ammonia volatilisation from feedlot manure; (A) Dietary Nitrogen, (B) Substrate pH and (C) Environmental Temperature.
Stewart (1970) examined applications of urine to wet and dry soil columns, observing that when urine was added to wet soil, every two days, less than 25% of total added N was lost as NH$_3$. However, when urine was added every four days to dry soil (urine was not added until more than 90% of water had evaporated from the previous application) more than 90% of the added N was lost as NH$_3$. Losses from the dry soil was relatively low following the first few applications, indicating some build up of NH$_4$ in the soil. As well as differences in moisture content, pH was significantly higher in the dry soil (<8.0) compared with the wet soil (<7), perhaps reflecting the greater accumulation of NH$_4$ in the wet soil (particularly in the upper layers of soil).

### 2.7 Quantification of Enteric Emissions

Initially, CH$_4$ was measured as an adjunct to the estimation of metabolisable energy (ME) from gross energy (GE) consumed by the animal e.g. Blaxter and Clapperton (1965) and Moe and Tyrrell (1979), however due to increased interest in CH$_4$ as a greenhouse gas focus has shifted from quantification of CH$_4$ as a form of feed energy loss, to quantification of CH$_4$ as an environmental pollutant. Correspondingly measurement methods have changed from an enclosed individual animal focus, to a whole farm approaches.

#### 2.7.1 Calorimetric/ Chamber Methods

The earliest established method of determining CH$_4$ output is enclosure of the animal in a respiration chamber. These chambers may be open or closed circuit, for open circuit chambers the change in CH$_4$ content of entry and exit air is determined. In closed chambers the build up of CH$_4$ is measured (Beauchemin and McGinn 2005). In general, to measure emissions from enteric and respiratory processes cattle are removed from the feedlot (or grazing system) and held in metabolism stalls, and enclosed in calorimeters (chambers; Grainger et al. 2007; Sommer et al. 2004). This method allows accurate measurement of CH$_4$ and CO$_2$, from respiratory and enteric pathways. However, measurements using this method are expensive and time consuming, and therefore impractical for regular measurements, or use under commercial conditions.

A further limitation of these chamber measurements is that the environment in which the measurements are conducted is artificial and highly controlled. Animal movement is restricted and behaviour and feed intakes are often decreased (Grainger et al. 2007; Murray et al. 1999). The reduced level of activity undertaken by animals enclosed in a chamber (compared with grazing) has also been cited as potential source of error. However, Blaxter and Clapperton
(1965) reported no significant difference between the CH$_4$ output of exercising and non-exercising sheep (McCrabb and Hunter 1999).

Chamber techniques do, however, provide information regarding the variability of emissions during the day (Grainger et al. 2007; Stackhouse et al. 2011). A distinct diurnal emissions pattern is reported for CH$_4$ emissions by a number of sources, thought to be associated with animal feeding behaviour. Grainger et al. (2007) report that for an individual cow, emissions peaked after feeding and lowest just before feeding, which is similar to observations with grazing and feedlot fed cattle (Loh et al. 2008; McGinn et al. 2006b). However, the artificial environment created within the chamber, and reliance on feed delivery (in this study feed was delivered at milking time) is likely to result in this diurnal variation not representing that observed under more “natural” feeding conditions, particularly grazing environments.

These factors limit the extrapolation of these results to ‘real-world’ situations (Johnson et al. 1994; Johnson et al. 2001; Murray et al. 1999). Despite this, the equations which are commonly used to predict CH$_4$ output are based on calorimetric data (Johnson et al. 1994). However, despite issues surrounding intake and alterations in behaviour chamber measurements continue to be the “gold standard” for comparing differences between feedings, and in assessing potential mitigation options, provided consideration is given to the limitations of the measurement method.

2.7.2 Tracer Techniques

More recently the use of a tracer (Sulphur-hexafluoride, SF$_6$) has become popular in the measurement of CH$_4$ emissions (Grainger et al. 2007; Kaharabata et al. 2000; McGinn et al. 2006a). This technique, developed in the United States by Johnson et al. (1994) uses SF$_6$ to account for the dilutions of gases being eructated via the mouth. SF$_6$ was selected as a suitable gas as it has a low background concentration, low detection limits and a similar solubility to CH$_4$ (Machmuller and Hegarty 2006). This technique involves inserting a source of SF$_6$, with a known release rate, into the rumen of each animal; eructated gases are collected at the mouth of the animal and analyses for SF$_6$ and CH$_4$ to estimate CH$_4$ production (Johnson et al. 1994). This assumes that the SF$_6$ emission rates exactly simulates CH$_4$ production and therefore that the dilution rates will be equal (Johnson et al. 2001). The CH$_4$ emission rate can then be determined using the measured CH$_4$ and SF$_6$ concentrations and the release rate of SF$_6$ (Equation 2.9). The concentration of CH$_4$ and SF$_6$ are determined using gas chromatography (Lassey 2007). This technique enables the measurement of emissions from grazing animals, and is the most suitable techniques for the measurement of CH$_4$ emissions from individual grazing animals (Johnson et al. 2001). This method is particularly suitable for
grazing animals, however for animals which are housed or managed at high densities contamination of sampled air can occur (Moate et al. 2011).

**Equation 2.9** Determination of CH₄ emission rate using SF₆

\[
Q_{\text{CH}_4} = Q_{\text{SF}_6} \times [\text{CH}_4]/[\text{SF}_6]
\]

The major problem surrounding the use of SF₆ as a tracer is that SF₆ in itself is a greenhouse gas (Machmuller and Hegarty 2006), with a global warming potential 23,900 times that of CO₂ and an atmospheric lifetime of 3200 years. Furthermore SF₆ limited by food and drug regulations; withdrawal (or withholding) period is long, which limits use in commercial situations (Vlaming et al. 2005). Measurements conducted using SF₆ have been observed to be more variable between animals than calorimetry (Grainger et al. 2007; Immig 1996). Grainger et al. (2007) report a coefficient of variation of 19.6% for between animal measurements using the SF₆ technique, compared with 17.8% for chamber measurements. Grainger et al. (2007) also report higher variation between days in emissions measured using the SF₆ technique, and suggests longer measurement periods (days) are required to achieve the same level of precision as chamber measurements. There have also be observed to be differences in accuracy (SF₆ compared with tracer) when different diets are fed. There appears to be a higher correlation between chamber and SF₆ technique diet for high forage diet (higher rates of rumen digestion; Johnson et al. 2001). Although no significant difference was determined using a wide range of diets by Grainger et al. (2007).

A small proportion of CH₄ is produced in the hind-gut of ruminants. Hindgut fermentation may become more important in ruminants when degradability of the substrate in the rumen decreases. This may occur when feeding ground or pelleted diets, or diets containing a large amount of starch or fat (Immig 1996). Methane produced in the hindgut is predominantly excreted via the bloodstream and the lungs, but a small part is emitted via the anus (Grainger et al. 2007; McGinn et al. 2006a). Emissions from the hindgut are not accounted for by the SF₆ technique, which is thought to contribute to reduced emissions measured using SF₆ than under chamber techniques. Recent comparisons in Canada and Australia of SF₆ with chamber techniques indicated an underestimation of CH₄ emissions of between 4 and 8% using SF₆ (Grainger et al. 2007; McGinn et al. 2004). While others suggest that CH₄ emissions measured using the SF₆ technique are 93 to 95% of those measured using whole animal chambers (McGinn et al. 2006a; Ulyatt et al. 1999). These authors suggest that the difference between SF₆ and chamber CH₄ emissions may indicate higher CH₄ production/ release from the hindgut (Grainger et al. 2007). Feeding a diet which increases hind gut fermentation may create increased uncertainty with SF₆ measurements (McGinn et al. 2004).


### 2.8 Quantification of Manure Emissions

As with enteric CH$_4$ emissions, approaches for measuring emissions from manure can be categorised into chamber and whole system measurements. The requirements for chamber measurements differ between CH$_4$, N$_2$O and NH$_3$, although overall approaches are similar. Kreuzer and Hinderichen (2006) suggest that methods of the quantification of manure CH$_4$ emissions are more demanding than those for measurement of enteric emissions, due primarily to the small magnitude of emissions and the potential duration of emissions.

#### 2.8.1 Chamber Approaches

Commonly, approaches for measuring CH$_4$, NH$_3$ and N$_2$O from manure involve the use of small chambers, which isolate a dung patch, sample of manure or small area of a compost stack. Chambers methods can be used to measure emissions directly from the manure pad; and from soils and provide sensitivity and portability (Denmead 1994). Chamber measurements can be either laboratory or field based. A number of recent studies have investigated NH$_3$ and CH$_4$ emissions from feedlot manure (Escue et al. 2004; McGinn et al. 2002), but measurements are often conducted in laboratory chambers, with artificial feed yard surfaces (Todd et al. 2006) rather than in the field meaning that the surface/ samples are likely to be more uniform than actual conditions. Field chamber measurements have been used for soil (de Klein et al. 2003) composting manure (Hao et al. 2001b) and cattle dung pats (Chadwick 2005; Holter 1997; Jarvis et al. 1995).

There are two primary types of chambers used in the measurement of fluxes from soil/ manure; dynamic and static. In static (closed) chamber systems there is no replacement of the air in the head space of the chamber, and gas concentration changes constantly. Boadi et al. (2004b) adapted a closed chamber approach and used a vented static chamber to measure emissions of CH$_4$ and N$_2$O from feedlot manure.

In open/ dynamic chamber systems a constant flow of air through the head space is maintained and gas concentration maintains a constant difference from the background concentration of air. In general, closed chamber systems are used more commonly, as they are mechanically simpler (Denmead 1994). However, the nature of NH$_3$ including dependence on a concentration gradient for volatilisation requires the use of an open chamber with an adequate ventilation rate (greater than 0.3 displacement/ minute) is required to avoid suppression of emissions (McGinn et al. 2002).

In a closed system rate of concentration change is used to calculate flux rate. Commonly, a linear regression approach is used to estimate emission rate (Kreuzer and Hinderichsen 2006;
Kulling *et al.* 2001) from a number of samples during the measurement period. However, there are a variety of methods used to calculate emissions from manure and soil. Sommer *et al.* (2004) applied a second order polynomial equation to the concentration of gas vs. time; however other authors assume a linear increase in gas concentration (de Klein *et al.* 2003). Kreuzer and Hinderichsen (2006) reports some issues with the repeatability of individual measurements and regressions using a linear approach (Hinderichsen *et al.* 2006). The number of gas samples taken during the incubation period varies, typically from one to three; de Klein *et al.* (2003) suggest that taking a minimum of three samples during the incubation period will help identify non-linear increases.

The use of chambers to measure emissions of both CH$_4$ and N$_2$O from agricultural systems gives increased sensitivity compared with open-path measurements. Denmead (1994) reports that chambers can measure fluxes 100 times smaller than micrometeorological methods; further benefits of chambers are portability and relative simplicity. The small size and area occupied allows small scale studies to be undertaken which would not be possible with micrometeorological methods. However, the small area covered also presents an issue given the possibility for significant spatial variability in emissions (Denmead 1994), particularly under grazing systems where small areas may be “hot spots” for N$_2$O/ NH$_3$ due to deposition of urine.

Additionally, the use of a static chamber can result in microclimate changes, affecting the rate of CH$_4$, NH$_3$ and N$_2$O emission (Denmead 1994). The increased concentration of gases in a closed chamber can also affect emissions, where they occur along a concentration gradient-this is particularly a concern for NH$_3$. Bekku *et al.* (1997) reported that over a short time period (two to four minutes in a closed chamber) the increase in CO$_2$ concentration would have little effect on the rate of CO$_2$ emission. Further, Ball *et al.* (1999) suggest sampling one hour after closure for N$_2$O and CO$_2$ and at intervals over one hour after closure for CH$_4$. A static (or closed) chamber technique also relies on the principle that gas diffuses from the manure/ soil, making it less suitable for use in composting manure piles where an outflow of air is created by heat produced by the composting process (Sommer *et al.* 2004). As discussed, particularly in terms of NH$_3$, a major contributing factor to the rate of volatilisation in the partial pressure difference between the manure media and the atmosphere (Sommer *et al.* 2004). The presence of a chamber can cause microclimate effects which influence temperature and wind speed, as well as causing an build up in gas concentration (in a closed system), all of which can affect measured concentration (Denmead 1994).
2.9 Whole System Measurements

The primary disadvantage of the individual approaches is the exclusion of the animal from its natural environment or requirement for frequent interaction with the animals (e.g. changing of gas collection canisters for SF₆) which alters behaviour, controls conditions and reduces usefulness under commercial environments. Measured emissions may therefore not be representative of the animal or source when it is managed under “natural” conditions (McGinn et al. 2009). Laubach and Kelliher (2004) suggest that measurement of emissions (and therefore verification of emissions estimates) will be improved with measurements which can incorporate the entire herd- rather than the individual. An approach which can measure emissions from both enteric and hind gut fermentation may also be more appropriate for feedlot rations, particularly those based on high starch grains (such as corn; McGinn et al. 2004), in contrast to the SF₆ technique which measures only enteric CH₄. The microclimate impacts of chambers, and significant spatial variability in emissions observed from manure heaps and soils indicate that “whole system” measurements may provide more accurate estimates of potential emissions.

Micrometeorological measurements can be used to measure emissions from the whole system, and have the added advantage of being non-intrusive, less labour intensive and enable estimates of emissions every few minutes (McGinn et al. 2009). The ability to measure a group of cattle removes the effect of between and within animal differences when considering treatment differences (McGinn et al. 2009). These approaches also mean that average intake can be determined on a pen/ group basis, which is considerably simpler than individual estimates of intakes. Micrometeorological approaches enable emissions measurement on a larger, whole farm, scale, enabling inferences to be made about management practices which may impact on emissions (McGinn et al. 2006b).

These measurements commonly involve measurement of gas concentration downwind of the source, and calculations based on micrometeorological parameters (wind speed, wind direction and turbulence). Open-path methods (e.g. tuneable lasers, FTIR, micrometeorology) are becoming increasingly common for the measurement of greenhouse gas emissions from feedlot systems (Denmead et al. 2000; Flesch et al. 2007; McGinn et al. 2009; McGinn et al. 2007). However, there are a number of alternatives for both measuring and calculating emissions based on micrometeorological measurements, a full discussion of which is beyond the scope of this review. Measurement approaches include mass balance (micrometeorological mass difference) appropriate for small areas (Denmead et al. 2000; Harper et al. 1999), flux gradient, integrated horizontal flux (Flesch et al. 2004; Laubach and
Kelliher 2005) and variations on the open-path measurements using different forms of dispersion modelling (McBain and Desjardins 2005). An open-path method can be separated from the other approaches utilising micrometeorological measurements primarily by the measurement approach used to determine concentration. In contrast to a point source measurement with sampling at varying heights an open-path method measures concentration of CH$_4$ over a fixed path length (a line average; Loh et al. 2008) allowing measurements of concentration under a wider range of wind conditions.

The most commonly used modelling method for agricultural systems is inverse dispersion modelling, although open-path measurements are also applicable to integrated horizontal flux estimates (Laubach and Kelliher 2005). The inverse dispersion technique effectively models the dispersion of gas from a target gas from a source, to make estimates of emission rate (McGinn et al. 2006b). An atmospheric dispersion model enables emission rate from a source to be determined indirectly, reducing the requirements for complex measurement equipment (Flesch et al. 2004). Inverse dispersion modelling has been used for varying scales and source complexities, including feedlots (Flesch et al. 2007). In contrast to the mass balance methods utilising an inverse dispersion approach enables calculation of emission rates based on a line or point concentration measurement based on knowledge of wind statistics (McBain and Desjardins 2005). The most common inverse dispersion technique is the backward Lagrangian stochastic model (bLS) as described by Flesch et al. (2007).

A limitation of these approaches is that emissions cannot be partitioned into sources (e.g. enteric vs. manure CH$_4$) and emissions, particularly if they are small, may not be able to be distinguished from background emissions. Measurement of emissions from feedlot systems are complicated by multiple sources contained within the same area. Total feedlot emissions of CH$_4$ consist of enteric fermentation, and emissions from the manure pad. Additionally, CO$_2$ (although not considered an emission for accounting purposes), is derived from the manure pad, and from enteric and respiration pathways in the animal. The primary source of N$_2$O is the manure pad; however background emissions may be present from soil, effluent ponds and composting areas. A further issue of micrometeorological measurements is the requirement for a large ‘fetch’ of wind on the upwind side of the emissions source, the ability to measure using a single open-path of point source is dependent on wind from a specific direction (Ulyatt et al. 1999). McGinn et al. (2006b) additionally determined that the location of the downwind sensor was critically important to the success of the technique, although assumptions about required distances will be required under ‘real world’ situations.
Open-path or mass balance methods can also be used to measure emissions from the manure pad, or soil/pasture surfaces (Sommer et al. 2004), this is particularly useful for NH$_3$, where the magnitude of fluxes can be significantly larger than N$_2$O or CH$_4$, and chamber designs need to be more complex. More recently, open-path/inverse dispersion techniques have been developed for measurement of NH$_3$ emissions from feedlots (Flesch et al. 2007), and applied by a number of authors to both commercial and research scale feedlots (Denmead et al. 2008; Loh et al. 2008; McGinn et al. 2008; McGinn et al. 2007; Rhoades et al. 2008; Todd et al. 2008). With the most common technique being the bLS inverse dispersion technique, as described above. The flux gradient method (also described above) has also been used to measure NH$_3$ fluxes (Baek et al. 2006; Todd et al. 2005). Although Denmead (1994) report on measurement of N$_2$O with micrometeorological measurements, the application of this approach to N$_2$O emissions from feedlots is not common practice.

The use of a tracer in combination with open-path, or point concentration measurements of emissions is a further modification of both tracer and micrometeorological measurements, and has been utilised for housed, lot feed and cell grazed animals (Griffith et al. 2008; Kaharabata et al. 2000). Kaharabata et al. (2000) observed only a 6% difference in CH$_4$ source strength using the modified tracer method, compared with using the barn as an enclosed chamber. They also assert that the technique was sensitive enough to capture intermittent fluctuations from feedlot housed cattle. They suggest that the technique may be useful in the measurement of emissions from structures which have complex aerodynamic flows. Griffith et al. (2008) also report on the use of a tracer technique in free grazing yearling cattle, determining emissions of 342 g/HEAD/day, higher than expected for yearling cattle (more consistent with grazing mature, high producing dairy cattle), however observations occurred for periods after feeding, which are known to be relatively high periods of emission. The tracer technique and an integrated horizontal flux technique agreed within 10% of daily mean CH$_4$ emissions.

### 2.10 Modelling Emissions

Measurement of greenhouse gas emissions can be expensive and complicated (Ellis et al. 2007), however prediction equations and models can be used to estimate emissions of enteric CH$_4$ and CH$_4$, NH$_3$ and N$_2$O from manure, without undertaking costly experiments for each estimation. Globally, there has also been limited validation of models commonly used for predicting emissions from feedlot cattle, particularly against respiration chamber experiments (Kebreab et al. 2006; Stackhouse et al. 2011).
2.10.1 Enteric CH$_4$

There are two main types of models used in the prediction of enteric CH$_4$ emissions; empirical and dynamic mechanistic models (Ellis et al. 2007). Empirical models relates nutrient intake to CH$_4$ output directly, whilst dynamic mechanistic models attempt to represent rumen fermentation biochemistry and predict CH$_4$ from these pathways. Mills et al. (2003) report that mechanistic models have been applied successfully; however they have limited applicability and usability in practical situations as they do not provide a quick solution based on dietary information.

As with measurement approaches, prediction of CH$_4$ using empirical approaches was originally approached from the perspective of estimating CH$_4$ as an energy loss for the prediction of metabolisable energy in feeds (Blaxter and Clapperton 1965; Moe and Tyrrell 1979) due to the cost associated with whole animal calorimetry. However, these approaches are still the basis of inventory estimates both in Australia and Internationally. As reported by Ellis et al. (2007) empirical models can be limited by the use of inputs which are not commonly measured. The applicability of these original models to beef feedlot cattle is questioned by number of authors (Ellis et al. 2009; Ellis et al. 2007; Kebreab et al. 2006; Kebreab et al. 2008), due to development based on different classes of animals (e.g. lactating dairy compared with beef cattle) and on different feeding regimes. Ellis et al. (2007) developed equations based specifically on beef cattle which performed better than existing Moe and Tyrrell (1979) and Blaxter and Clapperton (1965) equations when compared with North American and Canadian measured values. In a further publication Ellis et al. (2009) determined the equations of Ellis et al. (2007) to perform well using a database of Canadian and Northern American data. Further development and evaluation of statistical approaches for grazing beef cattle has been undertaken by Yan et al. (2009), Yan et al. (2000) and Ruiz-Suarez and Gonzalez-Avalos (1997).

Commonly, empirical models suffer from inaccuracies where the predictions rely on values (e.g. nutrient intakes and diet types) which are outside the range which were used in the development of the model (Ellis et al. 2007; Mills et al. 2003; Wilkerson et al. 1995). McCrabb and Hunter (1999) discuss this issue in more detail for Northern Australian production systems (in the context of a simple gross energy based equation), cited inaccuracies with predictive equations developed on B. Taurus cattle (and sheep) and on temperature forages compared with B. Indicus cattle and tropical forages. McCrabb and Hunter (1999) observed measured emissions 48% higher than estimates using Blaxter and Clapperton (1965) and 65% higher than IPCC Tier II approaches, although estimates from a high grain diet were more similar to measured values.
Mechanistic models have been more recently developed and evaluated for feedlot cattle (Kebreab et al. 2008) although they have received increased attention for lactating dairy cows (Benchaar et al. 1998; Mills et al. 2001; Mills et al. 2003). Kebreab et al. (2008) examined the MOLLY (Baldwin 1995) and COWPOLL (Dijkstra et al. 1992; Kebreab et al. 2004) mechanistic approaches in comparison with the IPCC (Tier II) and Moe and Tyrrell (1979) statistical approaches. In basic terms, MOLLY is a hydrogen balance model whilst COWPOLL is based around VFA production from three microbial populations. Kebreab et al. (2008) determined the MOLLY mechanistic model to produce the best estimates of CH$_4$, however the IPCC Tier II approach showed comparable results to MOLLY. The mechanistic approach was more responsive to dietary changes, including differences between maize and barley type rations. Kebreab et al. (2008) also assert that the model can account for biohydrogenation occurring with fat supplementation, which provides significant advantages to quantifying mitigation activities.

Mechanistic models have the advantage of being better able to assess the effectiveness of mitigation options, applied at the farm or national level (Kebreab et al. 2008) although the detailed input required may make them less suitable for inventory applications, or for use in individual operations. However, the more detailed models may provide useful information about the application of mitigation options and could be utilised to inform policy.

### 2.10.2 Modelling Manure Emissions

Emissions from animal manure can be predicted using a number of approaches. As with enteric CH$_4$, statistical models which relate dietary and animal factors to excretion of nutrients and emissions have been developed. However, unlike enteric CH$_4$, emissions of CH$_4$, NH$_3$, and N$_2$O from animal manure are significantly influenced by environmental factors (see section) making predictions from a (relatively simple) statistical model considering only dietary factors considerably less accurate. In the context of manure emissions, an empirical model uses simple emissions factors or equations to link manure composition to emissions. A dynamic mechanistic model attempts to represent processes within the manure, and predict emissions based on these processes.

### 2.10.3 Ammonia

Both empirical and mechanistic approaches have been used to estimate NH$_3$ emissions from manure Rotz and Oenema (2006). The USEPA use empirical methods to model NH$_3$ emissions from a variety of farming practices in the United States (Bunton et al. 2007), whilst Monteny and Erisman (1998) and Ni (1999) report on mechanistic models of NH$_3$ emissions.
from slurry and liquid manure. The IPCC (IPCC 2006) approach for modelling NH$_3$ emissions from feedlot manure estimates N excretion (both urine and faecal) and applies an emission factor of 0.3, or 30% of excreted N. However, it has been observed that NH$_3$ emissions range from 25 to 50% of excreted N (Hristov et al. 2011). Under Australian conditions, NH$_3$ emission factors have been estimated at 0.59 and 0.94 kg NH$_3$/kg excreted N (Chen et al. 2009; Loh et al. 2008). Despite increasing knowledge quantifying emissions from feedlots (Flesch et al. 2007; McGinn et al. 2007; Todd et al. 2008; Todd et al. 2005) relatively few studies have directly compared predicted and measured emissions for NH$_3$ using statistical approaches. The observed differences in emission factors highlights potential for considerable inaccuracy, particularly associated with inability to account for the impact of climatic conditions of NH$_3$ emissions using a single emissions factor.

Hristov et al. (2011) recommends a process based mechanistic modelling approach is used, and reviews modelling approaches in detail. Briefly, the advantage of a mechanistic model is that these account for physical and biochemical properties which influence emissions (Rotz and Oenema 2006). This type of model would provide additional information regarding the dynamics of emissions, in contrast to an empirical model, which would produce a single value (or range of value) for the given inputs. Rotz and Oenema (2006) report on the development of a process based model for volatilisation of NH$_3$ from different manure management practices, designed for use in on farm situations. This model predicted NH$_3$ emission rate as a function of NH$_4$-N content of manure, temperature, pH, manure moisture content and exposed manure surface area, as well as storage methodology. The dynamic model described predicted NH$_3$ losses averaging 40.6% of total N for a simulated feedlot in Pennsylvania and 47.1% for a simulated feedlot in Texas, compared with typical measured values of 45%. This demonstrates potentially increased accuracy with a dynamic model.

### 2.10.4 Nitrous Oxide

As it currently stands, the only approach to predict N$_2$O emissions specifically from a feedlot is statistical (as published by the IPCC (IPCC 2006). This approach predicts N excretion, and applies a system specific emission factor to estimate N$_2$O emissions (Brown et al. 2001). The issue with this type of approach is it does not account for differences in emission rates spatially or seasonally in line with climatic conditions. There is also no ability to allow for mitigation options to be assessed (Saggar et al. 2004a). There does not appear to have been any substantial validation of this approach for feedlot environments, coupled with the lack of measurement of N$_2$O from feedlot manure pads. Boadi et al. (2004b) determined manure pack
emissions to be approximately 50% of those predicted by the IPCC, although this appears to be expressed in terms of CO$_2$-e and a combination of manure CH$_4$ and N$_2$O.

Nitrous oxide from soils can also be predicted using mechanistic models, such as the DNDC model (Li et al. 1992a; Li et al. 1992b), NGAS-DAYCENT (Parton et al. 1998; Parton et al. 2001; Parton et al. 1996), WNMM (Li et al. 2007) which predict N$_2$O evolution from agricultural soils using knowledge of decomposition, crop and pasture growth, soil moisture, C cycling, N cycling and soil climate to estimate nitrification and denitrification. Muller et al. (1997) determined that the best estimates from a mechanistic model were achieved when distinct parameters were used for periods in which soil moisture was constant compared with periods where soil moisture status changed rapidly. This underlies the effectiveness of a process based model compared with a single emissions factor. Brown et al. (2001) compared IPCC estimates with the DNDC approach to estimated N$_2$O emissions at the farm scale (from dairy farms). The main differences between the approaches was in the prediction of N$_2$O from soil and from indirect sources (kg/ha). Soil emissions were higher using the DNDC approach, as were indirect emissions. They suggest that an issue with the current DNDC model is that it is not able to deal with returns of N via manure deposited at grazing (they used 5 discrete addition events). Similarly, Saggar et al. (2004a) modified the DNDC model to include excreta from grazing animals, and the value for WFPS and observed effective simulation of WFPS and N$_2$O emission pulses from grazed dairy pastures. Further calibration of the model for specific environments is needed for grazing environments (Brown et al. 2001) and it follows that the same would be required for feedlot environments. Models of this type do not appear to have been applied under feedlot environments, it is possible that considerable characterisation of N transactions in the manure pad and in the soil under pens would be required to produce accurate estimates. Although a mechanistic/ process based model may provide better estimates under site specific conditions (Brown et al. 2001) they are unlikely to be applied under inventory purposes, therefore further investigation and development of emissions factors for N$_2$O under feedlot environments is required.

2.10.5 Manure Methane

Consistently with the other manure emissions, CH$_4$ can be modelled using both process and emissions factor (empirical) approaches. Although as reported by Wagner-Riddle et al. (2006) modelling of manure CH$_4$ emissions is largely carried out using emissions factor approaches—such as the IPCC methodology. The IPCC methodology (IPCC 2006) estimates maximum manure CH$_4$ production potential for a given manure type using volatile solids content of
manure and a CH₄ conversion factor which takes climate and manure storage into account (Wagner-Riddle et al. 2006).

One of the major issues with an “emission factor” approach is in the suitability of the set emissions factor for the management system used, given the considerable impact that management and environmental conditions have on potential emissions of CH₄. Boadi et al. (2004b) suggested that there would be considerable differences in methane conversion factor (MCF) estimates of the IPCC and from Canadian feedlots, due to differences in cattle and manure management, feeding and climate. This may also be the case for Australian feedlots, with studies reporting variation in temperature, moisture and time of exposure of manure to air producing large differences in CH₄ production from feedlot managed cattle (Boadi et al. 2004b). Gonzalez-Avalos and Ruiz-Suarez (2007) report CH₄ emission factors up to 17 times smaller than those reported by the IPCC, and suggest CH₄ conversion factors of 1.07, 1.71 and 0.76% for intensive beef in Mexico (Gonzalez-Avalos and Ruiz-Suarez 2007), compared with 1.5 to 2.0% under IPCC guidelines (IPCC 2006). Although Boadi et al. (2004b) measured CH₄ emissions from manure of less than 2 g/head/day (for low and high forage rations), representing emissions from manure of be approximately 50% of those predicted by the IPCC.

Wagner-Riddle et al. (2006) reports the study of Sommer et al. (2004) to be the only process based model of CH₄ emissions from manure, which covers only CH₄ from storage and spreading of dairy slurry. There does not appear to be any current process based or dynamic models of CH₄ emissions from manure directly applicable to a feedlot.

2.11 Summary

The objectives of this thesis are to identify and quantify sources of variation in measured emissions of greenhouse gas between two commercial feedlot sites and in two seasons and test the validity of the modelling approach used specifically for feedlots and compare predicted and measured emissions. The thesis also aims to quantify the link between animal behaviour and diurnal emissions patterns.

This review has demonstrated that there are a variety of sources of variation, which could potentially contribute to differences in measured emissions at two commercial feedlot sites. Enteric CH₄ emissions are influenced to a considerable extent by ration composition. The higher grain rations fed in feedlot environments will produce lower CH₄ emissions than forage based rations. Differences in ration composition also influence emissions of manure CH₄, N₂O and NH₃. However, these emissions are more strongly controlled by physical and
biochemical factors in the substrate and the environment- which are likely to contribute to a greater extent to differences in emissions between sites and seasons.

The review explored approaches for modelling emissions of methane, as well as discussing the approach used in the national inventory methodology for predicting emissions of CH\textsubscript{4}, N\textsubscript{2}O and NH\textsubscript{3} from manure. Potential inaccuracies in these models were identified. Finally, this review explored methods for measurement of emissions, from individual animal and small chamber, to micrometeorological approaches which are suitable for the commercial sites utilised in this study. These approaches will enable monitoring of diurnal emissions patterns, which have been anecdotally observed to be connected with animal behaviour.

Greenhouse gas emissions from feedlots vary from those in grazing systems, due to animal, feed and environmental conditions. However, issues with model inaccuracy, and a lack of benchmarking data may lead to over estimation of emissions from commercial feedlots in Australia.
Chapter 3. General Methodology

This project aimed to evaluate aspects of greenhouse gas emissions from Australian beef feedlots, including the accuracy of current approaches for estimating greenhouse gas emissions, and interactions between animal behaviour and diurnal emissions profiles. Specifically, the National Inventory Methodology (AGO 2006) for estimating greenhouse gas emissions from feedlots was evaluated, validated with published data and estimate compared to measurements from Australian feedlot systems.

3.1 Site Selection

This study was conducted in conjunction with a larger project measuring emissions from Australian beef feedlots (Chen et al. 2009). Two feedlots, representative of Australian beef feedlots production systems, were selected for the study. A Southern Site was located near Charlton, Victoria (36°21'41" S, 143°24'5" E), and a Northern Site was located near Dalby, Queensland (27°8'14" S, 151°26'3" E). The Northern Site represented a typical production system in the Northern / subtropical zone with a distinct wet and dry season, warm wet “summers” and cool dry winters (Figure 3.1) The Southern Site represents typical Southern production systems in the Mediterranean/ temperate zone with dry hot summers and cool wet winters (Figure 3.2)

![Graph](image.png)

**Figure 3.1** Difference between mean minimum and maximum temperatures and mean monthly rainfall for the Northern feedlot site, 1992-2009, Bureau of Meteorology Climate Statistics (Dalby Airport Station). Mean annual maximum temperature; 26.9°C, mean annual minimum temperature; 12°C, total annual rainfall; 606.2 mm.
Data were collected during eight two-week field campaigns at the two sites, in summer and winter, of two consecutive years (Table 3.1) The Southern feedlot has a maximum capacity of 20 000 head, but was operating at between 13 000 and 18 000 head at the time of the four field campaigns. The Northern Site has a maximum capacity of about 17 000 head, but was operating between maximum capacity (16 800) and as low as one third of capacity (6 200) over the four field campaigns. Both feedlots were operating at or above the minimum space requirement of 9 m$^2$/head. All experiments were conducted in accordance with the Australian Code of Practice for the Care and Use of Animals for Scientific Purposes (www.nhmrc.gov.au).

**Figure 3.2** Difference between maximum and minimum monthly temperatures and mean monthly rainfall for the Southern feedlot site, temperature 1966-2000, rainfall 1966-2009, Bureau of Meteorology Climate Statistics (Donald Station-Charlton station has been closed since 1976). Mean annual maximum temperature; 21.3°C, mean annual minimum temperature; 8.8°C, total annual rainfall; 380.8 mm.

### 3.2 Emissions Measurements

A micrometeorological approach based on measurements of gas concentrations with open-path gas analysis and a backward Lagrangian Stochastic (bLS) dispersion model was used to calculate CH$_4$, NH$_3$ and N$_2$O emissions from beef cattle feedlots. This methodology has been successfully used to measure greenhouse gas emissions from beef cattle feedlots elsewhere in the world (Flesch et al. 2007; Laubach and Kellier 2005; Van Haarlem et al. 2008), but this was the first such Australian study. Full details of the methodology for emissions measurements can be found in Chen et al. (2009).
Table 3.1 Dates of sampling period, number of head and proportion of pens occupied during eight field campaigns at two beef cattle feedlots, Northern (Queensland) and Southern (Victoria) during winter 2006, summer 2007, Winter 2007 and Summer 2008.

<table>
<thead>
<tr>
<th>Campaign</th>
<th>Start Date</th>
<th>End Date</th>
<th># Head</th>
<th>Total Pen area (ha)</th>
<th>Occupied Pen Area (ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>2006 Winter</td>
<td>Queensland 24- Aug</td>
<td>31- Aug</td>
<td>16817</td>
<td>25.6</td>
<td>25.2</td>
</tr>
<tr>
<td></td>
<td>Victoria 1- Aug</td>
<td>10- Aug</td>
<td>18092</td>
<td>23.7</td>
<td>22.0</td>
</tr>
<tr>
<td>2007 Summer</td>
<td>Queensland 29- Jan</td>
<td>8- Feb</td>
<td>13583</td>
<td>25.6</td>
<td>25.1</td>
</tr>
<tr>
<td></td>
<td>Victoria 19- Feb</td>
<td>1- Mar</td>
<td>16713</td>
<td>23.7</td>
<td>22.7</td>
</tr>
<tr>
<td>2007 Winter</td>
<td>Queensland 3- Sep</td>
<td>8- Sep</td>
<td>10681</td>
<td>25.6</td>
<td>18.6</td>
</tr>
<tr>
<td></td>
<td>Victoria 1- Aug</td>
<td>10- Aug</td>
<td>13074</td>
<td>23.7</td>
<td>21.8</td>
</tr>
<tr>
<td>2008 Summer</td>
<td>Queensland 31- Jan</td>
<td>7- Feb</td>
<td>6192</td>
<td>25.6</td>
<td>13.7</td>
</tr>
<tr>
<td></td>
<td>Victoria 25- Feb</td>
<td>5- Mar</td>
<td>12926</td>
<td>23.7</td>
<td>20.8</td>
</tr>
</tbody>
</table>
3.2.1 Open-Path Spectroscopy

An open-path mid-infrared Fourier Transform Infrared (OP-FTIR) spectrometer constructed by the University of Wollongong measured CH₄, CO₂, N₂O and NH₃ simultaneously. Two open-path (OP) CH₄ lasers and two OP NH₃ lasers (GasFinder2.0, Boreal Laser Inc, Edmonton, Alberta, Canada) measured CH₄ and NH₃ (Plate 3.1).

3.2.2 Micrometeorology

A micrometeorological station (Plate 3.1) including a CSAT three-dimensional sonic anemometer, an OP gas analyser (Licor 7200) and a data logger (CR5000, Campbell Scientific, Logan, UT, USA); recorded wind speed, wind direction, and surface heat flux, at a central location on the feedlot. From these data, turbulence statistics including Monin-Obhukov length (L), friction velocity (u*) and surface roughness (z₀), were calculated to characterise atmospheric dispersion and turbulence across the feedlot. These data were averaged over 15-minute time intervals, to coincide with the line-averaged gas concentration data from the sensors. This methodology has been used in conjunction with the emissions measurements described above in a number of studies (McGinn et al. 2007).

3.2.3 Atmospheric Dispersion Modelling

Atmospheric dispersion modelling was carried out using the backward Lagrangian Stochastic (bLS) model as described by Flesch et al. (2004). Its application was facilitated by the use of WindTrax software package (Thunder Beach Scientific, Nanaimo, BC, Canada). Four source areas were identified and located in the feedlots, including occupied cattle pens, empty cattle pens, effluent ponds, and manure piles. Source areas and OP gas analysers (sensors) were geospatially referenced in the WindTrax model (Plate 3.2). Using a backward Lagrangian Stochastic (bLS) method (Flesch et al., 2004), WindTrax estimated CH₄, NH₃, N₂O and CO₂ fluxes from source areas, modelling back from the line-averaged gas concentrations at the sensors, via atmospheric dispersion and turbulence patterns defined by the micrometeorological data, to flux estimates from the source areas. Backward-modelled touchdowns define the source area from which emissions came, resulting in the measured concentrations at each sensor, during each 15-minute interval. The modelling process was repeated for all instruments and gases, over all time intervals, in all campaigns. In addition, for CH₄ and NH₃, average gas fluxes were determined from a combination of instruments (two OP lasers and the OP-FTIR), when multiple flux measurements from multiple instruments were available in each 15-minute interval.

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Filtering criteria were applied to remove unreliable data, following McGinn et al. (2006b), Flesch et al. (2007) and McGinn et al. (2007). Data for removal included Boreal laser data where the light signal intensities were less than 1800 or greater than 13000, or where the coefficient of variation ($r^2$) was less than 0.50. Meteorological data for removal included the following: absolute Monin-Obukov Length ($|L|$) < 10 (i.e., -10 > $L$ < 10); surface roughness ($z_0$) < 0 m or $z_0$ > 0.9 m; friction velocity ($u^*$) < 0.15. WindTrax flux estimates that had a footprint of less than 10% of the source area were removed.
Plate 3.1 Tuneable diode laser (left) and micrometeorological equipment (centre) at the Southern Site

Plate 3.2 An example of a WindTrax project screen depicting backward Lagrangian Stochastic (bLS) modelling of gas flux from a single sensor, during a single 15-minute period. Gas-flux is estimated as the rate of gas emission ($g_{\text{GAS}} \cdot m^2/s$) from the contributing source area (the footprint of red touchdowns) that would have resulted in the measured gas concentration at the sensor, given the prevailing wind speed, wind direction and turbulence measured by the micrometeorological station.
3.3 Units of Measurement/Calculations

Raw data from the open-path measurements is reported as line averaged concentrations. The bLS model as described above uses concentrations at the sensors (instruments), background concentrations of gases, assigned source areas and atmospheric dispersion and turbulence patterns to estimate flux from the source area. Flux can be reported in g/ha/day, or further calculations utilising number of livestock in the source area used to calculate g/head/day.

Emissions of CH$_4$ are reported here as fluxes of gas in g/head/day. The primary source of CH$_4$ in the feedlot is the enteric fermentation of the animals themselves; therefore it is most appropriate to express these emissions on a per animal basis.

Emissions of N$_2$O and NH$_3$ are reported here as g/head/day. Deposition of manure is considered on an individual animal basis under the National Inventory Methodology (AGO 2006) although emissions continue from the manure pad for a number of days after deposition and emissions could also be reported on a per ha basis.

3.4 Statistical Analysis

Analysis of variance (ANOVA) of emissions due to campaign (year, season and site) and diurnal variation (hour within campaign) were tested using general linear models in SAS (v9.1.3, SAS Institute Inc., NC, USA) by the ordinary least-squares method. Prior to ANOVA, data were transformed by natural log, to meet the assumption of homoscedasticity. Transformed data with a residual (actual-predicted) value, more than three standard deviations greater than, or less than, the mean were removed as outliers, to meet the assumption of standard normal distribution, following a Shewart Approach. Refer to Chen et al. (2009) for further detailed of the statistical approaches.

3.5 Feedlot Data Collection

Information was collected from each feedlot during each of the measurement campaigns, in the form of standard computer generated management reports; pen inventories, bunk sheets and stockman’s reports. These included number of animals in each pen, animal sex, estimated weight of animals at entry to the feedlot, estimated current weight, days on feed, cattle class, rations and amount of feed offered. This allowed animal movements, live weights and feed offered to be monitored for the duration of the measurement campaign.
3.6 Emissions Models

3.6.1 Structure
The emissions model is based on the National Inventory Methodology for the Estimation of Greenhouse Sources and Sinks (AGO 2006) and was based on a Microsoft Excel spreadsheet. The model integrates animal and production system data into a range of equations predicting greenhouse gas production from the feedlot system. The primary sources of greenhouse gas emissions from the feedlot are the animals themselves (CH$_4$), and the decomposition of manure which produces CH$_4$ and nitrogenous gases (NH$_3$ and N$_2$O), therefore the model incorporates equations to predict CH$_4$ from enteric fermentation; Blaxter and Clapperton (1965), Moe and Tyrell (1979), Ellis et al. (2007), Ellis et al. (2009) and IPCC Tier I and II (IPCC 2006), as well as for CH$_4$ emission from manure and N$_2$O and NH$_3$ from the nitrogenous compounds in manure.

The objective of this approach is to utilize information available from commercial feedlot operations in the prediction of greenhouse gas emissions. The basic structure of the model is outlined in Appendix 9.2, figure 9.3.

3.6.2 Data Source
The model has been designed so that data obtained from commercial feedlot systems can be utilized. Feedlot management software produces reports which contain detailed information about the current crop of cattle, including numbers, placement weights, estimated current weights, days on feed, class amounts of feed offered and detailed ration information. The model selected was required to utilize this type of data, with the addition of commonly measured feed nutritive characteristics.

3.6.3 Intake Model
Although the country specific methodology for dairy and non-feedlot beef cattle uses equations to predict intake based on requirement, for feedlot cattle a standard value of intake is used for each class (Table 3.2). The model also includes the equation reported by Minson and McDonald (1987); which predicts intake based on growth and liveweight of beef cattle (Equation 3.1), intake as a percentage of liveweight, and offered feed (as reported by feedlot operators or in published data). The range of approaches was built into the model to allow comparison of the effects on estimated emissions.
Table 3.2 Standard intake values (g/head/day) for feedlot cattle based on the National Inventory Methodology for the estimation of greenhouse sources and sinks (2006)

<table>
<thead>
<tr>
<th>Class</th>
<th>Source</th>
<th>Dom.</th>
<th>Export</th>
<th>Jap Ox</th>
</tr>
</thead>
<tbody>
<tr>
<td>Standard Value</td>
<td>AGO (2006); van Sliedregt et al. (2000)</td>
<td>9.8</td>
<td>11.7</td>
<td>11.0</td>
</tr>
<tr>
<td>% Live weight</td>
<td>AGO (2006) Working group estimate</td>
<td>2.4</td>
<td>2.2</td>
<td>2.0</td>
</tr>
</tbody>
</table>

Equation 3.1 Prediction of dry matter intake from the growth and live weight of beef cattle based on Minson and McDonald (1987).

Intake kg DM/day = (1.185 + 0.00454 W - 0.0000026 W^2 + 0.315 LWG)^2

Where W= weight in kg, LWG= live weight gain in kg/day.

3.7 Methane Model

The modelling approach for CH₄ emissions from the feedlot considered two aspects, emissions from the animals themselves through enteric fermentation, and emissions from deposited manure.

3.7.1 Enteric CH₄ Emissions

The IPCC national greenhouse gas inventory guidelines suggests three levels of complexity for prediction of enteric emissions (Kebreab et al. 2008), Tier I, which is equivalent to 164 g/head/day for feedlot cattle in Oceania (IPCC 2006), Tier II and Tier III, which is country specific (as discussed above). The Tier II model (under the Australian National Inventory Methodology) utilizes the gross energy content of the ration and a standard emission factor of 3.0 (±1) % for feedlot cattle (AGO 2006). The model includes five equations linking nutritive value to enteric CH₄ production; Blaxter and Clapperton (1965), Moe and Tyrrell (1979), IPCC Tier II, Ellis et al. (2007) and Ellis et al. (2009), in order to evaluate the accuracy and changes in variation around the predicted values from each equation, and the sensitivity to each equation to changes in animal and feed parameters. The models predict CH₄ in MJ CH₄, which is converted to g/head/day using a factor of 55.22 (Brouwer 1965) as the energy content of CH₄.

Moe and Tyrrell (1979) (Equation 3.2) is utilized as the Australian country specific model (Tier II) for feedlot cattle. It was developed based on dairy cattle fed high grain diets and
relates CH₄ production to the CH₄ carbohydrate fractions (soluble residue, cellulose and hemicelluloses) in the diet.

**Equation 3.2** Enteric CH₄ Production; Moe and Tyrell (1979)

\[
MJ \text{ CH}_4/\text{day} = 3.406 + 0.510 \text{ SRi} + 1.736 \text{ HEMi} + 2.648 \text{ CELLi}
\]

\[R^2 = 0.73, \text{ S}_{\text{y,x}} = 0.56\]

Where SRi= Intake of soluble residue, HEMi= Intake of hemicellulose and CELLi = Intake of cellulose

The equation of Blaxter and Clapperton (1965) calculates the gross energy content of the diet and estimates how much is converted to CH₄ based on digestibility at maintenance energy requirement and the level of feed intake relative to intake required for maintenance.

The approach of Blaxter and Clapperton (1965) (Equation 3.3) forms the basis of a number of national inventories; it is used as the Australian country specific model (Tier II) for dairy and grazing beef cattle. It was included in the current model primarily for the purposes of comparison, and to examine the observation (Ellis *et al.* 2007; Mills *et al.* 2003) that emission prediction based on diets outside those used for developing the model (e.g. high forage dairy rations compared with high grain feedlot rations) are consistently over estimated.

**Equation 3.3** Enteric CH₄ production; Blaxter and Clapperton (1965)

\[
MJ \text{ CH}_4/\text{day} = 1.3 + 0.112 \text{ DMD} + \text{ Ri} (2.37 - 0.050\text{DMD})
\]

\[R^2 = \text{Not Reported}\]

Where DMD = dry matter digestibility (assumed to be 80% for feedlot cattle), and Ri= relative intake; actual intake compared with intake required for maintenance.

A significant issue surrounding the use of both Blaxter and Clapperton (1965) and Moe and Tyrrell (1979) is that they were both developed based on dairy cattle and have difficulty predicting emissions outside the range on which they were developed (Ellis *et al.* 2007; Wilkerson *et al.* 1995) More recently a number of studies have developed models based on beef cattle (Ellis *et al.* 2009; Ellis *et al.* 2007) and a smaller number have evaluated equations for feedlot cattle (Ellis *et al.* 2009; Kebreab *et al.* 2008).

Ellis *et al.* (2007) developed a range of equations based on commonly measured dietary variables which could be used to predict CH₄ production. The model reported with the lowest root mean square prediction error (RMSPE), and therefore considered to produce the most accurate estimates (Equation 3.4) was selected for inclusion into the biophysical model in this
study. This equation utilizes fibre fractions (lignin and ADF) as well as metabolisable energy content.

**Equation 3.4** Enteric CH₄ Production; Ellis et al. (2007)

\[
\text{MJ CH}_4/\text{day} = 2.94 + 0.59 \text{ MEi} + 1.44 \text{ ADFi} - 4.16 \text{ LIGi}
\]

\[ R^2 = 0.85, \text{ RMSPE} = 14.4\% \]

Where MEi= metabolisable energy intake in MJ/day, ADFi= intake of acid detergent fibre in kg/day and LIGi= intake of lignin in kg/day.

This equation was developed based 14 different studies, however only a small number of these studies were feedlot cattle and include also include data from cattle fed high levels of pasture and forages, which is likely to produce a different relationship to high grain diets. Ellis et al. (2009) developed a further equation for beef cattle (Equation 3.5), using 12 different studies, with an increased proportion of feedlot based studies. This equation is similar to Moe and Tyrrell (1979) in that it utilizes the hemicelluloses and cellulose content of the ration; however it also includes the fat content. This has the potential to significantly improve estimates, as inclusion of lipids in the diet.

**Equation 3.5** Enteric CH₄ production; Ellis et al. (2009)

\[
\text{CH}_4 \text{ MJ/day} = 2.72 + 0.0937 \text{ MEi MJ/day} + 4.31 \text{ CELLi} - 6.49 \text{ HEMLi} - 7.44 \text{ FATi}
\]

\[ R^2 = 0.74, \text{ RMSPE} = 26.9\% \]

Where MEi= metabolisable energy intake in MJ/day, CELLi= cellulose intake in kg/day, HEMLi= hemicellulose intake in kg/day and FATi= fat intake in kg/day.

These two equations (Equations 3.4 and 3.5) have been included in the biophysical model to assess if there is any improvement in prediction of emissions from Australian feedlot cattle using equations derived from beef cattle data.

### 3.7.2 Manure CH₄ Emissions

Manure CH₄ is estimated according to IPCC standards (IPCC 2006). Dietary digestibility and intake are used to predict volatile solids production in manure (Equation 3.6). A standard emissions factor is then applied to estimate the amount of CH₄ produced (Equation 3.7). The dry packing arrangement used in Australian feedlots is thought to result in only a small amount of the potential CH₄ generated.
Equation 3.6 Volatile solids excretion

\[
\text{Volatile solids kg/day} = i \times (1 - \text{DMD}) \times (1 - \text{A})
\]

Where \(i\) = intake, \(\text{DMD}\) = dry matter digestibility expressed as a fraction (assumed to be 0.8 for feedlot diets) and \(\text{A}\) = ash content expressed as a fractions (assumed to be 0.08 for feedlot rations).

Equation 3.7 Methane production from manure

\[
\text{Manure CH}_4 \text{ kg/head/day} = \text{VS} \times \text{B} \times \text{MCF} \times \rho
\]

Where \(\text{VS}\) = volatile solids (kg/day), \(\text{B}\) = the emissions potential of \(\text{CH}_4\) (-0.17 m\(^3\) \text{CH}_4/kg VS), \(\text{MCF}\) = the \(\text{CH}_4\) conversion factor for the manure management system (for Northern Australia this value is 5% and for Southern Australia this factor is 1.5%, based on climatic conditions). \(\rho\) = the density of \(\text{CH}_4\) (0.662 kg/m\(^3\)).

3.8 Nitrogen Model

There are two primary components to the N portion of the biophysical model. The equations developed by the (SCA 1990) and Freer et al. (1997) as published in the National Inventory Methodology (AGO 2006) are used to estimate N transactions within the animal; N retained and excreted as urine and faecal N. The second component of the N model uses a set emissions factor for the drylot management system to estimate the emission of \(\text{N}_2\text{O}\) and volatilisation of \(\text{NH}_3\).

The SCA (1990) methodology is used in the National Inventory Methodology (2006) as it was developed in Australia and was therefore selected in preference to the IPCC standard (for N excretion, which is based on NRC guidelines (NRC 1996). This method uses a mass balance approach (N output= N input- N storage). This method for prediction N transactions was selected for the model as it is the Australian standard for calculating nutrient requirements for beef cattle, and as shown by its selection for the National Inventory Methodology (AGO 2006) over the IPCC standard values, is likely to form the basis of any methodology estimating N transactions in cattle.

Nitrogen intake is calculated based on the crude protein concentration of the ration, with crude protein considered a fixed 6.25 times N content. The N excreted in the faeces is calculated (Equation 3.8) based on equations developed by the SCA (1990) and considered the indigestible fraction of undegraded protein from feed, microbial protein and endogenous faecal N.
**Equation 3.8** Faecal N Excretion

\[
\text{Faecal N g/head/day} = (0.3 (\text{CPI} \times (1- ((\text{DMD} + 10)/100))) + 0.015 \times \text{ME} \times i \times 0.008) + (0.0152 \times i)/6.25
\]

Where CPI = crude protein intake, DMD = dry matter digestibility, ME = metabolisable energy content of the ration, i = intake and 6.25 represents a standard factor to convert crude protein to N.

Similarly, the amount of N retained in the body tissue is calculated (Equation 3.9) based on growth rates and relative intake of the animal based on level of intake required for maintenance and size of the animal relative to mature body size (based on a standard reference weight, for feedlot cattle this is 660 kg, AGO 2006).

**Equation 3.9** Nitrogen Retention

\[
\text{N retained g/head/day} = \frac{((0.212 - 0.008 (\text{Ri} - 1) - (0.140 - 0.008 (\text{L} - 2)/(1 + \exp^{6.25(0.04)})) \times (\text{LWG} \times 0.92))}{6.25}
\]

Where Ri = relative intake, determined by dividing feed intake by the intake required for maintenance, Z = relative size and LWG = live weight gain.

Finally, urinary N excretion is calculated using a mass balance approach (Equation 3.10), subtracting faecal N, retained N and dermal N loss from N intake.

**Equation 3.10** Calculation of Urinary N excretion using a mass balance approach

\[
\text{Urinary N g/head/day} = (\text{CPI}/6.25) - \text{N Retained} - \text{Faecal N} - ((1.1 \times 10^{-4} \times \text{W}^{0.75})/6.25)
\]

Where CPI = crude protein intake, 6.25 represents a standard figure to convert nitrogen to crude protein and W = live weight, with W^{0.75} = metabolic live weight.

The second component of the N model uses standard emissions factors for drylot management systems to convert the excreted N volatile NH₃ and emitted N₂O (Equations 3.11 and 3.12).

**Equation 3.11** Estimated N₂O Emissions from Faecal N

\[
\text{Faecal N₂O g/head/day} = \text{Faecal N} \times \text{MMS} \times \text{EF}_{\text{faeces}} \times \text{C}_\text{g}
\]

**Equation 3.12** N₂O Emitted from Urinary N
Urine $N_2O$ g/head/day = Urinary N x MMS x EF$_{mms}$ x C$_g$

Where MMS indicates the fraction of manure managed under the manure management system (for feedlot systems this is set at 1, with 100% of manure being managed under the dry lot management system), EF$_{mms}$ = the $N_2O$ emission factor for the manure management system (for drylot manure management systems the emission factor is 0.02. C$_g$ is a factor used to convert the elemental mass of $N_2O$ to molecular mass = 44/28.

Volatile NH$_3$ is estimated in a similar way, with the fraction of NH$_3$ volatilised under the manure management system set as 0.03 for feedlot systems (Equations 3.13 and 3.14).

**Equation 3.13** Loss of volatile NH$_3$ from faecal N

Volatile Faecal N g/head/day = Faecal N x Fraction of N volatile

**Equation 3.14** Loss of volatile NH$_3$ from urinary N

Volatile Urinary N g/head/day = Urine N x fraction of N volatile

Where fraction of N volatile is set at 0.03 for feedlot systems

### 3.9 Assumptions

The assumptions primarily deal with the gross composition of the feed (digestibility, energy), the conversion of energy to CH$_4$ in the animal and manure and the emission of $N_2O$ and NH$_3$. The main assumptions used in the calculations are outlined in Table 3.3.

Emissions of nitrogenous gases and CH$_4$ from manure are considered to be dependent on the management of manure. For the purposes of the model all manure is assumed to be managed under the dry lot management system. Emission factors for CH$_4$ are taken from the Australian Greenhouse Office (AGO 2006) (0.015 temperate, 0.05 warm) rather than the IPCC guidelines (IPCC 2006). However emission factors for nitrogenous gases are taken from the IPCC standard under the guidelines of the National Inventory Methodology (AGO 2006).

Detailed ration composition (cellulose, hemicelluloses and lignin) is not commonly reported, or provided in ration information. Where these values were required for model calculations as in Moe and Tyrrell (1979), Ellis *et al.* (2007) and Ellis *et al.* (2009), they were calculated based on Givens and Moss (1990) and the Cornell net crude protein system (CNCPS; Sniffen *et al.* 1992). This was required for all feed lignin and ADF values, as well as detailed composition of some unusual feedstuffs (protein meals). The gross energy of individual feeds is assumed to be 18.4, with a digestibility of 80% (AGO 2006).
Some assumptions were required to be made about the production systems in order to classify the output and provide information about basic management practices. There are assumed to be three production categories short fed domestic (<100 days), short fed export (100-200 days) and long Fed Export (>200 days; AGO 2006, van Sliedregt et al. 2000). Cattle were grouped into these classes based on data provided by the feedlot. This influences the standard reference weight, and intakes which can be predicted using the class of cattle (as a comparison to offered feed) based on van Sliedregt et al. (2000).

Table 3.3 Assumptions of the standard model

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Set Value</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Energy content of CH₄</td>
<td>55.22 MJ/kg</td>
<td>Brouwer (1965)</td>
</tr>
<tr>
<td>Density of CH₄</td>
<td>0.662 kg/m³</td>
<td></td>
</tr>
<tr>
<td>Gross energy content of feed</td>
<td>18.4 MJ/kg DM</td>
<td>SCA (1990)</td>
</tr>
<tr>
<td>Dry Matter Digestibility</td>
<td>80%</td>
<td></td>
</tr>
<tr>
<td>Yₘ (CH₄ conversion factor)</td>
<td>3%</td>
<td>IPCC (2006)</td>
</tr>
<tr>
<td>MCF (manure CH₄)</td>
<td>0.015 temperate</td>
<td>AGO (2006)</td>
</tr>
<tr>
<td>EF N₂O</td>
<td>0.02 kg N as N₂O/ kg N</td>
<td>IPCC (2006)</td>
</tr>
<tr>
<td>EF NH₃</td>
<td>0.03 kg N as NH₃/ kg N</td>
<td>IPCC (2006)</td>
</tr>
</tbody>
</table>

3.10 Validation

The first stage of the modelling process was the validation (described in chapter 4) of the various models using published data. Separate databases were developed for enteric CH₄, N transactions and N emissions.

3.10.1 Enteric CH₄ Model

In order to evaluate the ability of the model to predict changes in emissions associated with changes in parameters, published results were used to test the model. The basic model was evaluated for its ability to reflect the physiological changes associated with changes in diet and intake, primarily changes in digestibility and intake, and changes in the fermentation products produced by different diet types (e.g. propionate dominant in a high grain diet vs. acetate dominant in a high forage diet).

Full details of the studies selected for validation of the enteric CH₄ emissions model can be found in Chapter 4, Table 4.1. Briefly; studies in the database were published since 2000 and...
focused on manipulating dietary factors on CH\textsubscript{4} emissions from feedlot cattle. The primary factor under investigation was the energy density of the diet; high vs. low forage, with the exception of Hegarty \textit{et al.} (2007) which used the same ration at different intake levels. Rations used in the CH\textsubscript{4} validation (Table 4.2) studies ranged in forage proportion from 0.09 to 0.7. With ME values between 8.8 and 12.1 MJ/kg and CP from 120 to 150 g/kg. Animal production data used in the validation of the enteric CH\textsubscript{4} model is outlined in Table 4.3. Animal live weights ranged between 360 and 640 kg, with reported live weight gains of between 0.6 and 1.8 kg/day. Corresponding intakes ranged between 5.34 and 14.13 kg DM/day, with reported CH\textsubscript{4} outputs of 62 to 192 g/head/day.

\subsection*{3.10.2 Nitrogen Model}

As with the enteric CH\textsubscript{4} model, a database of studies was created to validate the prediction equations for nitrogenous gases under the Australian Inventory methodology (2006). However, in contrast to the model for enteric CH\textsubscript{4}, the N model was validated in two stages; primarily due to lack of data directly linking detailed animal and ration information to measured emissions. Initially a database was created providing data about N transactions in the animal and used to validate the equations predicting N intake, retention and excretion (Table 4.4). A second database was created focussing on volatilisation of NH\textsubscript{3} and emission of N\textsubscript{2}O from feedlot manure (Table 4.7). Studies selected for the validation of the N excretion model were published since 2000, and focused on changing the N concentration in the diet and effects on N retention and excretion in feedlot cattle. Average live weight in these studies ranged from 495 to 627 kg, with average growth rates between 0.92 and 1.76 kg/day. Intakes ranged from 6.4 to 11.4 with corresponding N intakes of 94.3 to 240 g/day (Table 4.5). Limited ration information was provided in these studies, with many reporting only CP and ME content (Table 4.6). Where ration nutritive value was not provided it was calculated based on Givens and Moss (1990). The database of studies examining the emissions of N\textsubscript{2}O and volatilisation of NH\textsubscript{3} contained data from six published studies (Table 4.7). All studies were published since 2000, but varied in the emissions measurement method. The direct emission of N\textsubscript{2}O from feedlot manure pad (compared with compost piles) has received limited attention, therefore data available to validation this component of the model was limited to the results of Boadi \textit{et al.} (2004b). Further, many recent studies examining the volatilisation of NH\textsubscript{3} from feedlot systems have been conducted using atmospheric dispersion and micrometeorological methods, therefore detailed information on the composition of the manure pad, rations and animal performance was not consistently available. Full details of the studies utilised in the validation of the N excretion and gas components of the model are found in Chapter 4.
3.11 Comparison with Australian Feedlot Measurements

Full details of the approach used to compare measured and modelled emissions from Australian feedlots are outlined in Chapter 5. Briefly, campaigns information was collected from the feedlot operators in the form of bunk sheets, lot sheets, rations and daily feed amounts. This information was collected for a minimum of five days for each 14 day measurement campaign. Information collected included number of head (cattle) in each pen, animal sex, estimated weight on entry and at the report date, days on feed, cattle class, rations and amounts of feed offered. The collected data was utilised as input for the model outlined in the general methodology (Sections 3.6 to 3.10). Three of the described equations for estimation of enteric CH$_4$ (Ellis et al. 2007; Moe and Tyrrell 1979 and IPCC Tier II) were utilised in the application of the model to this data, based on the results of Chapter 4. The predicted emissions of CH$_4$, NH$_3$ and N$_2$O were compared statistically to measured values.

3.12 Statistical Analysis

For both the validation and model application exercises, linear regression (Genstat v. 11 VSN International) was used to examine relationships between measured and predicted emissions. Lin’s concordance (Genstat v. 11 VSN International) was used to measure how well the modelled data reproduces the original data set (Lin 1989; Lin 2000). Lin (1989) reports that this approach can detect non-reproducability in cases where the Pearson’s correlation coefficient or paired t-test may provide misleading results. The concordance coefficient is calculated using a simple Pearson’s correlation coefficient (reported in text as the correlation between the variables) and the slope and origin of the line, known as $C_b$. Full equations can be found in Appendix 9.1.

The factor $C_b$ indicates the how far the best-fit line deviates from the 45° line (indicator of accuracy). The further $C_b$ is from 1, the greater the deviation. The Person correlation coefficient ($p$) measures how much each observation deviates from the best-fit line. The concordance correlation coefficient evaluates the degree to which pairs fall on the 45° line, with measurements of accuracy ($C_b$) and precision ($p$). For a perfect 45° line $P_c = 1$.

In biological systems, a correlation of 0.6 can be considered acceptable, however in the comparison between the models the relative values can be considered equally important.

3.12.1 Interpretation
The results of a Lin’s concordance analysis can be interpreted utilising the values of concordance and correlation. High correlation and concordance indicate high model accuracy, with a strong relationship between measured and predicted values, and a slope close to the line of unity. A high correlation coupled with a low concordance indicates that while there is a strong relationship between measured and predicted values, there are likely to be systemic bias or errors in the model. Low correlations and concordance indicate poor model performance, but may be reflective of a small range of data.

3.13 Animal Behaviour

In order to examine the relationship between diurnal emission pattern and animal feeding patterns observations of animal feeding behaviour were conducted in conjunction with the final two measurement campaigns (winter 2007 and summer 2008) at each site. Emissions were measured using the methods described in Sections 3.2, full details of the behavioural observations can be found in Chapter 6.

3.13.1 Pen Selection

Pens of cattle were selected to be representative of the general feedlot population (based on weight, days on feed and breed composition). Pens also needed to be located close to power supply and in an area where the camera could be mounted to view the entire pen (Plate 3.3).

3.13.2 Observations

Video recording was used to make continuous observations of the pen from sunrise to sunset. Due to the lack of lights observations were not possible during the night. Observations were recorded at 10 minute intervals by stopping the video and counting the number of cattle at the feed bunk, water trough (if visible), standing and lying. Walking and “other” behaviours were determined by viewing 15 seconds either side of the 10 minute time point and counting the number walking and undertaking activities classified as other (grooming, interacting with another animal). Number ruminating was determined using the same interval, by randomly selecting 20 animals and counting the number ruminating. The proportion determined using this method was then multiplied to give total number ruminating.

Video recording was validated using manual observations during the winter measurement periods by a single operator using a modified octave approach. The 12 hour period from sunrise to sunset was divided into three hour blocks and manual observations undertaken by a single operator for one of these three hour periods over four consecutive days. Observations were made in the categories defined above.
3.13.3 Animal Details

During the first (winter) measurement campaign at the Southern Site the pen contained 247 head of mostly *B. Taurus* steers, weight 434-442 kg, minimum 28 days on feed (DOF). The ration fed contained 69.9% wheat, 12.5% silage, 3.5% grass hay, 5% liquid finisher, 8% cottonseed and 1.1% veg oil. 75% DM, 14.42% CP, 13.15 MJ ME/kg DM. At the Northern Site the pen contained 161 head of *B. Indicus* beef steers. Estimated weight of 477 kg, approximately 30 DOF. The ration fed comprises 11.5% silage, 0.5% recycled oil, 1.5% straw, 4.6% liquid supplement, 75.9% sorghum grain and 6% cottonseed. 69.15% DM, 13.50% CP, 12.76 MJ ME/kg DM.

During the second (summer) measurement campaign the Southern Site contained 223 head of *B. Taurus* cattle, between 480 and 570kg and 60 to 75 days on feed. The ration fed contained 27% wheat, 14% silage, 1.5% grass hay, 4.8% liquid finisher, 8% cotton seed, 1.1% vegetable oil, 3.1% molasses and 40.5% barley. 73% DM, 13.5% CP, 12.8 MJ ME/kg DM. The Northern Site pen contained 196 head of *B. Taurus* steers, 642 kg, and 116 days on feed. The ration fed contained 39% sorghum, 4.5 liquid supplement, 1% straw, 1% cotton hulls, 10% cotton seed, 5% silage, 39% barley and 0.5% recycled oil. 73% DM, 13.83% CP, 12.67 MJE ME/kg DM.

3.13.4 Statistical Analysis

Ten minute behavioural observations were averaged over 30 minute periods for comparison with the emissions measurements (15 minute values averaged over 30 minute periods). Simple correlation analysis was used to evaluate the relationship between emissions of CH$_4$ and NH$_3$, and animal feeding behaviour, as well as temperature (°C).
Plate 3.3 Video camera mounted in the Northern (A) and Southern (B) feedlots
Chapter 4. Evaluation of a Methodology for Estimation of Greenhouse Gas Emissions from Feedlot Systems

4.1 Introduction

The Australian agricultural sector, like many other industries, faces considerable pressure to reduce emissions of the greenhouse gases. However, in order to assess the usefulness of mitigation options, current emissions need to be accurately measured and estimated. Measurement of greenhouse gas emissions can be expensive and complicated (Ellis et al. 2007), however prediction equations can be used to estimate emissions. Empirical models currently used in national inventories were developed in the 1960’s and 1970’s for metabolism based predictions, and not for the greenhouse inventories. Therefore, the predictions may not adequately reflect the Australian feedlot environment, particularly as most are based on calorimetric measurements designed to measure the energy value of feeds (Denmead et al. 2000) and not as a measurement of the contribution of these gases towards global warming (Ellis et al. 2007). Additionally, empirical models often have variables which are not commonly measured and many have been reported to be inaccurate for conditions outside those (rations, animal characteristics etc) which were used in the development (Ellis et al. 2007; Mills et al. 2003; Wilkerson et al. 1995).

The National Inventory Methodology (AGO 2006) for estimation of greenhouse sources and sinks is used in Australia for carbon accounting under the Kyoto protocol, and would be the likely basis for estimation under a future carbon constrained economy, such as the recently proposed Carbon Farming Initiative (Australian Government 2010a). However, the accuracy of the Australian Inventory Methodology (AGO 2006) for feedlot systems has not been evaluated under commercial Australian conditions. Furthermore reducing total N loss is important for the sustainability of the feedlot industry (Erickson et al. 2000). However, as with enteric CH₄ emissions measurement nitrogenous gases in situ can be complicated and expensive. Although emissions of NH₃ do not currently have to be reported under the UNFCCC guidelines, monitoring potential and actual NH₃ emissions will become increasingly important, particularly in terms of contribution to indirect N₂O emissions.

4.1.1 Inventories and Emissions Reporting

According to international agreements under the United Nations Framework Convention on Climate Change (UNFCCC), national inventories of greenhouse gas emissions are prepared annually. These inventories are prepared according to published methodologies and allow
comparison between the emissions profiles and effectiveness of mitigation options from various countries. The National Inventory Methodology (AGO 2006) documentation provides methods for the estimation of emissions from a number of sources, including Agriculture, in Australia (described here as the National Inventory Methodology (AGO 2006). For reporting purposes, it is emissions associated with anthropogenic activity are primarily considered “Emissions of CH\textsubscript{4}, N\textsubscript{2}O, oxides of nitrogen (NO\textsubscript{x}), carbon monoxide (CO) and non-CH\textsubscript{4} volatile organic compounds are produced when living and dead biomass is eaten, consumed, decays or is burnt. These emissions are modified by human activities including cultivation, addition of fertilisers, deliberate burning and by the introduction of ruminant animals” (NGGIC 2006).

IPCC guidelines (IPCC 2006) allow the use of default emissions factors or country specific options. A country specific methodology should (under the guidelines) be used when one is available. When this is not available the standard IPCC Tier II methodology is used where enteric fermentation is considered a key source category and when more detailed livestock characterisation is available (e.g. intakes, live weights etc). Tier I (the simplest form of the methodology) can be used where livestock are not considered a key source category, or when insufficient data are available for the characterisation of livestock populations and feed types.

Countries in which livestock are considered a key source category, and those which are signatories to the Kyoto protocol, will generally have produced a country specific model for estimating emissions of CH\textsubscript{4} from enteric fermentation, emissions of N\textsubscript{2}O and CH\textsubscript{4} from manure (particularly for housed animals) and emissions from manure application as a fertiliser.

The importance of validating studies for the specific location and industry is highlighted by Mills et al. (2003). They observed the same linear models of enteric CH\textsubscript{4} emission to overestimate emissions when evaluated with North American data, but underestimate for United Kingdom data. Therefore, the suitability of a model for a given situation cannot be established by its use in another situation.

4.1.2 Research Question

This chapter aimed to address the basic research questions, initially using published data and results to validate the model.

1. Is the current Australian National Inventory Methodology for estimating greenhouse sources and sinks accurate in predicting emissions of CH\textsubscript{4}, N\textsubscript{2}O and NH\textsubscript{3} from feedlots?
2. Are equations based on energy or carbohydrate more accurate in estimating CH$_4$ emissions?

3. What is the impact of using a single emissions factor in the model to predict NH$_3$ and N$_2$O?

4. Does introducing equations developed based on beef cattle data improve accuracy of estimates compared to measured values of CH$_4$ emissions compared with the current approaches?

5. How sensitive is the model to changes in parameters such as animal intake, and to the gross energy and dietary digestibility of the ration?

4.2 Model Development/Structure

The full model is outlined in detail in the general methodology (Section 3.6 to 3.10). Briefly, the concept of the model was to utilise the current National Inventory Methodology (AGO 2006) for estimating emissions (CH$_4$, N$_2$O and NH$_3$) from feedlot cattle, with additional equations for the prediction of enteric CH$_4$ emissions (developed specifically for beef or feedlot type cattle). This section will discuss the equations selected for the model, the development of these original models, and the usefulness/applicability of the parameters which are used, in the context of a feedlot system.

An empirical modelling approach was used, as these types of models will allow data which is collected in a commercial environment (or for which reasonably reliable tabulated data exist) to be utilised as inputs. The alternative approach is a dynamic mechanistic model which attempts to represent rumen fermentation biochemistry and processes (Mills et al. 2003). Although this type of model can provide much more accurate estimates, they require more complex information and do not enable estimates based on limited dietary information (Mills et al. 2003). The complexity of these models makes them unsuitable for application in a commercial environment or for use in updating National Inventory guidelines, where an empirical model is a more practical approach.

4.2.1 IPCC Tier I and Tier II Equations

The IPCC recommend the use of a Tier II country specific model for ‘other’ (non dairy) cattle, including feedlot cattle (IPCC 2006). However, it is valid to compare emissions calculated using the standard Tier II methodology and the single estimate used in Tier I with values measured for feedlot cattle for two reasons. Firstly, the Tier II approach is utilised in countries where agriculture is not considered a key source category, many of which have large
feedlot industries Secondly, feedlot production is increasing in developing countries, where there is likely to be insufficient information (animal population, production system characteristics and feed base details) to utilise a Tier II approach or more detailed country specific approach. Increasing understanding of the accuracy of these models for feedlot production will be beneficial in the ability to compare industries.

The Tier I methodology is based on cattle type and geographical region. Oceania is characterised as having “Commercialised dairy sector based on grazing. Separate beef cow herd, primarily grazing rangelands of widely varying quality with growing amount of feedlot feeding with grains”(IPCC 1996). Under the IPCC Tier I methodology beef cattle are estimated at producing 60 kg CH$_4$/head/year, equating to 164 g CH$_4$/head/day, assuming a calendar year.

Tier II is the simplest step of the methodology which allows differences in animal production systems to be accounted for. The IPCC standard Tier II methodology estimates gross energy intake based on dry matter intake and a standard value for the gross energy content of feeds. Gross energy intake is multiplied by a set factor for CH$_4$ as a percentage of GEi, with different factors for different production systems. For a feedlot system, 3 ±1 % of GEi is assumed to be emitted as CH$_4$, compared with 6.5 ±1 % for dairy and grazing beef systems. A criticism of this method is that a single factor is not representative of the proportion of gross energy converted to CH$_4$ for all feed types and production systems. Measurements from feedlot systems and from beef cattle fed feedlot type rations suggest emissions ranging from 62 g/head/day to190 g/head/day, or between 0.7 and 7.3 % GEi (Beauchemin and McGinn 2005; Boadi et al. 2004b). Even with a single diet type considerable variability has been observed; Boadi et al. (2004b) measured CH$_4$ production, as a percentage of GE, as between 0.9 and 6.9% for a high forage feedlot diet, and from 0.7 to 4.9% for a high grain diet. In this evaluation a figure of 3% of gross energy intake, and 18.4 MJ/ kgDM of gross energy are used, in accordance with the IPCC standard for feedlot cattle.

4.2.2 Moe and Tyrell (1979) Equation

This equation (Equation 3.2) is used as the Australian country specific model for beef feedlot cattle (Tier II, country specific), as it is considered to better represent the type of diets fed to feedlot cattle (higher grain/ concentrate) compared with a methodology based on energy and intake. Moe and Tyrell (1979) developed this equation based on studies with dairy cows, with the original experiments designed to examine the effects of dietary components (protein, concentrate proportion, different grain types, and form of grain) on energy utilisation in dairy
cattle diets. In the studies utilised in the development of the Moe and Tyrrell (1979) model all animals were Holstein dairy cows, with live weight ranging from 369 to 893 kg (average 617 kg), and corresponding intakes 2.7 to 22.9 kg DM/day (average 12.1 kg DM/day). The range of diets included corn and barley grain, oats, beet pulp, dried brewers grains and dried distillers grains.

Moe and Tyrrell (1979) report that the soluble residue (calculated as neutral detergent soluble minus crude protein and ether extract) utilised in the equation should represent most soluble and readily fermentable carbohydrates, consisting primarily of starch for the mixed rations evaluated. The other parameters of interest are cellulose (ADF- lignin and silica) and hemicellulose (NDF-ADF). Which were determined to have the most significant relationships with CH\textsubscript{4} production (other dietary parameters (crude protein, ether extract and lignin were determined to be non significant). Intake of soluble residue ranged from 0.95 to 9.66 kg/day (average 5.06 kg/day), cellulose from 0.41 to 3.90 kg/day and hemicellulose from 0.47 to 4.41 kg/day. CH\textsubscript{4} production as a percentage of GEi averaged 6.3% (1.6 to 9.9%).

The equation of Moe and Tyrrell (1979) is reported to have the advantage of utilising a wider range of rations than many other studies (e.g. Blaxter and Clapperton 1965). The basis of the equation on carbohydrate fractions additionally means that it is able to distinguish differences in emissions associated with changes in diet type. Variation in emissions from diets with different digestibility, diets increasing in grain content, and from different basal grains are commonly reported (Beauchemin and McGinn 2005; Boadi et al. 2004b); using an equation with a set value for GE or digestibility does not allow these differences to be distinguished.

A detailed review of a number of statistical models by Wilkerson et al. (1995) suggested that Moe and Tyrrell (1979) should be used for dairy cattle, however, as noted by Mills et al. (2003) this was developed based on North American data and therefore may not represent other feed types or animal genetics. Mills et al. (2003) observed this model to be the best estimate for both North American and United Kingdom data, and suggests its use for dairy cattle, making no recommendation for beef cattle. A disadvantage of more specific dietary approaches; is that parameters (such a cellulose, hemicellulose) are not commonly measured, in both commercial feed tests and for research purposes (Mills et al. 2003). Additionally, values for these feeds may vary considerably both between and within feed types and with measurement methods, which may increase the error associated with using a set value for emissions.
4.2.3 Blaxter and Clapperton (1965) Equation

This equation (Equation 3.3) uses intake of dry matter digestibility and relative intake to estimate \( Y_m \) (a CH\(_4\) conversion factor). This is then multiplied by the intake of gross energy to determine daily CH\(_4\) emissions (g/head/day). This equation was developed primarily based on sheep (16 sources, cattle contributing only five sources) which are likely to vary significantly from cattle, particularly those with high growth rates, (or high milk production) in terms of CH\(_4\) emissions. The data set included 55 diets. CH\(_4\) production as a percentage of dietary energy ranged from 6.2 to 10.8%. Primarily the diets were roughage based (29 diets), while the mixed diets (11 diets) also contained some hay.

A downfall of this equation as used in the current National Inventory Methodology (AGO 2006) (Tier II) country specific model for dairy and grazing beef cattle) is that a set value is given for both digestibility (80%) and feed gross energy content (18.4 MJ/kg DM). Digestibility and energy content varies not only between feeds (e.g. forages lower in both digestibility and energy than grains) but within single feeds. This limits the ability of this methodology to distinguish between differing diet composition.

4.2.4 Ellis et al. (2007) Equation

Ellis et al. (2007) developed a range of equations for beef and dairy cattle, using dietary variables. The source data utilised for this equation included both beef and dairy cattle, entirely of Canadian and North American origin, DMI ranged from 3.8 to 18.4 kg DM/day, live weights between 268 and 707 kg. Mean forage percentage in the beef rations was 79% (9 to 100%) and in the dairy rations 69% (28 to 100%). At the lower end this is similar to feedlot type diets, however the number of high grain diets evaluated was not reported. The best performing equation for beef cattle (Equation 3.4) uses dietary NDF, lignin and metabolisable energy intake (MEi), similar in principal to Moe and Tyrell (1979). The primary advantage of this equation is that a database focussed specifically on beef cattle was used in the development, which can be assumed to increase accuracy.

However Yan et al. (2009) determined high predictive error when validating a range of prediction equations developed by Ellis et al. (2007), with the lowest \( R^2 \) values (in their study) recorded for these set of equations. They attributed this to the use of both calorimetric (20 sources), SF\(_6\) (8 sources) and mass balance (one source) methods in the measurements used for the producing the equations. This will be discussed in more detail in the general discussion, however observations (Boadi et al. 2004b; Grainger et al. 2007) suggest that there are significant differences in CH\(_4\) emissions measured using chambers and SF\(_6\) techniques, even when feed type and intake is constant.
4.2.5 Ellis et al. (2009) Equation

Following on from the equations published by Ellis et al. (2007), Ellis et al. (2009) aimed to improve on the models developed by utilising more detailed animal production data (Ellis et al. 2009). In contrast to Ellis et al. (2007), Ellis et al. (2009) utilised only beef cattle studies. Live weights ranged from 243 to 497 kg, with corresponding intakes of 4.47 to 10.8 kg DM/day. CH₄ % of GEI varied between 3.90 and 7.09%. Dietary forage % averaged 53.7% (9 to 75%).

This equation has the benefit of including a parameter indicating the effect of a lipid in the diet, which is common practice in many Australian feedlots (often in the form of recycled oil). In theory, an equation which can manage the reduction associated with feeding a lipid should improve the accuracy of emission predictions. However, the disadvantage of this equation, as with the other equations based on fibre/CHO, is the use of parameters which are not commonly measured under commercial conditions (although tabulated values are available). Hemicellulose is one of the parameters used in the final equation developed, however, this is not commonly reported, and therefore many of the values used in the development of the original equation (Ellis et al. 2009) were calculated or based on North American averages. The measurement method for hemicellulose, and differences in feed can impact on the amount of hemicellulose determined, which may therefore result in inaccuracies in estimates using this method.

4.2.6 Manure CH₄ Estimation

Within the model, CH₄ from manure is estimated based on the standard IPCC methodology, using country specific emission factors (Equations 3.6 and 3.7). The model for manure CH₄ emissions was not validated using the same process as enteric CH₄.

4.2.7 Nitrogen Excretion and Emissions

The N model can be considered in two parts, firstly, estimating the partition of feed N into growth, urine and faeces based on Freer et al. (2007) and SCA (1990). Secondly set emissions factors are used to estimate emissions of N₂O and NH₃ from dry lot management systems (Equations 3.11 to 3.14). In this study, only direct emissions of N₂O have been considered, excluding N₂O from manure spreading, and indirect emissions from deposition of NH₃.

The National Inventory Methodology (AGO 2006) for estimating N₂O emissions and volatile NH₃ (although this is not required for reporting) uses a country specific method, with IPCC (IPCC 2006) default emission factors. The primary difference in the approach of the IPCC Tier II method for estimating N transactions in the animal is that the National Inventory
Methodology (AGO 2006) is based on the equations developed by SCA (1990), while the IPCC model is based on the NRC (1996). However, the underlying approach of both methodologies is similar, with a mass balance approach of N intake - N Retention = N excretion utilised. The SCA models partition N excretion into faecal and urinary N, while the IPCC methodology uses a single figure for N excretion.

4.3 Methods

The enteric CH₄ component of the model was evaluated using the results of published studies. The procedure followed will be discussed in more detail in the following sections, however briefly, a database was developed comprising studies which focussed specifically on enteric CH₄ emissions from beef feedlot cattle. Where data were not available specifically for beef feedlot cattle, the database was extended to include beef cattle with similar characteristics to feedlot cattle (weight, age, ration types, intakes).

The N model was validated in two stages. Initially a database was created providing data about N transactions in the animal and used to validate the equations predicting N intake, retention and excretion (in manure and faeces). A second database was created focussing on volatilisation of NH₃ and emission of N₂O from feedlot manure.

4.3.1 Enteric CH₄ Validation

Five published studies were selected for the validation of the enteric CH₄ model (Table 4.1). These studies all investigated the effect of changing dietary parameters (chemical composition, intake) on enteric CH₄ emissions. One study Boadi et al. (2004b) included data on emissions of CH₄ and N₂O from manure. Studies in the database were published since 2000 and focused on manipulating dietary factors on CH₄ emissions from feedlot cattle. The primary factor under investigation was the energy density of the diet, utilizing high and low forage proportion, with the exception of Hegarty et al. (2007) who utilized the same ration at different intake levels.

<table>
<thead>
<tr>
<th>Study</th>
<th>Primary Comparison</th>
</tr>
</thead>
<tbody>
<tr>
<td>Boadi et al. (2004b)</td>
<td>High Forage vs. Low Forage</td>
</tr>
<tr>
<td>Beauchemin and McGinn (2006)</td>
<td>Level of Grain</td>
</tr>
<tr>
<td>Hegarty et al. (2007)</td>
<td>Intake</td>
</tr>
</tbody>
</table>

Table 4.1 Studies selected for evaluation of the standard model, comparisons of the model and physiological response tested in the model
The studies (Table 4.1) were primarily conducted in the Northern hemisphere, with the exception of Hegarty et al. (2007). There are limited studies investigating changes in dietary parameters on CH$_4$ emissions of feedlot cattle under Australian conditions, therefore Northern hemisphere based studies were also included. Rations used in the CH$_4$ validation studies ranged in forage proportion from 0.09 to 0.7. ME values ranged between 8.9 and 12.1 MJ/kg DM and CP from 120 to 150 g/kg DM. Forage proportion ranged between 9 and 70% of the rations. Animal live weights ranged between 360 and 640kg, with reported live weight grains of between 0.6 and 1.8 kg/day. Corresponding intakes ranged between 5.3 and 14.1 kg DM/day, with reported CH$_4$ outputs of 62 to 192 g/head/day (Table 4.3).
Table 4.2 Ration characteristics of the studies utilised for validation of the enteric CH\textsubscript{4} model

<table>
<thead>
<tr>
<th>Study</th>
<th>Ration</th>
<th>DM %</th>
<th>GE MJ/kg</th>
<th>ME MJ/kg</th>
<th>CP g/kg DM</th>
<th>NDF g/kg DM</th>
<th>ADF g/kg DM</th>
<th>Forage Proportion*</th>
</tr>
</thead>
<tbody>
<tr>
<td>Beauchemin and McGinn (2004)</td>
<td>Corn Backgrounding</td>
<td>50.8</td>
<td>18.3</td>
<td>8.9*</td>
<td>146</td>
<td>350</td>
<td>150</td>
<td>0.70</td>
</tr>
<tr>
<td></td>
<td>Barley Backgrounding</td>
<td>42.2</td>
<td>18.6</td>
<td>9.4#</td>
<td>137</td>
<td>363</td>
<td>163</td>
<td>0.70</td>
</tr>
<tr>
<td></td>
<td>Corn Finishing</td>
<td>78.2</td>
<td>18.2</td>
<td>10.9#</td>
<td>134</td>
<td>127</td>
<td>35</td>
<td>0.09</td>
</tr>
<tr>
<td></td>
<td>Barley Finishing</td>
<td>78.2</td>
<td>18.1</td>
<td>10.6#</td>
<td>151</td>
<td>204</td>
<td>60</td>
<td>0.09</td>
</tr>
<tr>
<td>Boadi et al. (2004b)</td>
<td>Low F:G</td>
<td>73.9</td>
<td>17.8</td>
<td>11.5*</td>
<td>132</td>
<td>196</td>
<td>102</td>
<td>0.11</td>
</tr>
<tr>
<td></td>
<td>High F:G</td>
<td>55.6</td>
<td>18.8</td>
<td>10.6*</td>
<td>120</td>
<td>291</td>
<td>185</td>
<td>0.41</td>
</tr>
<tr>
<td>Beauchemin and McGinn (2006)</td>
<td>Low Grain</td>
<td>50.9</td>
<td>18.7</td>
<td>9.6#</td>
<td>168</td>
<td>348</td>
<td>218</td>
<td>0.70</td>
</tr>
<tr>
<td></td>
<td>High Grain</td>
<td>66.8</td>
<td>18.7</td>
<td>10.8#</td>
<td>160</td>
<td>203</td>
<td>116</td>
<td>0.30</td>
</tr>
<tr>
<td>Hegarty et al. (2007)</td>
<td></td>
<td>88.7</td>
<td>17.8</td>
<td>12.1</td>
<td>159</td>
<td>178</td>
<td>79</td>
<td>0.10</td>
</tr>
<tr>
<td>Lovett et al. (2003)^*</td>
<td>1 (65:35 F:C)</td>
<td>45.8</td>
<td>19.5</td>
<td>11.0*</td>
<td>158</td>
<td>473</td>
<td>272</td>
<td>0.60</td>
</tr>
<tr>
<td></td>
<td>2 (40:60)</td>
<td>60.2</td>
<td>18.8</td>
<td>10.5*</td>
<td>157</td>
<td>371</td>
<td>197</td>
<td>0.38</td>
</tr>
<tr>
<td></td>
<td>3 (10:90)</td>
<td>77.9</td>
<td>17.9</td>
<td>10.7*</td>
<td>155</td>
<td>244</td>
<td>104</td>
<td>0.12</td>
</tr>
</tbody>
</table>

* study included an investigation of the effects of inclusion of coconut oil into the above rations, however for simplicity only the results of alteration to the forage concentration of the diet were utilized. ^ offered forage proportion, * reported as GE kcal/kg DM ME calculated using digestibility of GE and a factor of 4.184 to convert kcal to MJ. ^ reported as GE KJ/g ME calculated.
<table>
<thead>
<tr>
<th>Study</th>
<th>Ration/ Parameter</th>
<th>Enteric CH₄</th>
<th>Initial Weight</th>
<th>Final Weight</th>
<th>ADG'</th>
<th>Intake</th>
<th>Days on Feed</th>
<th>FCE#</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>g/head/day</td>
<td>Kg</td>
<td>kg</td>
<td>kg/day</td>
<td>kgDM/day</td>
<td>Feed</td>
<td></td>
</tr>
<tr>
<td>Beauchemin and McGinn (2005)</td>
<td>Corn Backgrounding</td>
<td>170 ± 22.3</td>
<td>330 ± 19.8</td>
<td>380 ± 25.5</td>
<td>1.2 ± 0.3</td>
<td>6.9 ± 1.60</td>
<td>42</td>
<td>0.17</td>
</tr>
<tr>
<td></td>
<td>Barley Background</td>
<td>130 ± 22.3</td>
<td>325 ± 19.8</td>
<td>364 ± 25.5</td>
<td>0.9 ± 0.3</td>
<td>5.3 ± 1.60</td>
<td>42</td>
<td>0.17</td>
</tr>
<tr>
<td></td>
<td>Corn Finishing</td>
<td>62 ± 22.3</td>
<td>426 ± 19.8</td>
<td>452 ± 25.5</td>
<td>0.8 ± 0.3</td>
<td>6.8 ± 1.60</td>
<td>32</td>
<td>0.11</td>
</tr>
<tr>
<td></td>
<td>Barley Finishing</td>
<td>80 ± 22.3</td>
<td>412 ± 19.8</td>
<td>442 ± 25.5</td>
<td>0.9 ± 0.3</td>
<td>6.2 ± 1.60</td>
<td>32</td>
<td>0.11</td>
</tr>
<tr>
<td>Boadi et al. (2004b)</td>
<td>Low F:G</td>
<td>126 ± 47.0*</td>
<td>300 ± 32.0</td>
<td>569 ± 37.0</td>
<td>1.8 ± 0.2</td>
<td>11.7 ± 3.00</td>
<td>150</td>
<td>0.15</td>
</tr>
<tr>
<td></td>
<td>High F:G</td>
<td>90 ± 47.0*</td>
<td>302 ± 32.0</td>
<td>548 ± 37.0</td>
<td>1.5 ± 0.2</td>
<td>10.0 ± 3.00</td>
<td>164</td>
<td>0.15</td>
</tr>
<tr>
<td>Beauchemin and McGinn (2006)</td>
<td>Low Grain</td>
<td>132 ± 39.6</td>
<td>328 ± 28.0</td>
<td>430 ± 29.0</td>
<td>1.7*</td>
<td>6.2 ± 1.20</td>
<td>60</td>
<td>0.27</td>
</tr>
<tr>
<td></td>
<td>High Grain</td>
<td>151 ± 39.6</td>
<td>328 ± 28.0</td>
<td>430 ± 29.0</td>
<td>1.7*</td>
<td>7.5 ± 1.20</td>
<td>60</td>
<td>0.22</td>
</tr>
<tr>
<td></td>
<td>Ad Lib</td>
<td>169 ± 39.6</td>
<td>328 ± 28.0</td>
<td>430 ± 29.0</td>
<td>1.7*</td>
<td>8.3 ± 1.20</td>
<td>60</td>
<td>0.20</td>
</tr>
<tr>
<td></td>
<td>Restricted</td>
<td>114 ± 39.6</td>
<td>328 ± 28.0</td>
<td>430 ± 29.0</td>
<td>1.7*</td>
<td>5.4 ± 1.20</td>
<td>60</td>
<td>0.32</td>
</tr>
<tr>
<td>Hegarty et al. (2007)</td>
<td>Low Intake</td>
<td>142 ± 52.2</td>
<td>541 ± 44.4</td>
<td>642 ± 52.1</td>
<td>1.1 ± 0.3</td>
<td>8.4 ± 2.60</td>
<td>15</td>
<td>0.13</td>
</tr>
<tr>
<td></td>
<td>High Intake</td>
<td>190 ± 52.2</td>
<td>541 ± 44.4</td>
<td>642 ± 52.1</td>
<td>1.2 ± 0.3</td>
<td>14.1 ± 2.60</td>
<td>15</td>
<td>0.08</td>
</tr>
<tr>
<td>Lovett et al. (2003)</td>
<td>1 (65:35 F:C)</td>
<td>148 ± 120.0*</td>
<td>463 ± 12.0</td>
<td>512 ± 30.0</td>
<td>0.6 ± 0.6</td>
<td>6.9 ± 2.04</td>
<td>77</td>
<td>0.09</td>
</tr>
<tr>
<td></td>
<td>2 (40:60)</td>
<td>193 ± 120.0*</td>
<td>462 ± 12.0</td>
<td>525 ± 30.0</td>
<td>0.8 ± 0.6</td>
<td>8.4 ± 2.04</td>
<td>77</td>
<td>0.09</td>
</tr>
<tr>
<td></td>
<td>3 (10:90)</td>
<td>122 ± 120.0*</td>
<td>462 ± 12.0</td>
<td>543 ± 30.0</td>
<td>1.1 ± 0.6</td>
<td>8.2 ± 2.04</td>
<td>77</td>
<td>0.13</td>
</tr>
</tbody>
</table>

*ADG (average daily gain) and days on feed not reported, 60 days assumed based on class of cattle, ADG calculated based on weight gain and days on feed, ^reported in L/head/day, #feed conversion efficiency calculated as kg of gain/ kg feed intake, ‘average daily gain
4.3.2 Nitrogen Transactions

Studies selected for the validation of the N transaction model (Table 4.4) were published since 2000, and focussed on changing the N concentration in the diet and effects on N retention and excretion in feedlot cattle. Average live weight (Table 4.5) in these studies ranged from 495 to 627 kg, with average growth rates between 0.9 and 1.8 kg/day. Intakes ranged from 6.4 to 11.4 kg DM with corresponding N intakes of 94 to 240 g/day. Limited ration information (Table 4.6) was provided in these studies, with many reporting only CP and ME content. Where ration nutritive value was not provided it was calculated based on ration composition and tabulated values (Givens and Moss 1990).

**Table 4.4** Studies selected for validation of the model for nitrogen transactions in the animal and primary parameter investigated

<table>
<thead>
<tr>
<th>Source</th>
<th>Primary Comparison</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cole et al. 2003</td>
<td>Crude Protein concentration</td>
</tr>
<tr>
<td>Adams et al. 2004</td>
<td>Digestibility of protein</td>
</tr>
<tr>
<td>Cole et al. 2006</td>
<td>Crude protein concentration</td>
</tr>
<tr>
<td>Archibeque et al. 2007</td>
<td>Crude protein concentration</td>
</tr>
<tr>
<td>Vasconcelos et al. 2009</td>
<td>Crude protein and urea level</td>
</tr>
</tbody>
</table>

The database of studies examining the emissions of N\(_2\O\) and volatilisation of NH\(_3\) contained data from six published studies (Table 4.7). All studies were published since 2000, but varied in the emissions measurement method. The direct emission of N\(_2\O\) from feedlot manure pad (compared with compost piles) has received limited attention, therefore data available to validation this component of the model was limited to the results of Boadi *et al.* (2004b).

Further, many recent studies examining the volatilisation of NH\(_3\) from feedlot systems have been conducted using atmospheric dispersion and micrometeorological methods, therefore detailed information on the composition of the manure pad, rations and animal performance was not consistently available. For these studies, an estimate of N intake based on the data provided in the publication was used in the model to predict NH\(_3\) volatilisation (Table 4.8).
Table 4.5 Animal production data used in the validation of the model for nitrogen transactions (mean and standard deviation)

<table>
<thead>
<tr>
<th>Study</th>
<th>Trt.</th>
<th>Initial Weight</th>
<th>Final Weight</th>
<th>ADG</th>
<th>Days on Feed</th>
<th>N Intake</th>
<th>N Retained</th>
<th>Faecal N</th>
<th>Urine N</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>kg</td>
<td>kg</td>
<td>kg/day</td>
<td></td>
<td>g/day</td>
<td>g/day</td>
<td>g/day</td>
<td>g/day</td>
</tr>
<tr>
<td>Cole et al. (2003)^</td>
<td>12%</td>
<td>404 ± 23.4</td>
<td>495^</td>
<td>1.6 ± 0.36</td>
<td>56</td>
<td>198 ± 29.1</td>
<td>22 ± 7.38</td>
<td>29 ± 11.6</td>
<td>147 ± 25.9</td>
</tr>
<tr>
<td></td>
<td>14%</td>
<td>404 ± 23.4</td>
<td>517^</td>
<td>2.0 ± 0.36</td>
<td>56</td>
<td>240 ± 29.1</td>
<td>27 ± 7.38</td>
<td>31 ± 11.6</td>
<td>182 ± 25.9</td>
</tr>
<tr>
<td>Adams et al. (2004)^</td>
<td>T1. Con</td>
<td>324 ± 9.8</td>
<td>613 ± 78.4</td>
<td>1.6 ± 0.39</td>
<td>180</td>
<td>211 ± 5.9</td>
<td>26 ± 0.98</td>
<td>91 ± 15.7*</td>
<td></td>
</tr>
<tr>
<td></td>
<td>T1. Bran</td>
<td>325 ± 9.8</td>
<td>591 ± 78.4</td>
<td>1.5 ± 0.39</td>
<td>180</td>
<td>221 ± 5.9</td>
<td>24 ± 0.98</td>
<td>93 ± 15.7*</td>
<td></td>
</tr>
<tr>
<td></td>
<td>T2. Con</td>
<td>376 ± 9.8</td>
<td>574 ± 39.2</td>
<td>1.5 ± 0.29</td>
<td>132</td>
<td>236 ± 9.8</td>
<td>27 ± 1.37</td>
<td>79 ± 12.7*</td>
<td></td>
</tr>
<tr>
<td></td>
<td>T2. Bran</td>
<td>376 ± 9.8</td>
<td>569 ± 39.2</td>
<td>1.5 ± 0.29</td>
<td>132</td>
<td>239 ± 9.8</td>
<td>27 ± 1.37</td>
<td>91 ± 12.7*</td>
<td></td>
</tr>
<tr>
<td></td>
<td>T3. Con</td>
<td>335 ± 9.8</td>
<td>627 ± 22.5</td>
<td>1.8 ± 0.10</td>
<td>166</td>
<td>291 ± 3.9</td>
<td>36 ± 0.59</td>
<td>98 ± 7.8*</td>
<td></td>
</tr>
<tr>
<td></td>
<td>T3. Bran</td>
<td>337 ± 9.8</td>
<td>616 ± 22.5</td>
<td>1.7 ± 0.10</td>
<td>166</td>
<td>222 ± 3.9</td>
<td>34 ± 0.59</td>
<td>138 ± 7.8*</td>
<td></td>
</tr>
<tr>
<td>Cole et al. (2006)^</td>
<td>11.5%</td>
<td>315 ± 12.3</td>
<td>561 ± 9.4</td>
<td>1.4 ± 0.03</td>
<td>180</td>
<td>135 ± 8.0</td>
<td>23 ± 1.16</td>
<td>41 ± 8.4</td>
<td>71 ± 8.1</td>
</tr>
<tr>
<td></td>
<td>13%</td>
<td>315 ± 12.3</td>
<td>570 ± 9.4</td>
<td>1.5 ± 0.03</td>
<td>180</td>
<td>167 ± 8.0</td>
<td>23 ± 1.16</td>
<td>58 ± 8.4</td>
<td>86 ± 8.1</td>
</tr>
<tr>
<td>Archibeque et al. (2007)^</td>
<td>Low</td>
<td>305 ± 57.6</td>
<td>507 ± 127.2</td>
<td>1.0 ± 0.84</td>
<td>184</td>
<td>94 ± 51.6</td>
<td>35 ± 48</td>
<td>37 ± 18</td>
<td>24 ± 37.2</td>
</tr>
<tr>
<td></td>
<td>Medium</td>
<td>307 ± 57.6</td>
<td>590 ± 127.2</td>
<td>1.5 ± 0.84</td>
<td>184</td>
<td>131 ± 51.6</td>
<td>50 ± 48</td>
<td>43 ± 18</td>
<td>39 ± 37.2</td>
</tr>
<tr>
<td></td>
<td>High</td>
<td>304 ± 57.6</td>
<td>588 ± 127.2</td>
<td>1.5 ± 0.84</td>
<td>184</td>
<td>143 ± 51.6</td>
<td>40 ± 48</td>
<td>43 ± 18</td>
<td>60 ± 37.2</td>
</tr>
<tr>
<td>Vasconcelos et al. (2009)^</td>
<td>T1. 11.5</td>
<td>315 ± 16.6</td>
<td>507 ± 67.5</td>
<td>1.24’</td>
<td>154</td>
<td>112 ± 16.6</td>
<td>39 ± 6.8</td>
<td>35 ± 31.4</td>
<td>38 ± 79.2</td>
</tr>
<tr>
<td></td>
<td>T1. 13</td>
<td>315 ± 16.6</td>
<td>507 ± 67.5</td>
<td>1.24’</td>
<td>154</td>
<td>131 ± 16.6</td>
<td>38 ± 6.8</td>
<td>38 ± 31.4</td>
<td>55 ± 79.2</td>
</tr>
<tr>
<td></td>
<td>T1. 14.5</td>
<td>315 ± 16.6</td>
<td>507 ± 67.5</td>
<td>1.24’</td>
<td>154</td>
<td>142 ± 16.6</td>
<td>42 ± 6.8</td>
<td>40 ± 31.4</td>
<td>60 ± 79.2</td>
</tr>
<tr>
<td></td>
<td>T2. 11.5</td>
<td>353 ± 43.6</td>
<td>499 ± 67.5</td>
<td>0.92’</td>
<td>159</td>
<td>112 ± 16.6</td>
<td>39 ± 6.8</td>
<td>35 ± 31.4</td>
<td>38 ± 79.2</td>
</tr>
<tr>
<td></td>
<td>T2. 13</td>
<td>353 ± 43.6</td>
<td>499 ± 67.5</td>
<td>0.92’</td>
<td>159</td>
<td>131 ± 16.6</td>
<td>37 ± 6.8</td>
<td>39 ± 31.4</td>
<td>55 ± 79.2</td>
</tr>
<tr>
<td></td>
<td>T2. 14.5</td>
<td>353 ± 43.6</td>
<td>499 ± 67.5</td>
<td>0.92’</td>
<td>159</td>
<td>142 ± 16.6</td>
<td>42 ± 6.8</td>
<td>40 ± 31.4</td>
<td>60 ± 79.2</td>
</tr>
</tbody>
</table>

* total manure N, ^ SEM reported S.D shown are calculated values, ^ calculated from daily gain and days on feed; blank areas represent data which was not reported and could not be calculated based on published values, ‘ ADG (average daily gain) not reported, calculated from total gain and days on feed
Table 4.6 Ration details of the studies used in the validation of the model for nitrogen transactions

<table>
<thead>
<tr>
<th>Study</th>
<th>Treatment</th>
<th>ME</th>
<th>CP</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>MJ/kg DM</td>
<td>g/kg DM</td>
</tr>
<tr>
<td>Cole et al. (2003)</td>
<td>12% CP</td>
<td>12.9</td>
<td>121</td>
</tr>
<tr>
<td></td>
<td>14% CP</td>
<td>12.8</td>
<td>141</td>
</tr>
<tr>
<td></td>
<td>T1. Bran</td>
<td>11.0</td>
<td>131</td>
</tr>
<tr>
<td></td>
<td>T2. Control</td>
<td>12.2</td>
<td>138</td>
</tr>
<tr>
<td></td>
<td>T2. Bran</td>
<td>10.9</td>
<td>138</td>
</tr>
<tr>
<td></td>
<td>T3. Control</td>
<td>12.0</td>
<td>127</td>
</tr>
<tr>
<td></td>
<td>T3. Bran</td>
<td>11.1</td>
<td>128</td>
</tr>
<tr>
<td>Cole et al. (2006)</td>
<td>11.5% CP</td>
<td>11.9</td>
<td>111</td>
</tr>
<tr>
<td></td>
<td>13% CP</td>
<td>11.7</td>
<td>131</td>
</tr>
<tr>
<td></td>
<td>Low</td>
<td>12.9</td>
<td>91</td>
</tr>
<tr>
<td>Archibeque et al. (2007))</td>
<td>Medium</td>
<td>12.9</td>
<td>117</td>
</tr>
<tr>
<td></td>
<td>High</td>
<td>12.9</td>
<td>139</td>
</tr>
<tr>
<td>Vasconcelos et al. (2009)</td>
<td>T1. 11.5</td>
<td>11.8</td>
<td>104</td>
</tr>
<tr>
<td></td>
<td>T1. 13</td>
<td>11.9</td>
<td>104</td>
</tr>
<tr>
<td></td>
<td>T1. 14.5</td>
<td>11.6</td>
<td>115</td>
</tr>
<tr>
<td></td>
<td>T2. 11.5</td>
<td>11.7</td>
<td>115</td>
</tr>
<tr>
<td></td>
<td>T2. 13</td>
<td>11.5</td>
<td>128</td>
</tr>
<tr>
<td></td>
<td>T2. 14.5</td>
<td>11.6</td>
<td>128</td>
</tr>
</tbody>
</table>

The studies used in the validation of the N emissions component of the model utilised both chamber (in vitro) and micrometeorologica l methods (Table 4.7). For the chamber studies N intake ranged from 94 to 150 g/day with between 673 and 1277 g of N applied to the chamber. For the micrometeorological studies N intake ranged from 160-205 g/head/day.
Table 4.7 Studies selected for the validation of the model for nitrogen gas, major gas measured and measurement approach

<table>
<thead>
<tr>
<th>Source</th>
<th>Gas Measured</th>
<th>Measurement Method</th>
</tr>
</thead>
<tbody>
<tr>
<td>Boadi et al. (2004b)</td>
<td>N₂O</td>
<td>Chambers</td>
</tr>
<tr>
<td>Cole et al. (2005)</td>
<td>NH₃</td>
<td>Chambers</td>
</tr>
<tr>
<td>McGinn et al. (2007)</td>
<td>NH₃</td>
<td>Open-path Laser</td>
</tr>
<tr>
<td>Todd et al. (2008)</td>
<td>NH₃</td>
<td>Acid Washing Samplers</td>
</tr>
<tr>
<td>Van Haarlem et al. (2008)</td>
<td>NH₃</td>
<td>Open-path Laser</td>
</tr>
</tbody>
</table>
Table 4.8 Treatments, N intakes and excretion and measured N emissions (N\textsubscript{2}O and NH\textsubscript{3}) used in the validation of the N gas model

<table>
<thead>
<tr>
<th>Treatment</th>
<th>N Intake g/head/day</th>
<th>N Added to Chamber g</th>
<th>Faecal N Added g</th>
<th>Urine N Added g</th>
<th>NH\textsubscript{3}- Loss g/head/day</th>
<th>N\textsubscript{2}O- Loss g/head/day</th>
</tr>
</thead>
<tbody>
<tr>
<td>Boadi et al. (2004b)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Low forage: grain</td>
<td>98.1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.16</td>
</tr>
<tr>
<td>High forage: Grain</td>
<td>76.2</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.15</td>
</tr>
<tr>
<td>Cole et al. (2005)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>11.5% CP</td>
<td>129.2</td>
<td>840</td>
<td>382</td>
<td>459</td>
<td>17.55</td>
<td></td>
</tr>
<tr>
<td>13% CP</td>
<td>143.3</td>
<td>1155</td>
<td>434</td>
<td>721</td>
<td>35.09</td>
<td></td>
</tr>
<tr>
<td>14.5% CP</td>
<td>150.4</td>
<td>1113</td>
<td>466</td>
<td>647</td>
<td>29.41</td>
<td></td>
</tr>
<tr>
<td>McGinn et al. (2007)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Whole Feedlot</td>
<td>164</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>140</td>
</tr>
<tr>
<td>Todd et al. (2008)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Summer 2002</td>
<td>162</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>117</td>
</tr>
<tr>
<td>Winter 2003</td>
<td>169</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>51</td>
</tr>
<tr>
<td>Summer 2003</td>
<td>186</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>130</td>
</tr>
<tr>
<td>Winter 2004</td>
<td>201</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>84</td>
</tr>
<tr>
<td>Summer 2004</td>
<td>205</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>131</td>
</tr>
<tr>
<td>Spring 2005</td>
<td>193</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>118</td>
</tr>
<tr>
<td>Van Haarlem et al (2009)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Whole feedlot</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>318</td>
</tr>
</tbody>
</table>

Blank areas represent data which was not reported and could not be calculated based on published values. ^ SEM. reported, SD calculated
4.4 Sensitivity Testing

In a commercial feedlot situation a number of the parameters required for the model are estimated based on group measurements, for example. Individual animal intake is estimated based on an amount fed to the entire pen of cattle, and growth rate/ current live weight is estimated based on the ration fed, the class of cattle, weight at entry and number of days on feed. Therefore there are some issues surrounding the reliability of animal production data. The sensitivity of the models to changes in intake and live weight was therefore examined.

Further, a number of the models, particularly Blaxter and Clapperton (1965) and IPCC Tier II, use set parameters; for example the gross energy content of feed is assumed to be 18.4 MJ/ kg DM. The variation in these parameters based on the rations information from the validation studies was examined, and the effect on changes to these parameters on emissions using the various equations was evaluated.

4.4.1 Intake Prediction

All models of enteric CH$_4$ production require animal intake as an input value. When the ‘system’ modelled is experimental results, accuracy of estimating individual animal intake is not a significant issue, as intake will be exactly measured. However, in a commercial feedlot situation intakes can only be estimated per pen, and based on feed offered. Also, instead of using animal characteristics to estimate intake the National Inventory Methodology (AGO 2006) for feedlot cattle relies on a set class-based-intake value (van Sliedregt et al. 2000).

When cattle are the same class, and demonstrate similar weights and growth rates, alternative predictors of intake (rather than offered feed), often produce the same or very similar input values. This may not be appropriate for cattle which are new to feed, or outside common weight values. Bevans et al. (2005) introduced cattle to high grain diets (gradually or rapidly) and difference in individual intakes between 0.59 and 11.2 kg DM/day for heifers being rapidly adapted and 5.4 to 12.7 kg DM/day for heifers being gradually adapted. This indicates the degree of variation which may be observed within a single pen.

The sensitivity of the models to changes in intake, whilst still maintaining differences associated with ration type, was examined. The same database of studies (Table 4.1) used for the simple validation of the enteric CH$_4$ model were utilised to examine the changes to intake (Table 4.9). The results of Hegarty et al. (2007) were excluded from this examination, as the study did not examine the effects of different rations. CH$_4$ estimates from the model using the measured intake were compared with measured values, and estimates using a set value for
cattle class, intake as a % of live weight and intake calculated using an equation developed to predict forage intake from size and growth of beef cattle (Minson and McDonald 1987).

4.4.2 Gross Energy Estimation

The common value assigned for the gross energy content of feeds is 18.4 MJ/ kg DM. However, feeds commonly used in the feedlot range in GE content from 14.7 MJ/kg DM (grass silage) to 39.3 MJ/ kg DM (oils; Givens and Moss 1990). Weighted average GE concentrations were calculated using the data reported by Givens and Moss (1990) for the rations in the validation studies. Measured and weighted average gross energy contents were then used as inputs to the modelled equations, to examine the impact of the altered GE on predicted CH₄ emission. As discussed previously, there are two predominant kinds of equations used in the modelling process, the energy based models (Blaxter and Clapperton 1965 and IPCC Tier II) and carbohydrate based models (Moe and Tyrrell 1979; Ellis et al. 2007 and Ellis et al. 2009). The carbohydrate based equations do not incorporate an estimate of gross energy therefore the effect of changes to gross energy content was only examined on the equations which utilise GE as a specific input parameter.
Table 4.9 Variation in predicted intake for a number of studies based on a set class based value, the equation of Minson and McDonald (1987), and a value derived from percentage live weight.

<table>
<thead>
<tr>
<th>Study</th>
<th>Ration</th>
<th>Intake inputs</th>
<th>kgDM/head/day</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>As Fed</td>
<td>Method.</td>
<td>Minson</td>
</tr>
<tr>
<td>Corn Backgrounding</td>
<td>6.9</td>
<td>9.8</td>
<td>8.5</td>
</tr>
<tr>
<td>Barley Backgrounding</td>
<td>5.3</td>
<td>9.8</td>
<td>7.8</td>
</tr>
<tr>
<td>Corn Finishing</td>
<td>6.8</td>
<td>9.8</td>
<td>8.8</td>
</tr>
<tr>
<td>Barley Finishing</td>
<td>6.7</td>
<td>9.8</td>
<td>8.9</td>
</tr>
<tr>
<td>Beauchemin and McGinn (2005)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Low F:C*</td>
<td>11.7</td>
<td>9.8</td>
<td>12.2</td>
</tr>
<tr>
<td>High F:C*</td>
<td>10.0</td>
<td>9.8</td>
<td>13.2</td>
</tr>
<tr>
<td>Low Grain</td>
<td>6.2</td>
<td>9.8</td>
<td>10.2</td>
</tr>
<tr>
<td>High Grain</td>
<td>7.5</td>
<td>9.8</td>
<td>10.2</td>
</tr>
<tr>
<td>Lovett et al. (2003)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1 (65:35 F:C*)</td>
<td>6.9</td>
<td>11.7</td>
<td>9.35</td>
</tr>
<tr>
<td>2 (40:60 F:C*)</td>
<td>8.4</td>
<td>11.7</td>
<td>9.38</td>
</tr>
<tr>
<td>3 (10:90 F:C*)</td>
<td>8.2</td>
<td>11.7</td>
<td>9.38</td>
</tr>
</tbody>
</table>

*F: C = forage to concentrate ratio
4.5 Results

4.5.1 Enteric CH$_4$

A significant ($P<0.05$) linear relationship was determined only for prediction based on Moe and Tyrrell (1979) (Table 4.10). This is consistent with visual observations (Figure 4.1 and 4.2) indicating Moe and Tyrrell (1979) appears to have the best fit. Based on concordance (Table 4.11), the best estimates of CH$_4$ emission are those based on the CHO based equations of Moe and Tyrrell (1979) (0.26) and Ellis et al. (2007) (0.29), although concordance values are low. The highest correlation was observed for Moe and Tyrrell (1979) (0.63). This suggests that for feedlot cattle, an equation which is able to represent the feed characteristics will provide the most accurate estimate and allow the reduction in emissions associated with a high grain diet to be fully represented.

Table 4.10 Fitted Linear relationships between measured and predicted CH$_4$ emissions based on five equations utilising the results of published studies

<table>
<thead>
<tr>
<th>Equation</th>
<th>y-intercept</th>
<th>Slope</th>
<th>Significance</th>
<th>SE Obs</th>
<th>SE Pred</th>
</tr>
</thead>
<tbody>
<tr>
<td>Blaxter and Clapperton (1965)</td>
<td>155.5</td>
<td>0.66</td>
<td>$P=0.183$</td>
<td>67.1</td>
<td>17.3</td>
</tr>
<tr>
<td>Moe and Tyrell (1979)</td>
<td>123.5</td>
<td>0.51</td>
<td>$P=0.012$</td>
<td>24.9</td>
<td>6.4</td>
</tr>
<tr>
<td>Tier II</td>
<td>46.6</td>
<td>0.24</td>
<td>$P=0.148$</td>
<td>22.2</td>
<td>5.7</td>
</tr>
<tr>
<td>Ellis et al. (2007)</td>
<td>112.2</td>
<td>0.42</td>
<td>$P=0.141$</td>
<td>38.0</td>
<td>9.8</td>
</tr>
<tr>
<td>Ellis et al. (2009)</td>
<td>158.6</td>
<td>0.08</td>
<td>$P=0.855$</td>
<td>61.2</td>
<td>15.8</td>
</tr>
</tbody>
</table>

Table 4.11 Lin’s concordance correlation coefficients between measured and predicted emissions of CH$_4$ using five equations based on the results of published studies.

<table>
<thead>
<tr>
<th>Equation</th>
<th>Concordance</th>
<th>Correlation</th>
<th>$C_b$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Blaxter and Clapperton (1965)</td>
<td>0.11</td>
<td>0.36</td>
<td>0.29</td>
</tr>
<tr>
<td>Moe and Tyrell (1979)</td>
<td>0.26</td>
<td>0.63</td>
<td>0.41</td>
</tr>
<tr>
<td>Tier II</td>
<td>0.14</td>
<td>0.39</td>
<td>0.35</td>
</tr>
<tr>
<td>Ellis et al. (2007)</td>
<td>0.29</td>
<td>0.40</td>
<td>0.72</td>
</tr>
<tr>
<td>Ellis et al. (2009)</td>
<td>0.04</td>
<td>0.05</td>
<td>0.73</td>
</tr>
</tbody>
</table>
Chapter 4. Model Validation

Figure 4.1 Comparison of measured and predicted emissions (from published studies) of the energetic based models for prediction of enteric \( \text{CH}_4 \) emissions. IPCC Tier I (A), IPCC Tier II (B) and Blaxter and Clapperton (1965)(C). Horizontal error bars indicate SD of measured emissions (as published); vertical error bars indicate SD of predicted emissions (estimated from model output).
Figure 4.2 Comparison of measured and predicted emissions (from published studies) of the CHO based models for prediction of enteric CH\textsubscript{4} emissions. Moe and Tyrrell (1979)(A), Ellis et al. (2007) (B) and Ellis et al. (2009) (C). Horizontal error bars indicate SD of measured emissions (as published); vertical error bars indicate SD of predicted emissions (estimated from model output).
4.5.2 Nitrogen Transactions

In contrast to enteric CH\(_4\) (where different predictive equations were assessed) the objective of this section was to examine the accuracy of the current methodology in predicting N transactions, and therefore how inaccuracy may influence predicted emissions. Methodologies for prediction of N transactions are well established, and the current approach used in the National Inventory Methodology (2006) is based on SCA (1990) and Freer et al. (1997) (updated Freer et al. 2007).

Significant linear relationships \((P<0.05)\) were determined between measured and predicted values for all N transaction parameters (Table 4.12). Calculated concordance (Table 4.13) demonstrates a strong relationship \((0.94, \text{correlation} 0.94)\) between measured and predicted N intake. In contrast to the results of the CH\(_4\) study (Figures 4.1 and 4.2) where variability of measured/predicted emissions was considerably different; variability (as shown by the horizontal and vertical error bars) is quite consistent between observed and predicted value (Figures 4.3 and 4.4).

Table 4.12 Fitted Linear relationships between measured and predicted values for N excretion parameters based on published studies.

<table>
<thead>
<tr>
<th>Equation</th>
<th>y-intercept</th>
<th>Slope</th>
<th>Significance</th>
<th>SE Obs</th>
<th>SE Pred</th>
</tr>
</thead>
<tbody>
<tr>
<td>N Intake</td>
<td>17.6</td>
<td>0.88</td>
<td>(P&lt;0.001)</td>
<td>18.7</td>
<td>4.3</td>
</tr>
<tr>
<td>N Retained</td>
<td>23.7</td>
<td>-0.20</td>
<td>(P=0.034)</td>
<td>3.0</td>
<td>0.7</td>
</tr>
<tr>
<td>N Excreted</td>
<td>29.7</td>
<td>0.91</td>
<td>(P&lt;0.001)</td>
<td>7.09</td>
<td>1.6</td>
</tr>
<tr>
<td>N Urine</td>
<td>-1.6</td>
<td>0.76</td>
<td>(P=0.003)</td>
<td>35.4</td>
<td>8.1</td>
</tr>
<tr>
<td>N Faecal</td>
<td>42.9</td>
<td>0.18</td>
<td>(P=0.018)</td>
<td>12.4</td>
<td>2.9</td>
</tr>
</tbody>
</table>

Despite the significant \((P<0.05)\) relationship (Table 4.12) the model does not accurately predict N retention particularly when measured values are high. Concordance (Table 4.13) is considerably lower than observed with other N parameters (-0.08). Variability (Figure 4.4) (Standard deviations, horizontal error bars) of measured values is considerably higher than predicted values. Similarly to retained N, the accuracy of prediction of faecal N decreases with increasing level of measured N in the faeces (Figure 4.4).

Urinary N is over predicted at lower levels and under estimated at higher levels of measured urinary N. However, a significant \((P<0.05)\) linear relationship between observed and predicted was determined with a moderate concordance (0.53) and correlation (0.64). Variability in measured values is considerably higher than predicted values. When excreted N
is considered as a whole (faecal and urine N) the relationship improves, compared to when they are considered as individual components Lin’s concordance is also improved by considering N excretion as a whole (0.94, compared with 0.53 for urinary N and 0.31 for faecal N alone).

**Table 4.13** Concordance and correlations between measured and predicted values for N excretion based on published studies.

<table>
<thead>
<tr>
<th>Equation</th>
<th>Concordance</th>
<th>Correlation</th>
<th>$C_b$</th>
</tr>
</thead>
<tbody>
<tr>
<td>N Intake</td>
<td>0.94</td>
<td>0.94</td>
<td>0.10</td>
</tr>
<tr>
<td>N Retained</td>
<td>-0.08</td>
<td>-0.49</td>
<td>0.16</td>
</tr>
<tr>
<td>N Excretion</td>
<td>0.94</td>
<td>0.99</td>
<td>0.95</td>
</tr>
<tr>
<td>N Faecal</td>
<td>0.31</td>
<td>0.54</td>
<td>0.57</td>
</tr>
<tr>
<td>N Urine</td>
<td>0.53</td>
<td>0.64</td>
<td>0.83</td>
</tr>
</tbody>
</table>
Figure 4.3 Comparison between measured and predicted values of N intake (A), N retention (B) and Excretion of faecal N (C) based on published studies. Horizontal error bars indicate SD of measured emissions (as published); vertical error bars indicate SD of predicted emissions (estimated from model output).
Figure 4.4 Comparison between measured and predicted values of Urinary N excretion (A), Total N excretion (B) and volatile NH$_3$ (C) based on published studies. Horizontal error bars indicate SD of measured emissions (as published); vertical error bars indicate SD of predicted emissions (estimated from model output).
4.5.3 Gaseous Nitrogen Emissions

For a number of these studies, particularly where emissions were measured using micrometeorological methods, limited data was provided regarding the ration and intake (McGinn et al. 2007, Todd et al. 2008, van Haarken et al. 2008). Further, intakes and live weights (particularly for whole feedlot measurements) are likely to be an average of all animals on the lot at the time of measurement. Where ration information was limited, N intake as reported by the study was used as an input to the N Intake component of the model.

Concordance calculations (Table 4.15) demonstrate a low relationship (0.2827) between observed and predicted NH$_3$ emissions, although a significant ($P<0.001$) linear relationship was fitted (Table 4.14). The model appears to be able to reasonably accurately predict emissions at lower levels, but accuracy decreases as measured emissions increase, with considerable underestimation at high levels of measured NH$_3$ (Figure 4.4c). Variability was not reported for estimates on an individual animal basis in the majority of studies utilised in the validation process. These studies have primarily been micrometeorological work; therefore per head emissions are extrapolated from whole site measurements of concentration, variability is reported for measured concentrations, but not for values on a per head basis.

**Table 4.14** Linear relationships between measured and predicted emissions of ammonia based on published studies. There was insufficient data to evaluate a linear relationship between measured and predicted values of nitrous oxide.

<table>
<thead>
<tr>
<th>Equation</th>
<th>y-intercept</th>
<th>Slope</th>
<th>Significance</th>
<th>SE Obs</th>
<th>SE Pred</th>
</tr>
</thead>
<tbody>
<tr>
<td>NH$_3$</td>
<td>28.3</td>
<td>0.21</td>
<td>$P&lt;0.001$</td>
<td>6.7</td>
<td>2.0</td>
</tr>
</tbody>
</table>

**Table 4.15** Concordance and correlations between measured and predicted emissions of Nitrogenous gases

Very limited data is available which links animal production characteristics to N$_2$O emissions from feedlot systems, resulting in inability to fit a linear relationship (Table 4.14) and very low concordance (0.00, Table 4.14). However, a correlation of 1.0 was determined- related to the low sample size. Although measurements have increased in this area the majority are
undertaken using micrometeorological measurements, or using manure in chambers, which do not always provide the details required for the modelling approach used here. Boadi et al. (2004b) report N\textsubscript{2}O of 2.2 and 2.4 g/pen/day, equating to 0.15 g/head/day for the experimental pens of 14 head. This is considerably lower than predicted emissions from this study (based on the model described above) of 7.0 and 5.4 g/head/day. However, the study of Boadi et al. (2004b) was conducted under conditions of very low temperatures; therefore they are unlikely to represent the range of possible emissions. Modelled emissions based on the data provided in the other studies used in the gas validation (only NH\textsubscript{3} was reported) range from 3.3 to 10.7 g/head/day.

### 4.5.4 Sensitivity Testing

In most cases (excluding Boadi et al. 2004b) the estimates of intake (Table 4.9) using all alternate methods are considerably greater than intakes recorded. Simple linear regression and concordance calculations were used to examine the relationship between measured intake and the alternative estimates of intake utilised in the methodology. Significant linear relationships were determined between offered feed (measured intake) and intake predicted using Minson and McDonald (1987) \((P<0.01)\) and as a percentage of live weight \((P<0.001)\) (Table 4.16).

Lin’s concordance calculations (Table 4.17) showed the highest concordance between measured intake and intake estimated as a percentage of live weight (0.30, correlation 0.90). The equation of Minson and McDonald (1987) showed moderate correlation (0.53) with measured intake values, but no concordance (0.00), whilst a set value based on class of cattle had very low correlation and concordance (0.05 and 0.01) with measured intakes.

Despite the low concordance (Table 4.16) between measured and class-based-intake estimates, utilising a class-based-intake actually improves the correlation between measured and predicted CH\textsubscript{4} emissions.

Reported gross energy content (Table 4.17) varied from the set value (18.4 MJ GE/kg DM), with a range of 17.8 to 19.5 MJ GE/kg DM measured in the examined studies. Weighted average GE content was not concordant with the measured value (concordance -0.06, correlation 0.13). Weighted average GE content was more variable (more difference between min and max calculated values) for higher forage rations (e.g. backgrounding vs. finishing in Beauchemin and McGinn (2005) and the 65:35 in Lovett et al. (2003)). This reflects the variability in quality of the forage crops/products. It also suggests that using a fixed value for gross energy may not but suitable for all ration types, even within “feedlot” type rations.
Concordance calculations demonstrate a high level of concordance between the emissions predicted using the standard 18.4 MJ value for GE and the GE value reported by the studies (0.99 tier II and 0.97; Blaxter and Clapperton 1965) and with the standard value and a calculated weighted average based on ration composition (0.94 Tier II and 0.92; Blaxter and Clapperton 1965). Strong correlations (>0.95) are also observed between emissions predicted using the standard value, reported values and the calculated weighted average value for 18.4 (MJ GE/kg DM).

**Table 4.16** Concordance and Correlations between measured CH$_4$ output and CH$_4$ output predicted using measured intake, intake as a set value based on cattle class, intake calculated based on the equation of (Minson and McDonald 1987), and intake as a percentage of live weight. Predicted CH$_4$ is based on five different equations.

<table>
<thead>
<tr>
<th>Equation</th>
<th>Intake</th>
<th>Concordance</th>
<th>Correlation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tier II</td>
<td>Measured</td>
<td>0.01</td>
<td>0.03</td>
</tr>
<tr>
<td></td>
<td>Class</td>
<td>0.13</td>
<td>0.43</td>
</tr>
<tr>
<td></td>
<td>Minson and McDonald</td>
<td>-0.02</td>
<td>0.06</td>
</tr>
<tr>
<td></td>
<td>% Live weight</td>
<td>-0.13</td>
<td>-0.23</td>
</tr>
<tr>
<td>Blaxter and Clapperton (1965)</td>
<td>Measured</td>
<td>0.05</td>
<td>0.02</td>
</tr>
<tr>
<td></td>
<td>Class</td>
<td>0.02</td>
<td>0.31</td>
</tr>
<tr>
<td></td>
<td>Minson and McDonald</td>
<td>-0.01</td>
<td>-0.72</td>
</tr>
<tr>
<td></td>
<td>% Live weight</td>
<td>-0.03</td>
<td>-0.31</td>
</tr>
<tr>
<td>Moe and Tyrrell (1979)</td>
<td>Measured</td>
<td>0.19</td>
<td>0.52</td>
</tr>
<tr>
<td></td>
<td>Class</td>
<td>0.14</td>
<td>0.65</td>
</tr>
<tr>
<td></td>
<td>Minson and McDonald</td>
<td>0.11</td>
<td>0.47</td>
</tr>
<tr>
<td></td>
<td>% Live weight</td>
<td>0.08</td>
<td>0.45</td>
</tr>
<tr>
<td>Ellis et al. (2007)</td>
<td>Measured</td>
<td>0.09</td>
<td>0.14</td>
</tr>
<tr>
<td></td>
<td>Class</td>
<td>0.11</td>
<td>0.67</td>
</tr>
<tr>
<td></td>
<td>Minson and McDonald</td>
<td>0.04</td>
<td>0.13</td>
</tr>
<tr>
<td></td>
<td>% Live weight</td>
<td>0.00</td>
<td>0.01</td>
</tr>
<tr>
<td>Ellis et al. (2009)</td>
<td>Measured</td>
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<td>-0.21</td>
</tr>
<tr>
<td></td>
<td>Class</td>
<td>-0.06</td>
<td>-0.15</td>
</tr>
<tr>
<td></td>
<td>Minson and McDonald</td>
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<td>-0.28</td>
</tr>
<tr>
<td></td>
<td>% Live weight</td>
<td>-0.10</td>
<td>-0.32</td>
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</table>
### Table 4.17 Reported and calculated* weighted average GE concentrations of rations used in the validation studies

<table>
<thead>
<tr>
<th>Study</th>
<th>Ration</th>
<th>Reported</th>
<th>Average</th>
<th>Minimum</th>
<th>Maximum</th>
</tr>
</thead>
<tbody>
<tr>
<td>Beauchemin and McGinn (2005)</td>
<td>Corn Backgrounding</td>
<td>18.3</td>
<td>18.0</td>
<td>16.6</td>
<td>18.7</td>
</tr>
<tr>
<td></td>
<td>Barley Backgrounding</td>
<td>18.6</td>
<td>17.8</td>
<td>17.1</td>
<td>18.8</td>
</tr>
<tr>
<td></td>
<td>Corn Finishing</td>
<td>18.2</td>
<td>18.3</td>
<td>17.9</td>
<td>18.5</td>
</tr>
<tr>
<td></td>
<td>Barley Finishing</td>
<td>18.1</td>
<td>18.0</td>
<td>17.4</td>
<td>18.4</td>
</tr>
<tr>
<td></td>
<td>LF:G</td>
<td>17.8</td>
<td>17.5</td>
<td>17.0</td>
<td>18.0</td>
</tr>
<tr>
<td>Boadi et al. (2004b)</td>
<td>High F:G</td>
<td>18.8</td>
<td>15.2</td>
<td>14.6</td>
<td>15.9</td>
</tr>
<tr>
<td>Beauchemin and McGinn (2006)</td>
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<td>18.7</td>
<td>17.7</td>
<td>17.0</td>
<td>18.8</td>
</tr>
<tr>
<td></td>
<td>HG</td>
<td>18.7</td>
<td>18.2</td>
<td>17.7</td>
<td>18.8</td>
</tr>
<tr>
<td>Hegarty et al. (2007)</td>
<td>-</td>
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<td>16.8</td>
<td>16.2</td>
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</tr>
<tr>
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<td>65:35</td>
<td>19.5</td>
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<td>19.5</td>
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<td>Lovett et al. (2003)</td>
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<td></td>
<td>10:90</td>
<td>17.9</td>
<td>18.1</td>
<td>17.2</td>
<td>18.8</td>
</tr>
</tbody>
</table>

*based on (Givens and Moss 1990)
4.6 Discussion

4.6.1 Accuracy of the Models for Feedlot Systems

The accuracy of a number of the models used under Australian and International methodologies for reporting emissions has been questioned (ALFA 2008), particularly in the context of Carbon Trading and the Carbon Farming Initiative (Australian Government 2010a). The National Inventory Methodology (AGO 2006) for enteric CH$_4$ emissions from feedlot cattle; Moe and Tyrell (1979) and the equation developed by Ellis et al. (2007) performed the best, with a concordance of 0.26 and 0.29 respectively (although these correlations are still poor). The approach of these equations is similar, basing potential CH$_4$ emissions from a diet on fermentative properties (fibre/carbohydrate composition) rather than the total gross energy and intake of the diet (as in Blaxter and Clapperton 1965 and IPCC Tier II). The factors used are different between the two equations, but the most significant difference in terms of approach is the inclusion of metabolisable energy intake as a factor in Ellis et al. (2007). Both Moe and Tyrell (1979) and Ellis et al. (2007) have the disadvantage of using parameters which are not commonly measured in a commercial situation, for example Moe and Tyrell (1979) relies on estimated soluble residue, hemicellulose and cellulose, and Ellis et al. (2007) on ADF and lignin. This is suggested by a number of sources as a source of error in the prediction of CH$_4$ emissions, particularly when tabulated values are required. Yan et al. (2009) observed Ellis et al. (2007) to have a poor predictive ability for the range of diets evaluated.

In general, the equations based on energy appeared to have poor predictive ability for feedlot situations. The Tier II equation (concordance 0.14) appears to closely estimate emissions when the grain content of the ration is high (greater than 90%), however in practice, particularly in Australian feedlots, grain content is unlikely to be this high. This type of model may have potential with the use of a diet specific $Y_m$ factor, rather than a set value for a type of cattle (Kebreab et al. 2008). Estimates may also be improved by using offered feed in place of a set value for intake. Both the energy based equations (Tier II and Blaxter and Clapperton (1965)) are affected significantly by changing intake and GE estimation. These models are unable to reflect differences related to ration composition (when a consistent group of cattle are fed different rations), when intake is estimated on a class basis, or as a proportion of live weight, and when a set value is used for GE. The National Inventory Methodology (AGO 2006) currently uses class-based-intake values, and a set GE, so it is likely that it would be unable to identify feedlots which are using dietary changes to reduce CH$_4$ emissions if these equations were used.
The trend for all the more complex approaches (excluding Tier II, 3% GEi) is for over estimation of emissions. Over estimation (average) ranges from 33 g/head/day to 109 g/head/day, on average Tier II under estimates by 55 g/head/day. The primary reason for inaccuracies in any of the methodologies is consistently identified as an effect of applying the model on a range of data which is dissimilar to that on which the model was developed (Mills et al. 2003; Wilkerson et al. 1995). For CH₄ emissions, the most common source of this variation is in ration components, although animal characteristics may also contribute. A particular dietary parameter which is likely to result in overestimation of emissions using the models evaluated is starch. Increasing starch in the ration alters rumen fermentation, increasing propionate relative to acetate and decreasing hydrogen production (for the same amount of consumed dry matter). As noted by Boadi et al. (2004a) fermentation of structural CHO (cellulose, lignin etc) results in greater CH₄ loss than the fermentation of soluble starch and sugars. Structural CHO fermentation favours a decreased rate of passage and increased production of acetate relative to propionate. Further, rapid fermentation of starchy grains (at high intake levels) favours rapid rumen fermentation and increased propionate production. Decreased rumen pH also restricts growth and activity of methanogens (Hegarty and Gerdes 1998). An example of the effect of alterations in CHO composition (ADF, NDF and starch) with different grains is the differing CH₄ emissions observed for corn and barley based diets by Beauchemin and McGinn (2005). The starch content of barley grain was 54.5% and of corn 66.7%. When used in a high concentrate (finishing) diet, emissions were less for a corn based diet (9.2 g/kg DM) than barley based diet (13.1 g/kg DM).

Hindrichsen et al. (2004) tested a range of feed types varying in CHO properties (using isonitrogenous and isocaloritic diets) in vitro and concluded that equations predicting CH₄ from CHO fractions need to include starch and sugars as well as commonly evaluated carbohydrate fractions. Although the equation of Moe and Tyrell (1979) includes a parameter (soluble residue) which is primarily made up of starch, the proportion of soluble residue in the diets utilised in the development of the model averaged 1.8 kg/day (.27 to 3.83 kg/day). As previously discussed, inaccuracies in the majority of modelling approaches tend to arise when they are applied to data outside the range on which they were developed (Mills et al. 2003). For the majority of studies utilised in this evaluation intake of soluble residue was greater than 5 kg/day, which is significantly greater than those used in the development. This may result in artificially high predicted values, where the impact of a high starch feed on fermentation and microbial populations are not adequately represented by the models.

Further, these calculations of soluble residue are based on a standard value (according to the National Inventory Methodology, AGO 2006). Soluble residue of concentrates is set at 0.19,
grains 0.68 and roughages 0.21. However, as discussed in terms of GE, there can be considerable variation between individual grains and roughages within these feed types. Commonly used grains in Australian feedlot rations include barley and wheat in the south, and sorghum in the north. Givens and Moss (1990) reports starch contents of 562 g/kg DM of barley, 730 g/kg DM for sorghum and 674 g/kg DM for wheat, suggesting that a single value for a feed type is unlikely to represent the range of soluble residue in a feed type.

Of key importance of assessing the impact that difference in specific dietary parameters is the role of intake in influencing the amount of CH$_4$ produced/kg of component digested. Moe and Tyrell (1979) report that CH$_4$ production from carbohydrate depended more on the amount fermented than the type of carbohydrate. At higher intakes >3.5 times maintenance (intake requirement to maintain the animal in steady state) CH$_4$ production was influenced more by type of carbohydrate than at lower intakes. Moe and Tyrell (1979) determined at intakes below 1.5 times maintenance requirement CH$_4$ production is nearly as related to dry matter intake or total digestible nutrients, but at intakes greater than 1.5 times intake consideration of the amount of individual carbohydrate fractions digested. Relative intakes calculated for the studies utilised in this evaluation (for enteric CH$_4$) range from 0.5 to 1.05 (based on a standard reference weight of 660 kg), suggesting that dry matter intake or total nutrients may be more highly correlated with CH$_4$ emissions than carbohydrate fractions for this range of data. However, these studies may not be entirely representative of an ad lib feeding feedlot situation (especially where calorimetric methods may have affected intakes and behaviour). Beauchemin and McGinn (2005) reports that measured CH$_4$ emissions in their study may underestimate those from commercial feedlot cattle due to a drop in intake associated with chamber measurements.

The model for N transactions very accurately predicts N intake ($R^2$ 0.94, concordance 0.94). This is not surprising, given this requires only intake and feed N concentration (the greatest source of error here being intake estimation). Errors in the estimation of N intake occur at higher levels, perhaps associated with N concentration in the feeds or variation in the conversion factor between N and crude protein. In general, a factor of 6.25 is used to convert N to crude protein, however represents an average value (similar to the 18.4 MJ/kg DM set as the gross energy of feed). These higher N intake diets, such as Cole et al. (2005) and Adams et al. (2004), contain byproduct feeds such as cottonseed meal (Cole et al. 2005) and supplements containing urea, blood meal and feather meal (Adams et al. 2004). These supplements are higher in non-protein N than traditional grains and forages, which are not accounted for using a single factor for the N content of a feed. This is likely to have a flow on effect to N excreted and therefore the balance between predicted and measured emissions.
Nitrogen retention is poorly estimated by the model, \( R^2 = 0.49 \), concordance -0.01. This model uses relative intake, relative size and live weight gain in prediction of retained N. The errors in this estimation suggest problems with the standard reference weights, and calculations of relative intake may not be suitable for feedlot type cattle. Similarly, the model of \( \text{CH}_4 \) production, produced by Blaxter and Clapperton (1965), which utilises a value of relative intake has been found to overestimate emissions, compared with measured values. Both relative size and intake are based on the concept of a standard reference weight. Standard reference weight is defined as the weight of an animal when it reaches mature skeletal size and has a condition score in the middle of the range (Freer et al. 2007). Feedlot rations are designed to produce high rate of body growth and protein deposition, compared with standard cattle growth, particularly on forage based diets. It has been observed by a number of authors that N retention differs between ration types, with the primary effects related to N intake, DMI and effects of live weight gain (Adams et al. 2004; Farran et al. 2006). However, specific effects have been reported with changes to forage type, grain types and processing and method of protein feeding (e.g. phase feeding vs. constant protein level; Cole et al. 2006). Addition of rumen modifiers to the diet is suggested to influence propionate production and consequentially protein anabolism (Freer et al. 2007). The use of these compounds (e.g. monensin) is increasingly common in feedlot systems and may result in N transactions differing from predictions when its effects are not considered. Freer et al. (2007) suggest alternations of the efficiency of growth factor to compensate for these changes when modelling N retention.

Further, there are likely to be considerable difference between breeds in N retention. Moore et al. (1975) determined significant differences in N retention between Angus, Hereford and Brahman steers. \( B. \text{Taurus} \) breed steers exhibited higher N retention than \( B. \text{Indicus} \) steers fed high energy rations, although these differences are less pronounced when lower energy rations are fed. In addition to inaccuracies observed in the model here, utilising this approach for a wider selection of Australian feedlot producers. Cattle populations in Northern feedlots (depending on the season) can be predominantly \( B. \text{Indicus} \) type, while those in the Southern feedlots are more commonly \( B. \text{Taurus} \).

Validation of this component of the model is likely to be complicated by the methods used to produce N retention in the published studies. Nitrogen retention in the majority of studies used in the validation process is estimated based on animal live weights and live weight gains using the procedures outlined by the (NRC 1996); the same approach used in the IPCC guidelines. Effectively, this results in the results of one model being used to validate the
results of a different model for N retention. This is likely to be a contributing factor to the high level of variability observed in measured emissions.

The estimation of faecal N output is also fairly poor ($R^2$ 0.53, concordance 0.31). Greater errors are observed at higher levels of measured faecal N excretion. Faecal N excretion is estimated using digestibility, metabolisable energy, crude protein and feed intake. In the general modelling approach a fixed value of 80% is used for the digestibility of the ration, ME is also assumed based on this DMD (12 MJ ME/ kg MD). A fixed feed intake is additionally incorporated for feedlot cattle. In the initial validation these fixed values were used, some inaccuracy in the estimates of faecal N compared with measured values would therefore be expected.

The accuracy of faecal N estimates decreases, with increasing levels of measured faecal N. The higher measured values for faecal N result from the study of Adams et al. (2004). Steers were kept in small feedlot pens for periods of 180 days during winter and 132 days during summer. Total N intake and N excretion (measured from collection of manure and measurements of changes in soil N) were determined for the whole period. Manure N in this case was calculated based on the amount of material scraped from the pen, and the N content of this material. It is likely that this over estimated manure N, the value of faecal N in this study (Adams et al. 2004) ranged from (the equivalent of) 90 to 190 g/steer/day, compared with 30 to 60 g/head/day in the other studies (Cole et al. 2003; Cole et al. 2006; Vasconcelos et al. 2006). Despite the majority of urinary N being volatilised under most conditions there is likely to have been some contribution from this source, particularly during the winter measurement period. There may also have been a contribution of non volatile forms of N, for example hair and scurf, feed waste and bedding material. The results of Cole et al. (2003) are also inconsistent with other results, having considerably higher N intakes (average 218 g/head/day) and lower faecal N (average 30 g/day) compared with the other studies used in the validation. It is well known that altering the concentration and ruminal degradability of feeds alters N partitioning. It is possible that these diets, while having a higher level of N, contained protein supplements which were quickly degradable (e.g. Urea), resulting in more N being partitioned to urine (Cole et al. 2005).

In contrast to N retention and faecal N excretion urinary N excretion is estimated quite accurately by the model ($R^2$ 0.99, concordance 0.94). The model predicts urinary N excretion based on N intake, subtracting N retained and faecal N. Interestingly, despite the inaccuracy in retained N and faecal N, urinary N is still estimated well. This suggests that the model is able to estimate N “use” in the animal (if faecal N and retained N are considered together),
but has errors when attempting to partition N between these areas. As urinary N is the major pathway of N excretion, and the primary source of both volatile NH$_3$ and emissions of N$_2$O, it is possible that the inaccuracy associated with faecal N excretion may not result in significant alterations in predicted greenhouse gas emissions. This is confirmed by the examination of total N excretion (Figure 4.4b). Considering total N excretion rather than the individual components improves estimation based both on a regression between measured and modelled and on concordance and correlations.

The use of a single factor to convert urine and faecal N to potential volatile NH$_3$ and N$_2$O is the primary contributor to the inaccuracy associated with prediction of gaseous emissions. Despite a reasonable prediction of urinary N, which is the major contributor to emissions of both gases, NH$_3$ emissions are quite significantly under estimated (Figure 4.4c), although accuracy is better at lower measured values. Direct emissions of N$_2$O were also significantly over estimated by the model, although limited studies were able to be investigated. The major issue which arises from attempting to predict gaseous N emissions from dietary components and animal performance is the role which environmental conditions play in the emissions process. As outlined in the literature review (Sections 2.4 to 2.6), moisture content, pH, temperature, manure pack depth and organic matter content have a significant impact on nitrification and denitrification, and volatilisation of NH$_3$.

Emissions of NH$_3$ vary between seasons, based on temperature and rainfall (contributing to soil/ manure pack moisture). Todd et al. (2005) determined NH$_3$ emissions of 55% of feed N during summer, and only 27% of feed N during winter. They determined an emission factor of 15 kg NH$_3$/head/year, in contrast to the USEPA factors of 11 kg/head/day. Similarly Klopfenstein and Erickson (2002) determined that 60-70% of feed N was lost as NH$_3$ in summer, and 40% during winter. Todd et al. (2005) report that NH$_3$ emissions were greatest in hot dry weather, and precipitation suppressed NH$_3$ emissions. Denmead et al. (2008) determined emission rates of 69 g NH$_3$-N/animal/day for a temperate Australian site and 24 g NH$_3$-N/animal/day for a tropical Australian site (during winter). They attributed this to differences in surface wetness, which was higher at the temperate site and therefore allowed increased volatilisation of NH$_3$; this contrasts with the observation of Todd et al. (2005). Loh et al. (2008) report summer emissions from the same sites during summer, equal to 170 g NH$_3$-N/head/day (tropical) and 117 g NH$_3$-N/head/day (temperate). In a discussion of these results, Denmead et al. (2008) also cite surface wetness as the major factor influencing emission rate.
Although there is insufficient data to estimate concordance between measured and predicted N$_2$O emissions; predicted N$_2$O emissions from these studies ranged between 3.3 and 10.7 g/head/day. Denmead et al. (2008) report emissions of 1.1 and 1.8 g N$_2$O-N/head/day from Australian feedlots, whilst Boadi et al. (2004b) determined N$_2$O emissions equivalent to 0.16 g/head/day (using chamber incubations), which was approximately 50% of that estimated using IPCC calculations (the same coefficient used in the methodology examined here). Further examination of the source of the IPCC emission factor for dry lot manure management reveals a number of issues in the potential application of the source study to a feedlot situation. According to the IPCC methodology- the default emission factor of 0.02 for N$_2$O emissions was developed based on the work of Kulling et al. (2003) and consultation with the expert committee. The study of Kulling et al. (2003) was focussed on emissions from dairy manure comparing three management systems (slurry, liquid and farmyard manure) and two rations types (hay or grass based). It can be assumed that the farmyard manure management system was considered similar to feedlot, as this was the driest and most compacted material. However, this material was amended with straw, in order to replicate the bedding system used in Switzerland. The types of rations used, resultant changes in manure composition compared with a feedlot suggest that emission factors developed for this work are unlikely to be entirely applicable to dry lot management systems. However, Kulling et al. (2003) reports emissions of N$_2$O between 1.7 and 2.8 g/head/day, very similar to those measured in an Australian feedlot (Denmead et al. 2008).

4.6.2 Sensitivity of Models

Changing intake prediction influences the models differently. The simplest model, Tier II (IPCC 2006), which was shown by the validation process to be mid range in terms of concordance between measured and modelled emissions, is significantly affected by changing intake. When cattle are fed different rations, we would expect that a higher grain/ more digestible ration to result in lower CH$_4$ emissions, which is reflected in the measured values. When a class based or % live weight intakes are applied to a consistent group of cattle, the Tier II model does not reflect any differences in ration composition (this is associated with the use of a set factor for GE. A similar issue is associated with the Blaxter and Clapperton (1965) equation, which uses a set value for GE and digestibility, with the differences in emissions therefore resulting from intake (through the relative intake parameter). The carbohydrate equations, in contrast, are able to reflect differences in ration composition, even when intake is estimated using a fixed value for the class or live weight of the animal. If these equations were to be utilised for inventory purposes it would enable the reduction in enteric emissions associated with diet type to be accounted for, however, an assessment of intake
values set for a specific class may need to be undertaken, given the overestimation of intake for all equations examined here.

As the National Inventory Methodology (AGO 2006) currently uses a fixed value for the class of feedlot cattle (referred to in this discussion as the ‘methodology’ estimate of intake) the higher estimate is likely to result in over estimation of emissions, compounding the issues with over estimation which are already observed with most models of enteric \( \text{CH}_4 \). As the most concordant estimate of intake was the equation of Minson and McDonald (1987), it would be assumed that predicting intake using this equation would improve the estimate of \( \text{CH}_4 \) from the model, relative to the fixed values from the methodology and as a % of live weight. For inventory purposes, it would be impractical to record intake of individual animals on a daily basis, or even over the time spent on feed, for the calculation of emissions. Even calculating based on growth is impractical as measurements such as weight are not commonly calculated on an individual basis in the feedlot (as well offered feed is on a per pen basis). However, these results suggest that improved estimates of intake are required; if a set value (for intake) is going to be used in a Tier II approach.

### 4.6.3 Further Considerations

When evaluating the results of the validation process, the difference in standard deviation (as represented by the error bars) between measured and modelled emissions is a concern. For many of the models (e.g. Tier II, Moe and Tyrell 1979, Ellis et al. 2007 and Ellis et al. 2009) modelled emissions are significantly less variable than measured emissions. For Tier II, this is likely to be related to the parameters set by the model, feed gross energy is set at 18.4 MJ GE/ kg DM, and CH\(_4\) as a percent of gross energy at 3%. As discussed individual feed gross energy can range from 14.7 MJ/ kg DM to 39.3 MJ/ kg DM (Givens and Moss 1990), and the reported values for the validation study rations ranged from 17.8 to 19.5 MJ GE/ kg DM. Although a high level of concordance was observed between modelled emissions using a variety of different GE values using set values for these parameters would significantly reduce the variation associated with modelled emissions. A single figure of 3% does not, therefore, demonstrate the natural variation in emissions, although it provides a reasonable overall estimate for higher concentrate rations.

The reason for the lower variation in the other equations, which are feed/ carbohydrate based, is likely to be similar. These equations all use ration contents of factors such as hemicellulose, cellulose, soluble residue etc, and calculate potential emissions from these fractions. However, the accuracy of these estimates can be significantly affected by the method used in determination, and even by the feed and forage type. Hemicellulose and forage cellulose, key
parameters in both Moe and Tyrell (1979) and Ellis et al. (2009) is often estimated using NDF and ADF content of feeds, or using ADF minus sulphuric acid lignin (Hans-Joachim 1997). Hemicellulose, when estimated as NDF-ADF is often overestimated by the presence of non extracted proteins in the NDF (Hans-Joachim 1997). For this to provide a reasonable estimate of hemicellulose, NDF and ADF need to be sequentially determined. The original method of determination of NDF and ADF (Van Soest et al. 1991) has been modified considerably since its conception and therefore equations developed based on the original method (e.g. Moe and Tyrell 1979) may be inaccurate when values determined using the modified method are utilised in the calculations (Hao et al. 2001a).

Ellis et al. (2007) uses ADF and lignin; as reported by Hans-Joachim (1997) the measurement of lignin is complicated due to variable methods (at this point there was no definitive reference method). The most common method is acid detergent lignin, however evidence suggests that lignin is underestimated using this method (Hans-Joachim 1997). However, Hindrichsen et al. (2004) examined a variety of methods for estimating cellulose, hemicelluloses and lignin and determined high correlations between methods for cellulose (0.99) and lignin (0.93), but low correlation for hemicellulose (0.44). They attribute this to high amounts of soluble hemicellulose in some feeds, which was washed out during NDF analysis. However, this highlights the impact that measurement method may have on measured parameters and consequentially the potential for errors in prediction based on this data.

Further, the parameters used in this equation (soluble residue, cellulose and hemicelluloses) are not commonly measured in commercial feed testing, and are therefore estimated based on fixed values in the methodology. There can be a significant variation in the estimation of these parameters, depending on the feed type. An examination of the data used in the development of the model demonstrates the range in the contents of soluble residue, hemicellulose and cellulose with the rations used. Soluble residue varied from 18.1 to 54.1%, hemicellulose from 7.1 to 28.2% and cellulose from 8.5 to 34.3% (Moe and Tyrrell 1979).

As discussed by Yan et al. (2000) most of the earliest developed equations were based on animals offered dry or high DM forages, this may not reflect the lower DM silages used in feedlot rations. Fermentation processes in silages which result in low concentrations of water soluble CHO, and high levels of fermentation products (e.g. VFA) which result in differences in rumen processes (Yan et al. 2000), reducing the ability of the model to reflect rumen processes, hydrogen production and therefore CH$_4$ emissions.
Measurement method can have a significant impact on reported emissions, and therefore on the perceived accuracy of models. Boadi et al. (2004b) measured CO\textsubscript{2} and CH\textsubscript{4} emissions using both the SF\textsubscript{6} tracer technique and calorimetric methods; although animals and rations remained constant a significant difference was observed in CO\textsubscript{2} output. Similarly, Grainger et al. (2007) compared the chamber and SF\textsubscript{6} measurements of CH\textsubscript{4} and determined significant differences in measured emissions of CH\textsubscript{4}, which could not be entirely accounted for by hind gut CH\textsubscript{4} production. Yan et al. (2009) suggest that the poor predictive ability of some models (e.g. Ellis et al. (2007)) may be related to some of the data used in the development being measured using the SF\textsubscript{6} techniques, which does not measure emissions from the hindgut, therefore under estimating emissions. In contrast Ellis et al. (2009) did not find measurement method to have a significant effect on CH\textsubscript{4} output, when CH\textsubscript{4} was expressed as MJ/day, or as % of gross energy.

Although the National Inventory Methodology (AGO 2006) for estimating manure CH\textsubscript{4} takes into consideration differences between environmental conditions between tropical and temperate regions of the country (CH\textsubscript{4} conversion factors of 5% and 1.5% are used for tropical and subtropical areas respectively) this is not followed through to either volatile NH\textsubscript{3} or N\textsubscript{2}O. A number of studies have determined significantly different NH\textsubscript{3} emissions between seasons and feedlot sites. This suggests that at a minimum, modifications need to be made to the emission factor used in order to more accurately represent NH\textsubscript{3} volatilisation. The current emission factor used in the inventory is 30 % of excreted N, or 0.3 kg/kg excreted N. This is considerably lower than calculated emission factors of 0.6 (Denmead et al. 2008), 0.6-0.7 (Bierman et al. 1999) and 0.9 (Loh et al. 2008). As evidenced by the validation study, the emission factor (or fraction of volatile N) is too low under some circumstances, and does not account for differences in environmental conditions which have a significantly impact on emissions.

4.7 Conclusion

Although there has been recent research into emissions from Australian beef and dairy systems, including feedlots (Alford et al. 2006; Charmley et al. 2008; Chen et al. 2009; Denmead et al. 2008; Hegarty et al. 2007; Loh et al. 2008; McGinn et al. 2008), the accuracy of the current National Inventory Methodology (2006) for the estimation of Greenhouse sources and sinks (as it applies to feedlot systems) has not been formally examined under commercial conditions.

This validation exercise suggests that for the most accurate estimation of emissions from feedlot systems, and the ability to distinguish between diet types, a methodology based on
dietary composition is required. The National Inventory Methodology (AGO 2006) equation; Moe and Tyrell (1979), has the benefit of utilising this approach, and along with Ellis et al. (2007) provide the best estimates of CH$_4$ emissions. However, the IPCC Tier II approach merits further investigation.

Despite errors in estimating retained N and faecal N, urinary N and total N excretion are estimated reasonably well by the model. However, this is not translated into accurate estimation of emissions of NH$_3$. Insufficient data was available to determine accuracy of models for N$_2$O, however predicted emissions from the studies utilised were higher than the range of measured emissions from beef cattle. Errors in estimating the emissions of NH$_3$ using this methodology are likely to result from the use of a single “emission factor” which does not represent emission potential under all environmental conditions.
Chapter 5. Measured Emissions and Application of a Model for Estimation of Greenhouse Gas Emissions from Australian Beef Feedlots

5.1 Introduction

Under the United Nations Framework Convention on Climate Change (UNFCCC) the Australian government is obligated to produce inventories of annual emissions of greenhouse gases produced under agricultural (and other industrial) activities. The inventories are produced based on a methodology developed by The National Greenhouse Gas Inventory Committee (NGGIC), published by the Australian Greenhouse Office (AGO) in accordance with IPCC guidelines (IPCC 2006). Globally, there is a lack of data available on baseline emissions from cattle breeds (particularly tropical (B. Indicus) breeds) and life stages representative of the feedlot industry (Stackhouse et al. 2011), although there has been considerable focus on greenhouse gas emissions from Canadian and Northern American feedlot cattle, in more recent years (Beauchemin and McGinn 2005; Beauchemin and McGinn 2006; Boadi et al. 2004b; Kebreab et al. 2006; McGinn et al. 2009; van Haarlem et al. 2008). Locally, concerns have been expressed by industry bodies regarding the accuracy of this method (ALFA 2008) and about the impact of errors in emissions estimates under a carbon constrained future on profitability and industry sustainability. There has also been limited validation of models commonly used for predicting emissions from feedlot cattle, particularly with respiration chamber experiments (Kebreab et al. 2006; Stackhouse et al. 2011).

Meat and Livestock Australia studies suggest that the feedlots are already paying $2.35/standard cattle unit on environmental management, without the further cost of managing a carbon reduction/trading scheme (Monni et al. 2007). Accurate estimates are important under all prospective applications of a carbon constrained economy, in order for mitigation options to be accounted for, and to reduce the economic impact of a direct Carbon Tax.

The equations developed to predict emissions of CH$_4$ are based on measurements conducted in order to examine the metabolisable energy partition from feed GE (GE minus loss of energy in faeces, urine and gases). Blaxter and Clapperton (1965) produced their equation (which now forms the basis of a number of national inventories) in order to predict losses of energy as CH$_4$ by cattle and sheep from knowledge of feed composition and intake. Similarly,
Moe and Tyrrell (1979) assert the need for direct measurements of CH$_4$ production, or adequate means of estimating emissions. Although the current focus on CH$_4$ is as a greenhouse gas, CH$_4$ emitted from a system additionally represents a loss of energy (as reflected by the development of the original equations). In theory, reducing loss of CH$_4$ energy could provide increased energy for growth. However, the emission of both CH$_4$ and nitrogenous gas can be considered an inefficiency of the system, the reduction of which should have benefits for both productivity and economic performance. Accurate estimates may provide a tool for managers to improve efficiency of production as well as advising decisions on mitigation of emissions.

Chapter 4 evaluated a range of methodologies for the estimation of enteric CH$_4$ emissions, as well as examining the current prescribed model for predicting emissions of NH$_3$ and N$_2$O from livestock manure. Utilising the results of published studies based on feedlot cattle from (predominantly) the Northern hemisphere. The current Australian empirical equation for feedlots, Moe and Tyrrell (1979), was observed to have a relatively low (0.25) concordance between observed and predicted enteric CH$_4$ emissions. However, utilising an equation based on nutrient fractions improved predictive ability compared with equations based on energy consumption (concordance increase from 0.10 and 0.13 for the energy based equations to 0.25 and 0.28 for the nutrient fraction based rations). The results of Chapter 4 suggest issues with overestimation exist in the current methodology for feedlot cattle; however, as discussed by Mills et al. (2003) validation of a model for one location does not guarantee accuracy under different conditions, therefore the model needs to be tested under Australian commercial conditions.

Based on the results of Chapter 4, the two models for enteric CH$_4$ with the most potential in terms of accuracy of prediction were those based on carbohydrate fractions, Moe and Tyrrell (1979) and Ellis et al., (2007). However, the IPCC Tier II model (energetic based) shows potential particularly for higher grain rations and was therefore considered for potential application. The application of these models to data collected in Australian feedlots and measurements undertaken using micrometeorological methods will be discussed in this Chapter. Detailed climatic and animal production information was additionally collected, and reasons for differences between sites and seasons in measured CH$_4$, NH$_3$ and N$_2$O emissions will be evaluated.

An evaluation (Chapter 4) of the N partitioning and gaseous emissions estimated based on the current National Inventory Methodology (AGO 2006) suggested that while N excretion as a whole (combined urine and faecal N) is predicted relatively well (concordance 0.94, $R^2$ 0.85)
individual components of N partitioning in the animal (in particular faecal and retained N) are poorly estimated. Although this gives confidence that the amount of N being excreted by feedlot animals is estimated accurately, the evaluation process suggests that the use of fixed values to covert this figure to emissions of nitrogenous gases results in inaccuracies (concordance 0.27 for NH$_3$) related to the impact of environmental conditions. The 30% of excreted N figure is accurate at the lower range of measured emissions, but accuracy decreases as measured emissions increase.

5.1 Research Questions

This section aimed to build on the previous chapter and address the following questions utilising data and measured emissions from two commercial Australian feedlots.

1. Is the current Australian National Inventory Methodology for estimating greenhouse sources and sinks accurate in predicting emissions of CH$_4$, N$_2$O and NH$_3$ from feedlots?

2. Are equations based on energy or carbohydrate more accurate in estimating CH$_4$ emissions?

3. What is the impact of using a single emissions factor in the model to predict NH$_3$ and N$_2$O?

4. Does introducing equations developed based on beef cattle data improve accuracy of estimates compared to measured values of CH$_4$ emissions compared with the current approaches?

5. Which factors contribute to differences in measured emissions at each site and between seasons

5.2 Methodology

Full details of the feedlots studied can be found in the General Methodology Section 3.1. Briefly, micrometeorological measurements of greenhouse gas emissions were undertaken at two Australian feedlots during two seasons over two years. Details of the micrometeorological measurement methods can be found in the General Methodology, Section 3.2.

The objective of this modelling approach was to utilise data available from commercial feedlots. Therefore during the measurement campaigns information was collected from the feedlot operators in the form of bunk sheets, lot sheets, rations and daily feed amounts. This information was collected for a minimum of five days for each 14 day measurement
campaign. Information collected included number of head (cattle) in each pen, animal sex, estimated weight on entry and at the report date, days on feed, cattle class, rations and amounts of feed offered. This information is easily available using feedlot management software, and is collected and maintained by commercial feedlots on a routine basis. Therefore the ability to use this type of data to estimate emissions will enable estimates to be made simply for an individual operation.

The collected data was utilised as input for the model outlined in the general methodology. Three of the described equations for estimation of enteric CH$_4$ (Ellis et al. 2007; Moe and Tyrrell 1979 and IPCC Tier II) were utilised in the application of the model to this data, based on the results of Chapter 4.

5.2.1 Assumptions

Standardised inputs (Table 5.1) were utilised for the model (where appropriate) for both feedlot sites, and where these values are specified by the National Inventory Methodology (AGO 2006). Standard values are commonly used for feed composition where factors are not commonly measured, such as GE (energy content of ruminant rations is more commonly expressed as ME or net energy NE), soluble residue, cellulose and hemicellulose. Where required dietary parameters (e.g. lignin) were not given as standard values this information was taken from the appropriate feed type as reported by Givens and Moss (1990).

Although more detailed information regarding intakes, live weight gain (forecast) and current weights was available from the feedlot operators than would be used in producing inventory calculations animal class was required to be translated from the denominator used by the operators to a numerical representative of class. This was based on information provided by the feedlot management.

Inventory guidelines (AGO 2006) suggest intakes (kgDM/head) of 2 to 2.4% of live weight (dependent on animal class). Where intake/ offered feed data provided by the feedlot operators indicated intakes of greater than 3% of live weight these values were replaced by intakes predicted based on percentage live weight. Intakes higher than physically possible (due to large amounts of offered feed on a given day) result in artificially inflated emissions predictions, which are unlikely to occur in practice. Further, where no offered feed or very low offered feed (less than one kgDM/head) were reported these values were replaced by intake as a percentage of live weight. Intake in ruminant animals is limited by physical capacity of the rumen. It is unlikely that feedlot cattle will consume feed (even offered ad lib) greater than 3% of live weight. Holt et al. (2004) formulated ad lib grain rations for feedlot
cattle (starting at 324 kg LWT) at 2.6% LWT. Similarly, Hicks et al. (1990) measured intakes of feedlot cattle arriving in a feed yard, measuring DM intake weekly from entry to 150 days on feed, DMI as a percentage of live weight ranged from 2.7% at the start of feeding, to 1.6% during the finishing period. Maximum intakes were observed during the middle of the feeding period (approximately days 40 to 80), however based on growth rates provided DMI during this period was close to 2.5% LWT. These estimates support those reported in the National Inventory Methodology (AGO 2006) and by van Sliedregt et al. (2000).

**Table 5.1** Standardised values utilised in the prediction of greenhouse gas emissions from beef feedlot cattle.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Set Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Standard Reference Weight kg</td>
<td>660</td>
</tr>
<tr>
<td>GE content of rations MJ/kg DM</td>
<td>18.4</td>
</tr>
<tr>
<td>Digestibility of rations %</td>
<td>80</td>
</tr>
<tr>
<td>N₂O Emission Factor</td>
<td>0.02</td>
</tr>
<tr>
<td>NH₃ Proportion of Excreted N</td>
<td>0.3</td>
</tr>
<tr>
<td>Manure CH₄ conversion factor- temperate</td>
<td>0.015</td>
</tr>
<tr>
<td>Manure CH₄ conversion factor- hot</td>
<td>0.05</td>
</tr>
<tr>
<td>Soluble Residue* g/kg DM</td>
<td></td>
</tr>
<tr>
<td>Grains</td>
<td>0.68</td>
</tr>
<tr>
<td>Concentrates</td>
<td>0.19</td>
</tr>
<tr>
<td>Forages</td>
<td>0.21</td>
</tr>
<tr>
<td>Cellulose* g/kg DM</td>
<td></td>
</tr>
<tr>
<td>Grains</td>
<td>0.07</td>
</tr>
<tr>
<td>Concentrates</td>
<td>0.19</td>
</tr>
<tr>
<td>Forages</td>
<td>0.31</td>
</tr>
<tr>
<td>Hemi- Cellulose* g/kg DM</td>
<td></td>
</tr>
<tr>
<td>Grains</td>
<td>0.31</td>
</tr>
<tr>
<td>Concentrates</td>
<td>0.31</td>
</tr>
<tr>
<td>Forages</td>
<td>0.21</td>
</tr>
</tbody>
</table>

* Based on AFIC (1987)

Removing offered feed amounts larger than what can physically be consumed by a feedlot animal removes artificially inflated emissions- which do not represent the biology of the system. Additionally, feedlot cattle are managed to produce maximum growth, therefore the offering feed of less than 1 kg/head, less than 0.2% LWT for a 500 kg steer, would not be
indicative of the amount of feed offered (as reported in “bunk” or “lot” sheets) to support growth in these animals. It is likely that small amounts of feed were offered in these cases to provide a small top up from larger amounts offered in the previous days and do not reflect the actual amount consumed by these animals. Intake as a percentage of live weight was found in the previous Chapter (4) to have a moderate concordance (compared with a set value for live weight/class and the equation of Minson and McDonald (1987)); however this was selected for simplicity.

Metabolisable energy (ME) content can be estimated based on diet digestibility, with 80% digestibility equating to approximately 12 MJ ME/kg DM. Overall digestibility of the rations used in the feedlots was not available in the rations composition provided, therefore the standard value of 80% was used, however ME provided in the ration formulation was used in the calculations where required.

5.2.2 Emissions from Manure

Measurements using micrometeorological methods do not enable emissions of CH$_4$ to be partitioned between animals, or between enteric and manure pad emissions. Modelled estimates and results of Boadi et al (2004b) suggest that CH$_4$ emissions from the manure pad equate to less than 2 g/head/day, therefore emissions measured using micrometeorological methods can be attributed almost entirely to enteric emissions. For the purpose of this evaluation CH$_4$ measured using micrometeorological methods was assumed to reflect primarily enteric CH$_4$, and used in analysis as such (manure excluded from estimated CH$_4$).

The primary source of NH$_3$ and N$_2$O in the feedlot is the pen/manure surface and the effluent management system. The software used in calculating emissions using wind statistics and concentration measurements accounts for the effluent ponds as a source and discounts periods where wind statistics indicate the measured concentration is derived primarily from this area.

There is some evidence of N$_2$O production in the rumen (depending on dietary composition; Kaspar and Tiedje 1981); however this is unlikely to make a significant contribution of total measured emissions. There is also a possibility of NH$_3$ and N$_2$O production through fermentation and microbial reactions in feedstuffs, particularly if these become wet (through rainfall) or remain sitting in the feed bunk for long periods. Where nitrite or NO$_3$ is present in grass N$_2$O can be formed during ensiling, however a less acidic (low soluble carbohydrate) silage will show increased conversion of NO$_3$ to ammonium (NH$_4$; Hill 1999). The formation of anaerobic pockets in distributed feeds may result in further production of N$_2$O or NO$_3$; although this is unlikely to be as significant a source as manure decomposition. An emission
factor for N$_2$O release from grass silage of 15g N$_2$O-N/ kg NO$_3$-N (Hill 1999), however this represents the ensiling process, not post ensiling fermentation. For the purposes of this analysis measured emissions of NH$_3$ and N$_2$O are considered to be derived from the manure pad.

5.2.3 Statistical Analysis

Details of the bLS approach used in the measurement and calculation of fluxes/emissions in g/head/day are described in the general methodology (Chapter 3). Average emissions estimated for each of the three gases (CH$_4$, N$_2$O and NH$_3$) was calculated for each day of data collected from each measurement campaign. Average estimates reflect the average over the useable data collected for the entire measurement campaign. Standard deviations reflect the variation over the entire measurement campaign. Linear regression (Genstat v. 11 VSN international) was used to examine relationships between measured and predicted emissions. Significance represents the fit of a linear equation to the data, the probability that correlation is greater than 0. Lin’s concordance (Genstat v. 11 VSN international) was used to measure how well the modelled data reproduces the original data set (Lin 1989; Lin 2000). The concordance coefficient is calculated using a simple Pearson’s correlation coefficient (reported in text as the correlation between the variables) and the slope and origin of the line, known as $C_b$.

5.2.4 Input Data

Model input was derived from information collected from feedlot operators during the measurement campaigns (Table 3.1). An objective of this approach was to utilise information readily available from commercial feedlots, therefore data was collected from feedlot operators in the form of standard outputs from management software. The Northern feedlot had a capacity of 17,000 and was operating between maximum capacity and approximately 1/3 capacity during the four campaigns. The Southern feedlot had a maximum capacity of 20,000 head, and was operating between 12,000 and 18,000 head during the four measurement campaigns. Total number of head was higher in all campaigns at the Southern Site than at the Northern Site.

In contrast to experimental data, where live weight, live weight gain and feed intakes are recorded for individual animals, the information provided by the feedlot operators is on a pen or “lot” (cattle group) basis. Live weight is estimated for a group of cattle on arrival (based on the weight of the consignment of cattle (in the transport vehicle) and used to determine an average start weight. Live weight is then computed from number of days on feed, and
estimated live weight gain (for class, offered feed amount and ration type); very few cattle are weighed individually. Average live weight (Table 5.2) was higher and more variable at the Northern Site. Estimated daily live weight gain and offered feed were consistently higher at the Northern Site. This is related to the differing market targets of the two sites, which long fed export cattle being the primary output of the Northern Site compared with shorter fed domestic class cattle at the Southern Site.

Basic environmental conditions (Table 5.3) were measured at each site (including detailed wind statistics). Temperatures showed typical seasonal variations at both sites. However, there was more variation between summer and winter minimum temperatures at the Southern Site than the Northern Site. Rainfall was substantially higher during the final two measurement campaigns at the Northern Site, than for the remainder of the measurement periods. Mean wind speed (km/h) also tended to be higher in summer at both sites, with consistently higher “gust” speed during summers at the Northern site. Overall average wind speed was higher at the Northern site.
Table 5.2 Feedlot stock characteristics (mean and SD) averaged over the duration of eight measurement campaigns. Data was collected from feedlot operators in the form of standard management software outputs.

<table>
<thead>
<tr>
<th>Campaign</th>
<th>Number of Head</th>
<th>Live weight* (Kg)</th>
<th>Estimated Live weight gain (kg/day)</th>
<th>Offered Feed (kgDM/head/day)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Southern Summer 2008</td>
<td>12863</td>
<td>455 ± 68.2</td>
<td>1.3 ± 0.33</td>
<td>9.7 ± 1.89</td>
</tr>
<tr>
<td>Southern Winter 2007</td>
<td>13107</td>
<td>441 ± 91.5</td>
<td>1.3 ± 0.35</td>
<td>9.4 ± 1.76</td>
</tr>
<tr>
<td>Southern Summer 2007</td>
<td>16593</td>
<td>519 ± 101.0</td>
<td>1.4 ± 0.37</td>
<td>10.1 ± 1.96</td>
</tr>
<tr>
<td>Southern Winter 2006</td>
<td>18137</td>
<td>492 ± 93.5</td>
<td>1.3 ± 0.38</td>
<td>10.1 ± 1.64</td>
</tr>
<tr>
<td>Northern Summer 2008</td>
<td>6167</td>
<td>756 ± 132.8</td>
<td>1.6 ± 0.46</td>
<td>11.9 ± 2.04</td>
</tr>
<tr>
<td>Northern Winter 2007</td>
<td>10554</td>
<td>530 ± 117.8</td>
<td>1.8 ± 0.23</td>
<td>11.3 ± 1.95</td>
</tr>
<tr>
<td>Northern Summer 2007</td>
<td>13747</td>
<td>562 ± 118.3</td>
<td>1.8 ± 1.00</td>
<td>10.4 ± 1.81</td>
</tr>
<tr>
<td>Northern Winter 2006</td>
<td>16655</td>
<td>546 ± 214.5</td>
<td>1.9 ± 1.62</td>
<td>10.3 ± 2.33</td>
</tr>
</tbody>
</table>

*Current live weight is estimated based on entry weight, days on feed and estimated live weight gain (based on class, ration and feed offered).
Table 5.3 Environmental conditions at two feedlot sites (northern and southern Australia) during eight measurement campaigns (covering summer and winter)

<table>
<thead>
<tr>
<th></th>
<th>Average Daily Temperature (°C)</th>
<th>Rainfall (mm)</th>
<th>Average Daily Wind Speed (km/h)</th>
<th>Average Daily Relative Humidity (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Min</td>
<td>Max</td>
<td>Total</td>
<td>Mean</td>
</tr>
<tr>
<td>Southern Summer 2008</td>
<td>12.0</td>
<td>26.7</td>
<td>0.2</td>
<td>10.8</td>
</tr>
<tr>
<td>Southern Winter 2007</td>
<td>4.7</td>
<td>19.7</td>
<td>4.4</td>
<td>4.2</td>
</tr>
<tr>
<td>Southern Summer 2007</td>
<td>19.1</td>
<td>34.4</td>
<td>7.4</td>
<td>9.2</td>
</tr>
<tr>
<td>Southern Winter 2006</td>
<td>3.3</td>
<td>15.2</td>
<td>2.0</td>
<td>6.7</td>
</tr>
<tr>
<td>Northern Summer 2008</td>
<td>19.0</td>
<td>29.7</td>
<td>61.4</td>
<td>11.8</td>
</tr>
<tr>
<td>Northern Winter 2007</td>
<td>10.4</td>
<td>18.9</td>
<td>23.6</td>
<td>7.4</td>
</tr>
<tr>
<td>Northern Summer 2007</td>
<td>17.9</td>
<td>31.8</td>
<td>0.0</td>
<td>12.6</td>
</tr>
<tr>
<td>Northern Winter 2006</td>
<td>12.3</td>
<td>25.3</td>
<td>4.4</td>
<td>8.1</td>
</tr>
</tbody>
</table>
Measured CH₄ emissions (Table 5.4) ranged from 63.8 g/head/day (Northern Site, summer 2008) to 138.8 g/head/day (Northern Site, winter 2008). In general, emissions were slightly lower at the Southern Site (91 g/head/day to 127.8 g/head/day). N₂O emissions ranged from 0.1 to 5.7 g/head/day, whilst NH₃ emissions ranged from 94 to 324.4 g/head/day. There did not appear to be any distinct seasonal effects on emissions at either site.

The significantly lower CH₄ emission reported during the Summer 2008 measurement campaign (Table 5.4) at the Northern Site and higher standard errors reported for emissions of all gases during this measurement campaign indicates a possible issue with the measurement method during this campaign, rather than a real difference in emission rate. A further examination of the number of observations contributing to these measured emissions indicate only 34 15 minute time points with usable measurements (based on the filtering criteria for the bLS method, wind speed and turbulence statistics) for CH₄ and 51 (15 minute time points) for NH₃. The environmental conditions during the summer 2008 measurement campaign (Table 5.3) are not substantially different in terms of average wind speed, gust speed, temperature or humidity. However, rainfall was significantly higher during this campaign than others, which can affect measurements, for example reducing NH₃ through deposition. The lower number of observations is reflected in the variability reported (standard errors), which are more than 20 times greater than standard errors for other measurement campaigns for N₂O, and up to double standard deviations for NH₃ for the other campaigns (Table 5.4).

**Table 5.4** Means and standard errors of per-head emissions (g/head/day) of CH₄, NH₃ and N₂O during eight field campaigns. Measurements of CH₄ and NH₃ were made by three separate instruments (two open-path lasers and an open-path FTIR, combined values shown). N₂O measurements were made by open-path FTIR only.

<table>
<thead>
<tr>
<th></th>
<th>CH₄</th>
<th>NH₃</th>
<th>N₂O</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Mean</td>
<td>SE</td>
<td>Mean</td>
</tr>
<tr>
<td>Southern Summer 2008</td>
<td>91.0</td>
<td>1.5</td>
<td>102.0</td>
</tr>
<tr>
<td>Southern Winter 2007</td>
<td>122.8</td>
<td>1.6</td>
<td>305.1</td>
</tr>
<tr>
<td>Southern Summer 2007</td>
<td>127.8</td>
<td>3.2</td>
<td>153.0</td>
</tr>
<tr>
<td>Southern Winter 2006</td>
<td>98.9</td>
<td>2.8</td>
<td>151.4</td>
</tr>
<tr>
<td>Northern Summer 2008</td>
<td>63.8</td>
<td>14.8</td>
<td>324.4</td>
</tr>
<tr>
<td>Northern Winter 2007</td>
<td>138.8</td>
<td>3.7</td>
<td>94.0</td>
</tr>
<tr>
<td>Northern Summer 2007</td>
<td>127.4</td>
<td>3.0</td>
<td>133.2</td>
</tr>
<tr>
<td>Northern Winter 2006</td>
<td>131.5</td>
<td>4.7</td>
<td>143.1</td>
</tr>
</tbody>
</table>
A detailed investigation of animal behaviour and interactions with emissions is undertaken in Chapter 6. Diurnal emissions patterns observed during this study demonstrate peaks in emissions of NH$_3$ at around midday. CH$_4$ emissions tended to be lowest during the late morning and early afternoon, corresponding with feeding and high temperatures (decreasing rumination). However, for both NH$_3$ and CH$_4$ measured during this campaign (Northern Summer 2008) the periods with measured emissions occurred primarily between 9 am and 4 pm, capturing the peak emissions from NH$_3$, and lows in emission of CH$_4$. This is likely to lead to higher calculated emissions for NH$_3$, and lower for CH$_4$. For these reasons, the measured emissions from the Northern Summer 2008 campaign were not utilised in this evaluation.

Detailed ration information (Table 5.5) was also recorded at each site during each measurement campaign. At the Southern Site, barley and wheat are the most commonly used grains, whilst sorghum forms the basis of the Northern Site rations. Both sites use silages (whole crop sorghum silage at the Northern Site and grass silage at the Southern Site) and dry forages (hay and straw) and some by-product feeds (including bread crumbs). There is a greater use of cotton seed and processed products of cotton seed (meal and hulls) at the Northern Site. Proportion of grain tends to be higher on average at the Northern Site (61 to 71% at the Southern Site, compared with 71 to 81% at the Northern Site). Grain concentration tended to be more variable at the Southern Site (particularly during 2007) than the Northern Site. The Southern Site also reported the use of a hay only ration for newly arrived cattle transitioning to the feedlot.

Excluding the hay only ration (which is fed to a very limited number of cattle over short time frames) average DM (Table 4.6) was slightly higher (73% cf. 70%) at the Southern Site, as was CP % (14.0 % cf. 13.6%, ME (MJ/ kg DM, 12.2 compared with 12.0) and average fat content (kg/kg DM). At both sites, fat (recycled oil or vegetable oil) was included in a limited number of rations (approximately two at each site during each campaign).
Table 5.5 Major ration Ingredients and proportions of grain at two feedlots sites over 8 measurement campaigns

<table>
<thead>
<tr>
<th></th>
<th>Grains</th>
<th>Forages</th>
<th>Additives</th>
<th>Proportion Grain</th>
</tr>
</thead>
<tbody>
<tr>
<td>Southern Summer 2008</td>
<td>Wheat</td>
<td>Silage</td>
<td>Molasses</td>
<td>0 Hay only</td>
</tr>
<tr>
<td></td>
<td>Barley</td>
<td>Grass Hay</td>
<td>Liquid Starter</td>
<td>62 to 78% Mixed</td>
</tr>
<tr>
<td></td>
<td>Whole Cotton</td>
<td>Bread Crumbs</td>
<td>Liquid Finisher</td>
<td>70% average</td>
</tr>
<tr>
<td></td>
<td>Seed</td>
<td></td>
<td>Vegetable Oil</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Molasses</td>
<td></td>
</tr>
<tr>
<td>Southern Winter 2007</td>
<td>Wheat</td>
<td>Grass Hay</td>
<td>Liquid Starter</td>
<td>24 to 80% Mixed</td>
</tr>
<tr>
<td></td>
<td>Whole Cotton</td>
<td>Silage</td>
<td>Liquid Finisher</td>
<td>61% Average</td>
</tr>
<tr>
<td></td>
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<td>Bread Crumbs</td>
<td>Vegetable Oil</td>
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<td>Liquid Starter</td>
<td>24 to 80% Mixed</td>
</tr>
<tr>
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<td>Bread Crumbs</td>
<td>Liquid Finisher</td>
<td>61% Average</td>
</tr>
<tr>
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<td>Seed</td>
<td></td>
<td>Vegetable Oil</td>
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<td></td>
<td></td>
<td>Molasses</td>
<td></td>
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<tr>
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<td>Wheat</td>
<td>Grass Hay</td>
<td>Liquid Starter</td>
<td>0 Hay only</td>
</tr>
<tr>
<td></td>
<td>Barley</td>
<td>Silage</td>
<td>Liquid Finisher</td>
<td>62 to 82 % Mixed</td>
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<tr>
<td></td>
<td>Canola Meal</td>
<td>Oaten Hay</td>
<td>Vegetable Oil</td>
<td>71 % Ave Mixed</td>
</tr>
<tr>
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<td>Whole Cotton</td>
<td></td>
<td>Molasses</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Seed</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Northern Summer 2008</td>
<td>Barley</td>
<td>Straw</td>
<td>Liquid Supplement</td>
<td>51 to 88% Mixed</td>
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<tr>
<td></td>
<td>Sorghum</td>
<td>Cotton Hulls</td>
<td>Starter Supplement</td>
<td>71% Ave Mixed</td>
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<td></td>
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<td>Silage</td>
<td>Recycled Oil</td>
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<td>Hay</td>
<td>Cotton Hulls</td>
<td>Liquid Supplement</td>
</tr>
<tr>
<td>---------------------</td>
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</tr>
<tr>
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<td>Cotton Seed Meal Sorghum Whole Cotton Seed</td>
<td>Sorghum Silage Straw</td>
<td>Sorghum Cotton Seed Meal Whole Cotton Seed</td>
<td>Liquid Supplement</td>
</tr>
<tr>
<td></td>
<td>Wheat Maize</td>
<td></td>
<td>Wheat Hay</td>
<td>Liquid Finisher</td>
</tr>
<tr>
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<td>Cotton Seed Meal Sorghum Whole Cotton Seed</td>
<td>Wheat Straw Liquid Finisher</td>
<td>Liquid Starter Recycled Oil</td>
<td>79% Ave Mixed</td>
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Table 5.6 Ration composition and nutritive values required for predicting enteric CH₄ (using five equations), manure CH₄, NH₃ and N₂O emissions from feedlot beef cattle.

<table>
<thead>
<tr>
<th>Ration</th>
<th>DM %</th>
<th>CP %</th>
<th>ME MJ/kg DM</th>
<th>Soluble Residue kg/kg DM</th>
<th>Cellulose kg/kg DM</th>
<th>Hemicellulose kg/kg DM</th>
<th>ADF kg/kg DM</th>
<th>Lignin kg/kg DM</th>
<th>Fat kg/kg DM</th>
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<td></td>
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<td></td>
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<td></td>
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<td>77.0</td>
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<td>11.0</td>
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<td>0.154</td>
<td>0.147</td>
<td>0.029</td>
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<td>0.138</td>
<td>0.127</td>
<td>0.027</td>
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<td>14.0</td>
<td>12.1</td>
<td>0.553</td>
<td>0.100</td>
<td>0.126</td>
<td>0.112</td>
<td>0.025</td>
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<td>12.8</td>
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<td>0.310</td>
<td>0.367</td>
<td>0.060</td>
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<td>0.310</td>
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<td>11.9</td>
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<td>12.8</td>
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<td>13.0</td>
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<td>0.057</td>
<td>0.083</td>
<td>0.090</td>
<td>0.029</td>
</tr>
<tr>
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<td>2</td>
<td>74.8</td>
<td>13.7</td>
<td>10.9</td>
<td>0.575</td>
<td>0.082</td>
<td>0.113</td>
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<td>11.9</td>
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<td>0.100</td>
<td>0.114</td>
<td>0.037</td>
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<td>0.608</td>
<td>0.057</td>
<td>0.083</td>
<td>0.090</td>
<td>0.029</td>
</tr>
<tr>
<td>Northern Winter 2006</td>
<td>2</td>
<td>73.3</td>
<td>13.9</td>
<td>11.4</td>
<td>0.558</td>
<td>0.067</td>
<td>0.091</td>
<td>0.139</td>
<td>0.040</td>
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<td>13.8</td>
<td>11.4</td>
<td>0.577</td>
<td>0.056</td>
<td>0.081</td>
<td>0.120</td>
<td>0.037</td>
</tr>
</tbody>
</table>
Chapter 5. Model Application

<table>
<thead>
<tr>
<th></th>
<th>DM%</th>
<th>CP%</th>
<th>MEcontent</th>
<th>ADF</th>
<th>Lignin</th>
<th>Fat</th>
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<td>11.6</td>
<td>0.440</td>
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<td>12.9</td>
<td>0.594</td>
<td>0.053</td>
<td>0.079</td>
</tr>
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<td>71.9</td>
<td>13.7</td>
<td>12.8</td>
<td>0.617</td>
<td>0.051</td>
<td>0.078</td>
</tr>
</tbody>
</table>

DM %, CP% and ME content were taken from feedlot ration information. Soluble residue, cellulose and hemicellulose were calculated (using weighted averages) based on ration concentration of roughage and concentrate using standard values outlined in the methodology (refer to section. Concentrations of ADF, lignin and fat were calculated as weighted averages based on proportions of individual feeds in the ration and chemical compositions reported by (Givens and Moss 1990).
Chapter 5. Model Application

5.3 Results

5.3.1 Accuracy of Prediction

A visual examination of measured and predicted emissions (Figure 5.1) suggests the best estimate of enteric CH\(_4\) emissions is made using the Tier II equation. NH\(_3\) (Figure 5.2) appears to be estimated well by the model at lower levels of measured emission with accuracy decreasing as measured emission increases. In contrast to NH\(_3\), accuracy of prediction of N\(_2\)O emission decreases with decreased measured emissions, further N\(_2\)O is consistently overestimated by the model (whilst NH\(_3\) is consistently underestimated by the model, Figure 5.2). No significant (P>0.05) linear relationship could be determined between measured and predicted enteric CH\(_4\) emissions (Table 5.7). A significant linear relationship was determined between measured and predicted emissions of N\(_2\)O (P<0.05).

Table 5.7 Fitted linear relationships and SE for measured and predicted greenhouse gas emissions at two feedlot sites over seven measurement campaigns

<table>
<thead>
<tr>
<th>Equation</th>
<th>y-intercept</th>
<th>Slope</th>
<th>Significance</th>
<th>SE Obs</th>
<th>SE Pred</th>
</tr>
</thead>
<tbody>
<tr>
<td>Moe and Tyrell (1979)</td>
<td>177.9</td>
<td>0.05</td>
<td>P=0.879</td>
<td>13.3</td>
<td>5.0</td>
</tr>
<tr>
<td>Tier II</td>
<td>77.5</td>
<td>0.12</td>
<td>P=0.165</td>
<td>5.23</td>
<td>2.0</td>
</tr>
<tr>
<td>Ellis et al. (2007)</td>
<td>186.4</td>
<td>0.27</td>
<td>P=0.160</td>
<td>7.2</td>
<td>2.7</td>
</tr>
<tr>
<td>N(_2)O</td>
<td>0.016</td>
<td>0.15</td>
<td>P=0.048</td>
<td>0.3</td>
<td>0.1</td>
</tr>
<tr>
<td>NH(_3)</td>
<td>64.46</td>
<td>-0.02</td>
<td>P=0.465</td>
<td>3.84</td>
<td>1.5</td>
</tr>
</tbody>
</table>

Lin’s concordance (Lin 1989, 2000) demonstrates how well a “new” set of data is able to reproduce another data set. The highest concordance between measured and predicted data (for the 7 measurement campaigns) was observed using the Tier II equation (Table 5.8). Although this equation does not demonstrate a strong correlation (0.59), the value of Cb is considerably higher than for the remainder of the enteric CH\(_4\) prediction equations. This indicates close association between the slope and origin of the “old” and “new” data. Low to moderate correlations were observed for the other enteric CH\(_4\) equations. Low concordance was observed for both N\(_2\)O and NH\(_3\). A strong correlation was observed between measured and predicted values for N\(_2\)O (0.76) reflective of the significant relationship. The low correlation and concordance determined for NH\(_3\) is indicative of the significant underestimation in predicted values at high levels of measured NH\(_3\) (Figure 5.2). Similarly the low concordance between measured and predicted N\(_2\)O (Table 5.8) reflects the overestimation at lower levels of measured N\(_2\)O (g/head/day) (Figure 5.2).
Table 5.8 Concordance and correlations between measured and predicted greenhouse gas emissions at two feedlot sites over eight measurement campaigns

<table>
<thead>
<tr>
<th>Equation</th>
<th>Concordance</th>
<th>Correlation</th>
<th>Cb</th>
</tr>
</thead>
<tbody>
<tr>
<td>Moe and Tyrell (1979)</td>
<td>0.01</td>
<td>0.07</td>
<td>0.09</td>
</tr>
<tr>
<td>Tier II</td>
<td>0.18</td>
<td>0.59</td>
<td>0.30</td>
</tr>
<tr>
<td>Ellis et al. (2007)</td>
<td>0.02</td>
<td>0.59</td>
<td>0.03</td>
</tr>
<tr>
<td>N₂O</td>
<td>0.07</td>
<td>0.76</td>
<td>0.10</td>
</tr>
<tr>
<td>NH₃</td>
<td>-0.01</td>
<td>-0.33</td>
<td>0.04</td>
</tr>
</tbody>
</table>
Figure 5.1 Comparison between measured and predicted emissions of enteric CH$_4$ from two Australian feedlot sites using three equations; IPCC Tier II (A), Moe and Tyrrell (1979) (B) and Ellis et al. (2007) (C). Horizontal error bars indicate SD of measured emissions (averaged over the whole measurement campaign); vertical error bars indicate SD of predicted emissions. (averaged over the whole measurement campaign).
5.3.2 Southern Site

The differences in production systems between Northern and Southern Sites including animal live weight, growth rates and rations plus differences in manure management and climatic conditions are likely to result in differences in greenhouse gas emissions. When predicted and measured emissions were compared for a single feedlot site (Figures 5.3 and 5.4 compared with Figures 5.5 and 5.6), similar trends were observed to the overall (Figure 5.1 and 5.2). No significant \( P>0.05 \) linear relationships were determined between measured and predicted \( \text{N}_2\text{O} \) or \( \text{NH}_3 \) emissions.

**Figure 5.2** Comparison between measured and predicted emission of \( \text{NH}_3 \) (A) and \( \text{N}_2\text{O} \) (B) from two Australian feedlot sites. Horizontal error bars indicate SD of measured emissions (averaged over the whole measurement campaign); vertical error bars indicate SD of predicted emissions. (averaged over the whole measurement campaign).
Table 5.9 Fitted linear relationships and SE for predicted and measured greenhouse gas emissions from a Southern Australian feedlot site over four measurement campaigns

<table>
<thead>
<tr>
<th>Equation</th>
<th>y-intercept</th>
<th>Slope</th>
<th>Significance</th>
<th>SE Obs</th>
<th>SE Pred</th>
</tr>
</thead>
<tbody>
<tr>
<td>Moe and Tyrell (1979)</td>
<td>213.2</td>
<td>-0.29</td>
<td><em>P=0.499</em></td>
<td>11.0</td>
<td>5.5</td>
</tr>
<tr>
<td>Tier II</td>
<td>95.3</td>
<td>0.02</td>
<td><em>P=0.889</em></td>
<td>4.4</td>
<td>2.2</td>
</tr>
<tr>
<td>Ellis et al. (2007)</td>
<td>202.8</td>
<td>0.12</td>
<td><em>P=0.682</em></td>
<td>7.6</td>
<td>3.8</td>
</tr>
<tr>
<td>N₂O</td>
<td>6.2</td>
<td>0.11</td>
<td><em>P=0.406</em></td>
<td>0.4</td>
<td>0.2</td>
</tr>
<tr>
<td>NH₃</td>
<td>62.7</td>
<td>-0.01</td>
<td><em>P=0.815</em></td>
<td>4.7</td>
<td>2.3</td>
</tr>
</tbody>
</table>

Concordance (Table 5.10) between all equations and measured emissions were very low, however a moderate correlation was observed between measured CH₄ emissions and predictions based on Moe and Tyrell (1979). Consistent with the overall analysis C_b was highest for estimated based on Tier II. Concordance between measured and predicted values for both N₂O and NH₃ are lower at the Southern Site compared with overall estimates, as is correlation between measured and predicted N₂O (0.59) compared with (0.76, overall correlation).

Table 5.10 Concordance and correlations between predicted and measured greenhouse gas emissions from a Southern Australian feedlot site over four measurement campaigns

<table>
<thead>
<tr>
<th>Equation</th>
<th>Concordance</th>
<th>Correlation</th>
<th>C_b</th>
</tr>
</thead>
<tbody>
<tr>
<td>Moe and Tyrell (1979)</td>
<td>-0.03</td>
<td>-0.50</td>
<td>0.07</td>
</tr>
<tr>
<td>Tier II</td>
<td>0.03</td>
<td>0.11</td>
<td>0.26</td>
</tr>
<tr>
<td>Ellis et al. (2007)</td>
<td>0.01</td>
<td>0.22</td>
<td>0.02</td>
</tr>
<tr>
<td>N₂O</td>
<td>0.05</td>
<td>0.59</td>
<td>0.09</td>
</tr>
<tr>
<td>NH₃</td>
<td>-0.01</td>
<td>-0.19</td>
<td>0.03</td>
</tr>
</tbody>
</table>
Figure 5.3 Comparison between measured and predicted emissions of enteric CH₄ using IPCC Tier II (A), (Moe and Tyrrell 1979) (B) and (Ellis et al. 2007) (C) from a southern Australian feedlot site. Horizontal error bars indicate SD of measured emissions (averaged over the whole measurement campaign); vertical error bars indicate SD of predicted emissions (averaged over the whole measurement campaign).
Figure 5.4 Comparison between measured and predicted emissions of NH$_3$ (A) and N$_2$O (B) from a southern Australian feedlot site. Horizontal error bars indicate SD of measured emissions (averaged over the whole measurement campaign); vertical error bars indicate SD of predicted emissions. (averaged over the whole measurement campaign).

5.3.3 Northern Site

No significant ($P>0.05$) linear relationships (Table 5.11) were observed between measured and predicted emissions of enteric CH$_4$ at the Northern Site (Figure 5.5 and 5.6). The linear relationship between measured and predicted emission of N$_2$O approached significance ($P=0.071$).
Table 5.11 Fitted linear relationships and SE for predicted and measured greenhouse gas emissions from a Northern Australian feedlot site over four measurement campaigns.

<table>
<thead>
<tr>
<th>Equation</th>
<th>y-intercept</th>
<th>Slope</th>
<th>Significance</th>
<th>SE Obs</th>
<th>SE Pred</th>
</tr>
</thead>
<tbody>
<tr>
<td>Moe and Tyrell (1979)</td>
<td>-98.0</td>
<td>2.16</td>
<td>P=0.460</td>
<td>14.8</td>
<td>8.5</td>
</tr>
<tr>
<td>Tier II</td>
<td>-26.2</td>
<td>0.99</td>
<td>P=0.293</td>
<td>3.8</td>
<td>2.2</td>
</tr>
<tr>
<td>Ellis et al. (2007)</td>
<td>16.9</td>
<td>1.56</td>
<td>P=0.154</td>
<td>3.0</td>
<td>1.7</td>
</tr>
<tr>
<td>N₂O</td>
<td>5.8</td>
<td>0.21</td>
<td>P=0.131</td>
<td>0.13</td>
<td>0.1</td>
</tr>
<tr>
<td>NH₃</td>
<td>82.3</td>
<td>2.29</td>
<td>P=0.071</td>
<td>0.66</td>
<td>0.4</td>
</tr>
</tbody>
</table>

The highest concordance for measured and predicted (Table 5.12) enteric CH₄ emissions was observed for the Tier II equation (0.07) consistent with the lowest residuals for this equation. The strongest correlations between measured and predicted CH₄ emissions were observed for the Ellis et al. (2007) (0.97) and Tier II equations (0.90), although Moe and Tyrrell (1979) also showed a moderate correlation (0.75). Correlation between measured and predicted N₂O emissions was higher at the Northern Site (Table 5.12) than the Southern Site as was the correlation between measured and predicted emissions of NH₃.

Table 5.12 Concordance and correlations between measured and predicted greenhouse gas emissions from a Northern Australian feedlot site over four measurement campaigns.

<table>
<thead>
<tr>
<th>Equation</th>
<th>Concordance</th>
<th>Correlation</th>
<th>Cb</th>
</tr>
</thead>
<tbody>
<tr>
<td>Moe and Tyrell (1979)</td>
<td>0.04</td>
<td>0.75</td>
<td>0.05</td>
</tr>
<tr>
<td>Tier II</td>
<td>0.07</td>
<td>0.90</td>
<td>0.08</td>
</tr>
<tr>
<td>Ellis et al. (2007)</td>
<td>0.01</td>
<td>0.97</td>
<td>0.01</td>
</tr>
<tr>
<td>N₂O</td>
<td>0.14</td>
<td>0.98</td>
<td>0.14</td>
</tr>
<tr>
<td>NH₃</td>
<td>-0.05</td>
<td>-0.99</td>
<td>0.05</td>
</tr>
</tbody>
</table>
Chapter 5. Model Application

Figure 5.5 Comparison between measured and predicted emissions of enteric CH$_4$ using IPCC Tier II (A), Moe and Tyrrell (1979) (B) and Ellis et al. (2007) (C) from a northern Australian feedlot site. Horizontal error bars indicate SD of measured emissions (averaged over the whole measurement campaign); vertical error bars indicate SD of predicted emissions. (averaged over the whole measurement campaign).
5.3.4 Seasonal Differences

During both summer and winter, no significant \( P > 0.05 \) linear relationships could be determined between measured and predicted \( \text{CH}_4 \) emissions for the range of data evaluated. During winter, significant linear relationships \( P < 0.05 \) were determined between measured and predicted values of \( \text{NH}_3 \) and \( \text{N}_2\text{O} \). During summer a significant relationship \( P < 0.05 \) was only observed for \( \text{NH}_3 \). During winter, the strongest concordance was observed between measured values and predictions based on Tier II (0.16), although this was not strong. Concordance between measured and predicted values of \( \text{NH}_3 \) and \( \text{N}_2\text{O} \) were very low, despite strong correlations \( \text{N}_2\text{O} 0.96 \) and \( \text{NH}_3 -0.71 \). A similar effect was observed during summer, with low concordance, but strong correlations \( \text{N}_2\text{O} 0.67 \) and \( \text{NH}_3 0.86 \). A strong correlation

Figure 5.6 Comparison between measured and predicted emissions of \( \text{NH}_3 \) (A) and \( \text{N}_2\text{O} \) (B) from a northern Australian feedlot site. Horizontal error bars indicate SD of measured emissions (averaged over the whole measurement campaign); vertical error bars indicate SD of predicted emissions. (averaged over the whole measurement campaign).
was observed for measured and predicted enteric emissions based on the Tier II equation (0.99) and a strong negative correlation for Moe and Tyrell (1979) equation (-0.94).

5.3.6 Combined Data
In Chapter 4 the National Inventory Methodology (AGO 2006) for predicting greenhouse gas emissions (CH$_4$, N$_2$O and NH$_3$) was evaluated using the results of published studies. This evaluation indicated that for prediction of CH$_4$ emission utilising an equation based on the CHO fractions in the ration provided the most accurate estimates, although an over estimation was observed for CHO based equations and an under estimation for the Tier II (3% GEI) equation. This was consistent with the results of the application of the model to Australian measurements as described in this Chapter.

Insufficient published data was available to validate the accuracy of estimates of N$_2$O emission. However, combining published estimates emissions of NH$_3$ with the measured data demonstrated that estimates were more accurate (predictions closer to measured values) at lower emissions levels; however the results of the validation exercise suggested that a single emission factor to estimate either N$_2$O or NH$_3$ does not allow the impact of environmental conditions on emissions to be accounted for. This leads to inaccuracies in estimates, despite the methodology producing reasonable estimates of N excretion.

Examination of the combined dataset (Figure 5.7 and 5.8) demonstrates the similarity of measured CH$_4$ emissions (Figure 5.7) to those reported in the literature, and consistency of predicted emissions based on the three equations utilised in this evaluation. Variability (standard deviations as shown in error bars) was also similar between the measured emissions reported in this Chapter (5), Chen et al. (2009) and the published studies. The combination of the published emissions data (and corresponding models) and the measured emissions/predictions results in changes in the significance of linear relationships; using the combined data (Table 5.13) a significant ($P<0.01$) linear relationship was determined between measured and predicted emissions based on the equation of Moe and Tyrell (1979). Despite no significant ($P>0.05$) linear relationship being observed for the Tier II equation standard errors of observation and prediction were comparable to those for Moe and Tyrell (1979). Using the combined data, estimates of enteric CH$_4$ based on Ellis et al. (2007) were less accurate than those for the Australian data, reflecting reduced accuracy in the published predictions.

A significant linear relationship (Table 5.13) was observed between the combined measured and predicted data for N$_2$O. In contrast to the published data there was no significant relationship between measured and predicted emissions of NH$_3$ using the combined data set.
Table 5.13 Fitted linear relationships and SE for measured and predicted emissions of CH₄, N₂O and NH₃ based on published values and measurements from two Australian feedlots over seven measurement campaigns. Predicted emission of CH₄ is based on three equations.

<table>
<thead>
<tr>
<th>Equation</th>
<th>y-intercept</th>
<th>Slope</th>
<th>Significance</th>
<th>SE Obs</th>
<th>SE Pred</th>
</tr>
</thead>
<tbody>
<tr>
<td>Moe Tyrell (1979)</td>
<td>127.8</td>
<td>0.48</td>
<td>P=0.003</td>
<td>21.6</td>
<td>4.6</td>
</tr>
<tr>
<td>Tier II</td>
<td>65.6</td>
<td>0.16</td>
<td>P=0.287</td>
<td>21.8</td>
<td>4.7</td>
</tr>
<tr>
<td>Ellis et al. (2007)</td>
<td>154.7</td>
<td>0.23</td>
<td>P=0.402</td>
<td>41.0</td>
<td>8.7</td>
</tr>
<tr>
<td>N₂O</td>
<td>36.2</td>
<td>0.15</td>
<td>P&lt;0.001</td>
<td>10.1</td>
<td>2.4</td>
</tr>
<tr>
<td>NH₃</td>
<td>6.1</td>
<td>0.13</td>
<td>P=0.170</td>
<td>0.502</td>
<td>0.2</td>
</tr>
</tbody>
</table>

The highest concordance (Table 5.14) between measured and predicted emissions of enteric CH₄ was observed for predictions based on Moe and Tyrell (1979), consistent with the results of the published studies (Chapter 4) a similar correlation (0.60 compared with 0.62) was also observed. Predictions of enteric CH₄ using both Tier II and Ellis et al. (2007) showed low concordance and correlations (0.10 and 0.23 for Tier II and 0.09 and 0.18 for Ellis et al. (2007) when the combined data set was examined. Concordance between measured and predicted emissions of NH₃ (0.17) increased compared with the Australian feedlot data (-0.01), but decreased compared with published data (0.29). Utilising the combined data set decreased concordance between measured and predicted emission of N₂O (0.06 compared with 0.07); however correlation was reduced (0.50 to 0.76).

Table 5.14 Concordance and correlations between measured and predicted emissions of CH₄, N₂O and NH₃ based on published data (as used in the evaluation reported in Chapter 4) and measurements from two Australian feedlots over seven measurement campaigns.

<table>
<thead>
<tr>
<th>Equation</th>
<th>Concordance</th>
<th>Correlation</th>
<th>Cᵇ</th>
</tr>
</thead>
<tbody>
<tr>
<td>Moe and Tyrell (1979)</td>
<td>0.20</td>
<td>0.60</td>
<td>0.33</td>
</tr>
<tr>
<td>Tier II</td>
<td>0.10</td>
<td>0.24</td>
<td>0.41</td>
</tr>
<tr>
<td>Ellis et al. (2007)</td>
<td>0.09</td>
<td>0.19</td>
<td>0.47</td>
</tr>
<tr>
<td>N₂O</td>
<td>0.06</td>
<td>0.50</td>
<td>0.12</td>
</tr>
<tr>
<td>NH₃</td>
<td>0.17</td>
<td>0.78</td>
<td>0.22</td>
</tr>
</tbody>
</table>
Figure 5.7 Comparison between measured and predicted enteric methane emissions based on a database of published studies (literature), and measurements from two Australian feedlot sites (Aus). Predictions were based on three equations IPCC Tier II (A), Moe and Tyrrell (1979) (B) and Ellis et al. (2007) (C).
Chapter 5. Model Application

5.4 Discussion

This Chapter aimed to apply the predictive equations examined in the previous Chapter (4) to data collected from two Australian feedlots over seven measurement campaigns, and compared predictions to measured emissions during these periods. Sources of variation in measured emissions between sites and seasons were also examined.

5.4.1 Measured Emissions

Measured CH$_4$ emissions are assumed to represent primarily enteric CH$_4$ as manure CH$_4$ was estimated to contribute less than 2 g/head/day (Hao et al. 2001a). Measured emission from the Southern Site averaged 110 g/head/day, whilst emissions from the Northern Site averaged 132 g/head/day (excluding summer 2008, 64 g/head/day).

Figure 5.8 Comparison between measured and predicted emissions of nitrogenous gases; NH$_3$ (A) and N$_2$O (B) from a database of published studies (Literature) and measurements from two Australian feedlot sites (Aus).
Using calorimetric methods Beauchemin and McGinn (2005) measured emissions from high grain rations of 62 g/day (corn based) and 80 g/day (barley based) and 170 g/day (corn based) and 130 g/day (barley based) high forage diets. Similarly Boadi et al. (2004b) measured emissions of 90 g/head/day and 128 g/head/day (for high and low grain rations) using calorimetric methods. Beauchemin and McGinn (2006) reports emissions of 114 g/head to 169 g/head and Hegarty et al. (2007) report emissions from Australian feedlot cattle (measured using SF6) of 142 g/head/day (low intake, 8.4 kg DM/head) and 190 g/head/day (high intake 14.1 kgDM/head/day). McGinn et al. (2007) used micrometeorological methods to measure CH$_4$ emissions from feedlots in both Australia (also reported by Loh et al. 2008) and Canada reporting 166 g/head/day for the Australian feedlot, and 214 g/head/day for the Canadian feedlot.

Emissions measured during the seven measurement campaigns are consistent with enteric emissions reported for beef feedlot cattle of similar weights and intake levels. The high grain diets evaluated by Beauchemin and McGinn (2005) with reported emissions of 62 and 80 g/head/day demonstrated very low intakes (5.3 to 6.9 kg DM/day) compared with those during the measurement campaigns reported here (9-10 kg DM/day at the Southern Site, and 10–11 kg DM/day at the Northern Site). Reduced intakes are consistently cited as an effect of using calorimetric methods, with alterations to animal feeding behaviour and energy requirements due to reduced activity. Beauchemin and McGinn (2005) observed reductions in intake of 31% on a high forage diet, and 22% on a high grain diet when animals were moved to calorimetric chambers for measurements. Measured emissions reported in this study are consistent with those reported by Hegarty et al. (2007) and Boadi et al. (2004b) where intakes and live weights are similar to those reported by the feedlot operators during the measurement campaigns.

Emissions of CH$_4$ vary both between site and season; although there does not appear to be any distinct seasonal trends in emissions. However, emissions during 2007 at the Southern Site were higher (123 and 128 g/head/day) than those in 2006 (989 g/head/day) and 2008 (91 g/head/day). The major difference between these measurement periods where reported intakes, live weights and target live weight gains were similar was the grain types and average grain proportion in the rations. During 2007, average grain concentration was 61%, compared with 70 and 71% during the other measurement periods, whilst barley was the primary grain source in 2007, compared with wheat for the other campaigns. Whilst Beauchemin and McGinn (2005) determined a significant difference in emissions between barley and corn diets due to differences in the fermentative properties of the grain. The difference between wheat and barley; slightly higher cellulose (46.6 cf. 21.4 g/kg DM) and lignin (17.0 cf. 11.2
g/kg DM) in barley and higher starch (674.1 cf. 561.7 g/kg DM) in wheat (Givens and Moss 1990) are likely to be less pronounced than differences between barley and maize (corn). Maize (corn) contains more rumen protected starch, reducing starch digestibility in the rumen, depending on processing method (Archibeque et al. 2006). Starch digestion in the rumen is affected by both starch concentration and physical form of the starch containing grain (Oba and Allen 2003) although starch concentration is higher in grain such as sorghum and maize (compared to wheat or barley) it is less available due to enclosure by a hard endosperm (Oba and Allen 2003; Remond et al. 2004). Less available starch passes to the intestine, where it is degraded (with variable efficiency); without contributing to rumen fermentation. Although this does not explain differences in emissions observed between measurement campaigns at the Southern Site, it may contribute to differences in emissions between northern and southern feedlot systems, where different base grains are utilised.

There is little difference in the average calculated concentrations of soluble residue, cellulose, hemicellulose and lignin between the average ration fed during each measurement campaign at the Southern Site. However, the lower grain concentration (9-10% difference) in the rations during this period is likely to result in differences in fermentation patterns, primarily associated with higher available starch, and an increase in hydrogen and CH$_4$ production which may not be reflected in a simple examination of dietary properties. Measured emissions at the Northern Site were more similar between measurement campaigns, which appear to reflect greater consistency of dietary grain concentration (79-81%). A strong negative correlation (-0.96) was observed between grain concentration and measured emission (g/head/day) for the Southern Site, which is commonly observed in both beef and dairy cattle (Beauchemin and McGinn 2005).

Todd et al. (2008) measured NH$_3$ emission rates from a commercial feedlot in the Southern high plains of Canada and determined mean summer emissions of almost twice those in winter. In general, emissions were higher during summer (117 and 130 g/head/day) than winter (51 and 84 g/head/day). Similarly, McGinn et al. (2007) used micrometeorological measurements and a dispersion model to measure NH$_3$ emissions from a Canadian feedlot during Summer, and determined emissions averaging 140 g/head/day. Emissions measured during the 8 measurement campaigns were similar to these results (excluding the Northern Site summer 2008 and the Southern Site winter 2007), although there did not appear to be any distinct seasonal trends for these sites. Excluding emissions measured at the Northern Site in summer 2008 (as discussed above) NH$_3$ emissions from the Northern Site averaged 123 g/head/day compared with 178 g/head/day from the Southern Site.
Ammonia emissions can be expressed as a percentage of animal N intake, Todd et al. (2008) report NH₃ emissions annually of 53% of N intake (32 to 72%). Whilst other studies report NH₃ losses as a percentage of total N intake to be between 29 and 63% (Cole et al. 2006; Flesch et al. 2007; Todd et al. 2008; Todd et al. 2005). Within site, N intakes were similar (13 to 13.5% CP for the Southern Site, 13.4 to 13.7% Northern Site), despite larger difference in NH₃ emissions between sites and seasons (within sites). This contrasts with the observations of Todd et al. (2008); where despite similar N intake NH₃ loss (g/head/day) was 30-40% higher during summer than in winter. This indicates that differences in manure management and climatic conditions are primarily responsible for the differences in NH₃ observed (although this may not be seasonally dependant). NH₃ emissions were more strongly correlated with humidity (min 0.64, max 0.54) and wind (-0.67) than min or max temperature (min -0.44, max -0.21) and rainfall (-0.24; Todd et al. 2008).

Measured N₂O emissions from both sites were variable, averaging 2.6 g/head/day at the Southern Site (0.1 to 5.3 g/head/day) and 3.6 g/head/day (1.6 to 5.7 g/head/day) at the Northern Site (excluding summer 2008, 4.8 g/head/day). Boadi et al. (2004b) reports N₂O emissions equivalent to 0.16 and 0.15 g/head/day. However, these measurements were based on small chamber measurements (in situ). Chambers were 20 cm diameter, and 6 used in each pen area. There is considerable spatial variation within a single pen, therefore the chamber measurements may not reflect potential emissions from the whole surface. Further, environmental conditions (reported as cold temperatures, averaging 0.4, -8.0 and -6.4 °C for the three measurement periods (Boadi et al. 2004b)) are considerably lower than those observed in the measurement campaigns reported here, which is likely to limited microbial activity in the observations of Boadi et al. (2004b).

There is a significant lack of published information regarding the loss of N₂O from feedlot surfaces, although an increasing number of studies (Chadwick 2005; Hao et al. 2001a) have examined losses from manure composting, spreading and effluent systems (mostly for housed dairy cattle; Jungbluth et al. 2001; Kulling et al. 2003). In general, measurements of nitrogenous gas loss from feedlot systems focus on the volatilisation of NH₃, which although considered a direct greenhouse gas, is a secondary source of N₂O, and an environmental polluter. The association between NH₃ and other malodorous volatile compounds produced in feedlot systems is likely to result in calls for a reduction in NH₃ (McGinn et al. 2003). Further investigation is required into the deposition of NH₃ (and potential N₂O emissions) as well as the impact of a reduction in NH₃ volatilisation on direct N₂O emissions. In this study there was a moderate negative correlation (R² -0.68) between emissions of NH₃ and N₂O (over 7 measurement campaigns). Conditions which favour volatilisation result in decreased N
concentrations in the manure pad/soil substrate, reducing available N. This moderate correlation indicates that practices which decrease NH$_3$ volatilisation may in fact increase direct N$_2$O loss (although feedlot conditions generally do not favour N$_2$O production), which has more significant consequences as a greenhouse gas.

5.4.2 Emissions Prediction

In Chapter 4, which evaluated the prediction equations for enteric CH$_4$, N transactions and emission of nitrogenous gases, it was concluded that using an equation based on the carbohydrate concentration/composition (e.g. Moe and Tyrell 1979 and Ellis et al. 2007) improved emissions estimates (based on concordance, which is a measure of both the correlation between the values and the slope and origin of the line). The Tier II equation demonstrated the lowest SE of prediction and observation, suggesting low residuals and improved accuracy of estimates, however concordance was reduced compared to CHO based equations. It was concluded that utilising an equation based on the CHO composition of the ration would improve estimates compared with the energy based models.

For the measurement campaigns and periods of data collection at Australian feedlots utilised here the Tier II equation appears to provide the most reasonable estimates. The Tier II equation demonstrates the highest concordance observed (utilising Australian measurements) between measured and predicted emissions. As detailed in the general methodology the Tier II equation estimates enteric CH$_4$ emission based on intake of Gross Energy, and a CH$_4$ conversion factor, which is loosely based on ration type. A value of 3 ± 1% of gross energy is prescribed for feedlot cattle consuming a high (greater than 90%) grain diet. The primary difference between the range of studies utilised in the evaluation, and those fed in the measurement campaigns outlined here is the content of grain in the rations. Using the Tier II predictions (Figure 5.3) at the Southern Site as an example (and based on the trends discussed above) the need for an emission factor which considers grain concentration at an individual site is demonstrated clearly. For the lower measured emissions (and higher grain concentrations) predictions demonstrate residuals of less than 5 g/head/day. Higher measured emissions (and high grain concentration) show residuals closer to 30 g/head/day. The Tier II equation is suggested by the IPCC for diets with grain concentrations above 90%, however the range of grain concentrations for which this is effective has not been established. Although studies selected for validation were those feeding “feedlot type” rations, the overall grain content ranged from 30 to 90% (65% average), which is likely to result in underestimation in predicted emissions at the lower end of grain concentration. Average grain percentages reported over the 8 measurement campaigns at the Northern and Southern
feedlots ranged from 61 to 81%, which is consistent with improved predictive ability for higher grain rations.

The evaluation of the model for nitrogen transactions in Chapter 4 suggested that whilst faecal excretion (and retained N) was poorly estimated overall N excretion could be predicted quite accurately (concordance 0.94). Due to the measurement methods utilised for this study, detailed examination of N excretion was not able to be undertaken, however based on the evaluation in Chapter 4, it is assumed that excretion is being estimated accurately. The aim of this section was therefore to examine the accuracy of the emission factors applied to excreted N in order to predict NH$_3$ and N$_2$O emission. The implication of using a single emission factor to estimate volatilisation of NH$_3$, which is affected not only by excreted N, but by environmental conditions, is demonstrated clearly by these results. At low levels of measured NH$_3$ accuracy of prediction is reasonable, however as measured NH$_3$ increases the ability of the model to predict emission decreased substantially. A significant linear relationship could not be determined ($P>0.05$), and both concordance (0.005) and correlation (0.15) were low. This effect can be considered to be primarily due to the use of the single emission factor (0.3) which does not reflect the wide range of environmental conditions influencing NH$_3$ volatilisation (temperature, moisture, pH). Measured NH$_3$ emissions range from 94 ± 2.8 g/head/day to 324 ± 34.7 g/head/day (mean and SE), with no distinct seasonal or site trends. Percentage loss of NH$_3$ has been reported to vary between 29 and 60% of N intake, and up to 70% of N excreted (McGinn et al. 2007); this alone suggests the difficulty in setting a single emissions factor.

As reported by McGinn et al. (2007) a significant factor contributing to the under estimation of NH$_3$ emissions may be related to accumulated manure in the pens. Although the majority of NH$_3$ is quickly volatilised, the accumulated manure may still be a source of volatile NH$_3$. These emissions are not accounted for by the empirical equation utilised in the methodology. There is also a difference in manure management strategies between sites, which is likely to contribute to differences in measured emissions, and in accuracy of prediction. The chemical composition of manure is also affected by animal size, age and condition, water consumption, feedlot surface, animal density, amount of type of bedding and handling of the manure/surface (Miller et al. 2003). Volatilisation of NH$_3$ is influenced by not only N excreted (and available in the substrate) but by ambient and substrate temperature, pH and moisture content, leading to considerable variation not only with a day, but between site and season. The use of a single emission factor, and similarity in ration crude protein, weight gain and feed offered results in very low variability in predicted volatile NH$_3$, which is not reflective of the biological variability.
In order to more accurately estimate emissions a more detailed model describing microbial transactions, effects of manure pad composition and climatic conditions is required. Given the significant role of environmental conditions and manure/soil composition on emissions of NH$_3$ it is unlikely that a single emissions factor will be appropriate for estimating emissions, particularly over short time scales. However, longer term studies may reveal the emissions factor to be accurate on an annual basis.

Despite the significant ($P<0.05$) linear relationship between measured and predicted emissions of N$_2$O, a low concordance was observed. In contrast to NH$_3$ emissions of N$_2$O are over predicted, particularly at lower measured levels. However, similarly to NH$_3$, the inaccuracies at one extreme of measured values are primarily due to the impact of environmental conditions, which are not represented by a single emissions factor. Further, as detailed in Chapter 3, the emissions factor for this study was derived from *in vitro* measurements (Kulling *et al.* 2003); using manure collected from a variety of dairy manure management/storage methods. A key difference between the manure management practices described in the study of Kulling *et al.* (2003) is the addition of straw bedding to the manure. Misselbrook and Powell (2005) examined the addition of various types of bedding on emissions of NH$_3$ from dairy manure (in vitro), based on evidence that NH$_3$ volatilisation can be reduced with the addition of bedding. Chemical changes associated with addition of bedding including changes in temperature, oxygen availability, pH, cation exchange capacity and provision of a C source can influence microbial transactions in the soil/manure. Misselbrook and Powell (2005) suggest that increased C in the substrate may increase immobilisation of NH$_4$-N. It is possible that this increases N in the substrate available for nitrification/denitrification reactions, increasing N$_2$O emitted. This is likely to affect the accuracy of application of this emission factor to feedlot situations.

Similarly to NH$_3$, a more detailed methodology which is able to account for the impact of substrate composition and environmental conditions on N$_2$O emissions is required. Further study examining emissions of N$_2$O from a variety of manure management options, but particularly dry lot manure management systems will be required before accurate estimations can be made.

### 5.4.3 Site Specific Differences

On average, measured CH$_4$ emissions (Table 4) from the Southern Site (110 g/head/day) were slightly lower than from the Northern Site (132 g/head/day). NH$_3$ emissions averaged 178 g/head/day at the Southern Site compared with 173 g/head/day for the Northern Site. Despite the similarities in average emissions, differences in characteristics of the different sites were
hypothesised to result in differences in the accuracy of the various equations utilised in the prediction of enteric \( \text{CH}_4 \) emissions. The major differences in animal characteristics between the sites are the average live weight (440 to 520 kg at the Southern Site compared with 530 to 750 kg at the Northern Site, Table 5.2), offered feed amount (9.8 kg DM/head/day Southern Site, 10.7 kg DM/head/day Northern Site), the grain content of the rations (Table 5.5) and the types of grains used (predominately wheat at the Southern Site and sorghum at the Northern Site). Climatic conditions vary between sites; temperatures (both minimum and maximum) were higher and less variable at the Northern Site. Average wind speed and summer gust speed was faster at the Northern Site, potentially increasing diffusion gradients by removing emitted gases from the surface.

The slightly higher \( \text{CH}_4 \) emissions measured at the Northern Site could be attributed to the higher live weights and intakes observed. Average \( \text{CH}_4 \) emissions at the Northern Site equated to 12.4 g \( \text{CH}_4/\text{kg DM} \) compared with 11.2 g \( \text{CH}_4/\text{kg DM} \) intake at the Southern Site, the additional 0.8 kg DMI at the Northern Site could be expected to increase average \( \text{CH}_4 \) output by 9.3 g/day to 119 g/day (based on average production \( \text{CH}_4 \) g/kg DM at the Southern Site). This indicates that the higher \( \text{CH}_4 \) emissions observed are not simply an artefact of higher intakes.

The effect of a different grain type cannot be ignored. The previous discussion focused on the difference between emissions associated with wheat or barley based rations at the Southern Site, however the small difference in fermentative characteristics (considered as starch, cellulose and hemicellulose as these are key model components) between wheat and barley was considered unlikely to be the primary cause of the differences in measured emissions between years. Grain concentration of rations showed a strong negative correlation with measured emission at the Southern Site. However, a more significant difference exists in grain types between sites. Grains at the Northern Site are predominantly sorghum, whilst wheat and barley are more common at the Southern Site. Sorghum contains considerably higher starch (730 g/kg DM) than barley (560 g/kg DM), more comparable to maize grain (699 g/kg DM), although lignin is considerably lower in maize (6.3 g/kg DM) than in sorghum (23.8 g/kg DM; Givens and Moss 1990).

Differences in metabolism between tropical (\textit{Bos Indicus}) and temperate (\textit{Bos Taurus}) type cattle may contribute to differences in emissions between sites. Typical of Northern Australian feedlots, the Northern Site contained tropical breeds (e.g. Brahman) during all measurement campaigns, while the Southern Site contained predominantly temperate/British breeds (e.g. Angus and Hereford). Kurihara \textit{et al.} (1999) reports that relationships between
CH$_4$ production, energy utilisation and live weight of *Bos Indicus* (tropical) cattle when fed tropical forages (e.g. Ruzi grass and Rhoades grass) differ from those of *Bos Taurus* cattle fed temperate forages, however they do not report comparative CH$_4$ production for tropical cattle fed temperate forages. Hunter (2007) in a letter regarding the results of Kurihara *et al.* (1999) this indicate that CH$_4$ production from tropical cattle fed high grain diets (per unit of gross energy intake) is higher than for *Bos Taurus* cattle. The high number of tropical cattle in the Northern Australian production system is likely to be a contributing factor to the higher emissions observed at this site, and particularly the higher emissions/kg DMI.

The Tier II equation could be expected to more accurately predict emissions from the Northern Site; due to the higher grain content (71-81%), compared to the Southern Site (61 to 71%). The Tier II equation demonstrates the highest concordance (Table 5.12) of the CH$_4$ prediction equations evaluated for both sites. However, closer examination of the predictions for individual measurement campaigns shows underestimation of emissions. Similarly to the overall analysis, no significant ($P>0.05$) linear relationships were determined between measured and predicted emissions at either site. Concordance was significantly reduced for all CH$_4$ prediction equations at both sites. From the Southern Site, the lowest standard errors (of observation and prediction) were observed for Tier II (4.35 and 2.17); however Ellis *et al.* (2007) showed the lowest standard errors (observation and prediction) and highest correlation for the Northern Site. These results suggest that the difference in composition of rations (the major difference between sites in terms of CH$_4$ emission), particularly the source of grain, influences the accuracy of emissions estimates. Specifically, the distinct variation in emissions related to grain type reported by Beauchemin and McGinn (2005) and observed to some extent in this study (sorghum vs. barley), as well as forage concentration, suggests that utilising an equation based on ration nutrient fractions, even when these are estimated, will probably allow more accurate estimates of CH$_4$ output based on the individual site characteristics.

Although Tier II shows the highest concordance at the Northern Site, the reduced accuracy, as indicated by higher standard errors of observation and prediction compared to the Southern Site, may be influenced by the factors discussed above in the context of between site differences. Such as alterations in fermentative characteristics of grain types (sorghum compared with wheat or barley) and in the metabolism of tropical compared with temperate cattle breeds.

Ammonia emissions from the Northern Site averaged 123 g/head/day, compared with 178 g/head/day from the Southern Site. Nitrous oxide emissions averaged 2.6 g/head/day from the
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Southern Site and 3.6 g/head/day from the Northern Site. The primary factors influencing emissions of N\textsubscript{2}O and volatilisation of NH\textsubscript{3} are N content (N excretion), moisture content, temperature, pH, soil compaction, oxygen availability and microbial activity. These factors are all likely to vary both between and within sites, and contribute to the differences in emissions observed.

Nitrogen intakes were similar between sites, 212 to 233 g/head/day (averaging 223 g/head/day) at the Southern Site and 222 to 250 g/head/day (averaging 233 g/head/day) at the Northern Site. The slightly higher N intake at the Northern Site could be expected to result in higher NH\textsubscript{3} emissions, however this was not observed. Over feeding N relative to requirements results in increased excretion of N, primarily in the urine, this is readily volatile as NH\textsubscript{3}. The difference in fed N g/kg live weight (0.42 Northern Site, 0.47 Southern Site) indicates that feeding level of N is slightly higher at the Southern Site; however it would be unlikely to contribute to the magnitude of difference observed in emissions. Consistently, estimated N excretion (calculated based on the modelling approach described) was similar between sites (204 g/head/day south, 208 g/head/day north). Ammonia emissions equate to 87% (south) and 59% (north) of excreted N, or 80% and 53% of N intake. The Southern Site values are considerably higher than the normal range reported. However, NH\textsubscript{3} emissions measured using micrometeorological methods reflect not only freshly deposited manure but built up manure in pens, which may contribute significantly to total emissions. Manure pad management differed between the sites, manure was observed to be more built up in pens at the Southern Site, which were generally scraped clean after individual “lots” of cattle leave the feedlot. In contrast, pens were cleaned more frequently at the Northern Site, with manure mounded in the centre of pens. The less frequent cleaning results in a wetter, deeper and more variable surface. As discussed by McGinn et al. (2009) this build up of manure is likely to contribute to higher measured NH\textsubscript{3} emissions. Less frequent cleaning and manure build up also changes the composition of the manure pad below at the surface. Todd et al. (2005) report that NH\textsubscript{3} volatilisation in the feedlot comes from a pool of rapidly hydrolysed urea, replenished regularly by urination, and from NH\textsubscript{4} mineralised from organic forms. Built up manure is likely to contain organic N, which can be mineralised (a slower process) to provide a more constant source of NH\textsubscript{3} (Todd et al. 2008).

The lower N\textsubscript{2}O emissions observed at the Southern Site (and higher at the Northern Site) reflect the greater loss of N (in the form of NH\textsubscript{3}) at the Southern Site, where conditions appear to favour volatilisation over nitrification/ denitrification. Stewart (1970) determined that under laboratory conditions the amount of NH\textsubscript{3} volatilised and nitrate (NO\textsubscript{3}) accumulated depended on the moisture content of the soil/ substrate; when urine was added to dry soil (at 4
day intervals) more than 90% of N was lost as NH$_3$. When soil was initially wet about 65% of N was converted to nitrate. However, Miller and Berry (2005) determined that at very low moisture contents nitrogenous compounds were less likely to volatilize or convert to more volatile forms, and that increased moisture enhanced the volatilisation of NH$_3$.

Miller and Berry (2005) also determined that manure content (in the soil pan incubations) altered the effect of moisture on greenhouse gas emissions (CH$_4$ and N$_2$O), suggesting that with increased manure content (relative to the soil in the incubation) water availability for microbial activity decreased. At low manure contents under fermentative (high moisture) conditions, N$_2$O was the dominant greenhouse gas emission, whilst at high (75%) manure concentration N$_2$O emissions were absent. Consistently, Adams et al. (2004) determined that more frequent pen cleaning reduced N losses from open yard feedlots (and compost). This supports the observation that higher NH$_3$ loss at the Southern Site, associated with a greater build up of manure, further these observations go some way to explaining the differences in N$_2$O observed between sites. The more frequently cleaned, lower manure substrate at the Northern Site (which was observed in most campaigns to be drier) showed higher average emissions whilst the generally higher manure surface at the Southern Site showed lower average emissions. Additionally, Saggar et al. (2004b) reports increased N$_2$O emissions at higher temperatures, consistent with the higher average temperatures at the Northern Site.

Differences in animal characteristics, manure management practices and ration composition are likely to result in differences in accuracy of emission prediction between the sites (Boadi et al. 2004b). Although NH$_3$ was consistently under estimated, and N$_2$O consistently over estimated, the range in measured values (at a quite constant predicted value) highlights the issues with using a single emission factor. As discussed, predicted N excretion (based on animal and dietary parameters) is multiplied by an emissions factor. The differences in accuracy (as indicated by concordance) between sites, for both NH$_3$ and N$_2$O concordance was slightly higher for the Northern Site, perhaps indicates that if a more complex model is not applied, then a site specific parameter should be used. This approach is utilised for prediction of manure CH$_4$ in the National Inventory Methodology (AGO 2006), with different emissions factors for temperate and tropical regions. However, seasonal conditions could be expected to have a greater impact than site characteristics.

5.4.4 Season Specific Differences

Environmental and climatic conditions have a significant impact on emissions of N$_2$O and volatilisation of NH$_3$ (Saggar et al. 2004b), which could therefore be expected to contribute to both site and seasonal differences in emissions, and in the accuracy of a single emissions
factor. Miller et al. (2003) determined chemical properties of feedlot manure to be significantly different between seasons, including total C and P, NO$_3$-N, NH$_4$-N. Bacterial content of the manure was also seasonally affected, including numbers of E.coli and total coliforms. Saggar et al. (2004b) measured soil moisture, temperature, N availability and soluble C in dairy pastures during four seasons and determined that N input and water filled pore space were the factors which most affected N$_2$O emissions. They also determined large variability throughout the year in both grazed and un-grazed pastures. The differing composition between seasons is likely to contribute to differences in potential emissions, moderated by the effects of temperature (Miller et al. 2003).

As discussed above, there does not appear to be a distinct seasonal difference in measured emissions of NH$_3$. Average NH$_3$ emissions in summer were 129 g/head/day, while average winter emissions were 173 g/head/day. N$_2$O emissions averaged 2.9 g/head/day in summer, and 3.2 g/head/day in winter, with considerably greater variability observed between winters (at both sites) for both gases. Higher emissions of NH$_3$ would be expected when conditions favour a wetter surface, which promotes NH$_3$ volatilisation (Denmead et al. 2008). However, a substrate which promotes NH$_3$ volatilisation could be expected to result in decreased N$_2$O emission.

There was no substantial difference in the crude protein concentration of the rations between seasons, therefore differences in emissions between seasons is associated primarily with environmental conditions. Todd et al. (2008) attribute the seasonal differences observed in their study to increase in dietary N. Increasing CP above requirements increases excretion of urinary N, of which a large proportion is volatile NH$_3$. In the study of Todd et al. (2008) CP increased from 13.5 to 15% increasing N intake up 24 g/head/day, and increasing volatile N loss. However, there was little discussion regarding the impact of environmental or manure management conditions on these emissions. Crude protein concentrations of the rations (from the Australian feedlots reported) ranged from 13.0 to 13.7%, with no correlation with season ($R^2 < 0.05$). This suggests the differences in emissions between sites and seasons are related more closely to environmental conditions than differences in ration N (crude protein) concentration. However, it should be noted that live weights and growth rates are lower at the Southern Site. Despite lower average DMI, cattle at the Southern Site are offered on average 0.98 kg CP/kg gain, compared with 0.85 kg CP/kg gain at the Northern Site. Potentially contributing to higher NH$_3$ emissions at this site (reduced efficiency of N use).

Significant linear relationships ($P<0.05$) were determined between measured and predicted emissions of NH$_3$ for both seasons, however standard errors of prediction suggest Summer
estimates were more accurate, supported by higher concordance. The discussion above suggested a drier surface led to more accumulation of N in the substrate (Miller and Berry 2005) and reduced NH$_3$ emissions (Stewart 1970), suggesting that the conditions in Summer better represent the current emissions factor. As argued for the site differences, the differences in accuracy between seasons highlight the need for either a more complex model of NH$_3$ emissions. This model needs to account for not only differences in composition associated with seasonal conditions, but with the contribution of built up manure, and differences in pen cleaning interval.

Significant ($P<0.05$) linear relationships were determined for emissions of N$_2$O separated by summer and winter (SE of both observation and prediction was decreased substantially compared with the overall data set). However, in contrast to NH$_3$, concordance and correlation demonstrated more accurate predictions during winter than summer. This suggests that the 0.02 emission factor is more appropriate for the cooler temperatures and possible wetter surface (particularly at the Southern Site) observed during winter than the desiccated surface commonly observed in summer, despite the greater variability in emissions. Saggar et al. (2004a) determined more variable emissions during winter and spring than the drier autumn measured. They suggest that a model which accounts for climatic variation in rainfall and soil texture is better able to predict emissions more accurately than a single emission factor (Saggar et al. 2004a).

Environmental conditions are likely to have less impact on emissions of CH$_4$ than on the nitrogenous gases. CH$_4$ emissions averaged 122.9 g/head/day in winter, compared with 115.4 g/head/day in summer. However, as discussed there did not appear to be any seasonal trends in emissions within site. Alterations in ration composition did not appear to have any distinct seasonal trends, although there was some difference in types of grain used between years, particularly at the Southern Site. Alterations in intake behaviour associated with higher day time temperatures may influence total intake, leading to differing emissions between summer and winter. Reported intakes average 10.0 kg DM/head/day for the Summer campaigns, and 10.3 kg/head/day for the Winter campaign, resulting in 11.5 g CH$_4$/kg DM during Summer, and 12.0 g CH$_4$/kg DM during Winter. Blaxter and Wainman (1961) determined that CH$_4$ production did not change in response to environmental temperature (between -5 and 35 °C), therefore differences between seasons in CH$_4$ emissions appears to be primarily associated with slight differences in intake. McGinn et al. (2008) reports lower CH$_4$ emissions during daylight hours from the Northern feedlot utilised in this study, which they attribute to changes in intake associated with heat stress. They report that shade seeking behaviour, and a reduction in intake to manage heat load (Mader et al. 2002) may result in reduced intake and
therefore lower emissions. This effect is likely to be translated to total daily emissions, which would not be accounted for in the intake figures reported by the feedlot operators (these are offered feed amounts kg DM/head/day).

The short term nature of the measurement campaigns contributes to difficulty in making assumptions regarding seasonal differences in emissions. This is particularly important for the emissions of nitrous oxide and ammonia. Pulses or peaks of N$_2$O have been observed following rainfall events in dairy pastures (Saggar et al. 2004a) for long periods after urine/N deposition. The short term measurements may (or may not) have captured periods of high emissions. Longer term monitoring of emissions, N consumption and manure pad characteristics would increase the ability to assess these emissions and accuracy of emissions estimates on a longer term basis. However, these results indicate that seasonal differences are unlikely to have a significant impact on accuracy of enteric CH$_4$ emission prediction, where they are not associated with substantial changes in ration composition. The consistency of rations utilised in a feedlot environment should enable the same predictive equation to be used to estimate emissions on an annual basis, once they are further refined to improve accuracy of estimates.

### 5.4.5 Combined Data Set

In contrast to the evaluation based on the two feedlot sites, when the combined data set is utilised, the equation of Moe and Tyrrell (1979) demonstrates increased accuracy and a significant linear relationship between measured and predicted emissions. Differences in accuracy of CH$_4$ prediction equations between the published data and that reported in this study may be related to the measured animals. Feedlots house animals at varying stages of growth, size, intakes and rations (Stackhouse et al. 2011) in contrast to the more consistent animals used in calorimetric measurements. On an individual animal/pen basis, over a smaller range of live weights, intakes and rations composition the more specific equation based on ration composition may prove more accurate. Over an entire feedlot, the simpler Tier II equation appears to be more “generally” right, although it is less suitable for use with more specific studies.

### 5.4.6 Role of DMI

As determined by Bell et al. (2009) daily DMI is highly correlated with live weight and gross or metabolisable energy intake. Based on the observations discussed previously in this Chapter, the Tier II equation linking gross energy consumption to CH$_4$ production provides a reasonable estimate of CH$_4$ emissions over the range of data evaluated. The difference
between the approach used here, and that used in the National Inventory Methodology (2006), is that offered feed was utilised as the predictor of intake, rather than a set value for feedlot cattle class. In Chapter 4 (section 4.4) this set value was observed to have the lowest concordance with offered feed, with set values generally considerably higher than offered feed. However, using a class based estimate of intake improved accuracy of emissions based on the three best performing equations, compared to offered feed, feed based on live weight % and feed intake predicted based on growth (Minson and McDonald 1987).

For the data reported utilising a class-based-intake improves the concordance from Moe and Tyrrell (1979) (from 0.01 to 0.02) and correlation between measured and predicted emissions (-0.32 compared with 0.07). In contrast, concordance and correlations reduced for Tier II and Ellis et al. (2007) compared with offered feed based measurements. As Stackhouse et al. (2011) suggests, the cattle in a feedlot are made up of a range of live weights, intakes, growth stages and potential emissions. When offered feed is used as a predictor, the Tier II equation appears to be account for the differences in emissions over the feedlot, providing the most accurate estimates. When a class-based-intake value is used, it appears to add an additional layer of uncertainty, decreasing accuracy of estimates. The improvement in estimates based on (Moe and Tyrrell 1979) with the class-based-intake could potentially be related to the more specific accounting for ration composition.

5.5 Conclusion

The application of the model to data collected from commercial feedlots highlights the observations of Mills et al. (2003) that validation of an approach in one situation does not guarantee accuracy in another. In contrast to the published data evaluated in the previous Chapter, for an analysis covering both feedlots the Tier II methodology provided the most accurate estimates. However, this may be attributed in some part to the micrometeorological measurement method, which encompasses a wider range of animal characteristics, intakes, rations and potential emissions. The combination of Moe and Tyrrell (1979) and a class-based-intake prediction is the current methodology for estimating emissions from feedlot cattle in Australia. This evaluation indicates that while Moe and Tyrrell (1979) shows the best estimate when a class-based-intake is used; Tier II appears more accurate when a feedlot average offered feed is applied.

The differences in accuracy of CH₄ prediction equations between seasons and sites indicate that a modified CHO based equation, or a ration specific emission factor (based on grain
content) will be more accurate than a single value. However, further research will be required to determine these for Australian feedlot conditions.

For \( \text{NH}_3 \) and \( \text{N}_2\text{O} \), the application of the model of the data here demonstrates clearly the primary issue with the approach of the current methodology. The use of a single emissions factor estimating emissions from excreted N does not allow the effects of manure composition, management and climatic conditions to be accounted for. The short term nature of these measurement campaigns means that it is difficult to determine if these single emissions factors will prove to be accurate on an annual basis (although they are not on a daily or weekly basis). Similar to \( \text{CH}_4 \), more detailed studies focussing on greenhouse gas emissions from Australian beef feedlots are required in order to increase reliability of modelling approaches.
Chapter 6. Correlations between Diurnal Patterns of Greenhouse Gas Emissions and Feeding Behaviour of Feedlot Cattle

6.1 Introduction

Distinct diurnal emissions patterns have been observed for \( \text{NH}_3 \) and \( \text{CH}_4 \), particularly when continuous measurements are conducted using micrometeorological techniques. Methane emissions appear to be cyclical and greatest during periods of rumination (Denmead et al. 2000; Harper et al. 1999). Denmead (1994) observed short term cycles in \( \text{CH}_4 \) production (as well as a distinct diurnal pattern) of peaks during rumination and troughs during grazing. Lockyer (1997) reported \( \text{CH}_4 \) emissions which increase during daylight, reaching a peak near sunset and then declining towards sunrise. \( \text{NH}_3 \) emissions appear to follow the reverse pattern, being low in the morning and evening and highest during the middle of the day. Flesch et al. (2007) observed \( \text{NH}_3 \) emissions below early in the morning, increase into the early afternoon and then decline until sunset, a similar pattern was recorded by Loh et al. (2008).

Lockyer (1997) compared observed diurnal pattern of \( \text{CH}_4 \) with published records of sheep grazing behaviour and young cattle and concluded that the pattern of \( \text{CH}_4 \) emissions was comparable to published grazing behaviour. Methane emissions increased as grazing activity increased (with the accumulation of feed into the rumen), as rumination or idling increased following sunset \( \text{CH}_4 \) emission decreased (decrease in rumen contents; Lockyer 1997). A similar pattern was observed by Murray et al. (1999) who observed an increase in \( \text{CH}_4 \) emissions directly following time of feeding (grass). Murray et al. (1999) also noted a large flux when animals were fed, which they attributed to displacement of \( \text{CH}_4 \) from the rumen by the introduction of feed material. Harper et al. (1999) determined that short term variability in \( \text{CH}_4 \) emissions were primarily related to animal behaviour, however presented only a small part of the data identifying the correlation between rumination index (rather than total number ruminating) and \( \text{CH}_4 \) flux. It is well documented that the feeding behaviour of feedlot and grazing cattle follows a diurnal pattern, and is influenced by temperature and timing of feed delivery (for housed or feedlot cattle), as well as composition and characteristics of feed, and social hierarchies (DeVries et al. 2003; Fell and Clarke 1993). Hoffman and Self (1973) observed the first peak in feeding between 6 and 9 am, followed by a decline between 9 and 12, with a gradual increase to a second peak between 3 and 9 pm. Cattle are generally observed to most of their eating at sunrise and sunset, particularly during Summer (Gonyou and Stricklin 1984; Ray and Roubicek 1971). Feeding behaviour is also influenced by season, primarily due to temperature and photoperiod differences (Gonyou and Stricklin 1984).
Hoffman and Self (1973) observed the greatest activity in late afternoon and early evening in summer, and in later afternoon (3-6pm) in winter. Cattle were also observed to spend more time under shade in Summer than in winter (Hoffman and Self 1973). Feeding at sunrise and sunset appears to be relatively independent of the timing of feed delivery (Gonyou and Stricklin 1984).

However, animal behaviour may not be the only contributing factor to diurnal patterns of emissions. Measurements undertaken using micrometeorological methods may suffer from microclimate issues, primarily related to the inversion layer, or increased boundary layer stability, at night (Flesch et al. 2007; McGinn et al. 2009). This effect appears to be more pronounced for CH\textsubscript{4}. Flesch et al. (2007) observed little modulation of emissions related to temperature and wind speed, citing time of day (and animal activity) as a greater predictor of NH\textsubscript{3} emissions. Knowledge of behavioural patterns and associations with emissions may be useful in attempting to select a period of the day for short term measurements of emissions (which are then extrapolated to whole day estimates).

**6.1.2 Research Questions**

This section aimed to address the following research questions;

1. Is there a significant differences in emissions of CH\textsubscript{4} and NH\textsubscript{3} between times of day, which contributes to diurnal flux patterns?

2. How closely correlated are fluxes of greenhouse gas, animal behaviour and environmental conditions?

**6.2 Materials and Methods:**

**6.2.1 Site Selection**

Full details of the sites selected can be found in the General Methodology (Section 3.1). Behavioural observations were conducted in conjunction with the final winter (2007) and summer (2008) measurements at both sites.

**6.2.2. Animal Characteristics**

Observations of animal behaviour took place over two measurement campaigns; winter and summer at each site (four campaigns in total). A single pen was selected at each site during each campaign, which was representative of the range (size, days on feed, breed) of cattle currently on feed at each site.
6.2.2.1 Winter 2007

During the first (winter) measurement campaign at the Southern Site the pen contained 247 head of mostly *B. Taurus*, weight 434-442 kg, minimum 28 DOF. The ration fed contained 69.9% wheat, 12.5% silage, 3.5% grass hay, 5% liquid finisher, 8% cottonseed and 1.1% vegetable oil; 75% DM, 14.42% CP, 13.15 MJ ME/kg DM. Measured emissions (averaged over the whole site) were 122.8 g CH$_4$/head/day and 305.1 g NH$_3$/ head/day (Table 5.4). At the Northern Site the pen contained 161 head of *B. Indicus* steers with an estimated weight of 477 kg, approximately 30 days on feed. The ration fed comprises 11.5% silage, 0.5% recycled oil, 1.5% straw, 4.6% liquid supplement, 75.9% sorghum grain and 6% cottonseed; 69.15% DM, 13.50% CP, 12.76 MJ ME/kg DM. Measured emissions averaged 138.8 g CH$_4$/head/day and 94 g NH$_3$/head/day (Table 5.4).

6.2.2.2 Summer 2008

During the second measurement campaign the Southern Site contained 223 head of *B. Taurus* steers, between 480 and 570 kg and 60 to 75 days on feed. The ration fed contained 27% wheat, 14% silage, 1.5% grass hay, 4.8% liquid finisher, 8% cotton seed, 1.1% vegetable oil, 3.1% molasses and 40.5% barley; 73% DM, 13.5% CP, 12.8 MJ ME/kg DM. Emissions averaged 91.0 g CH$_4$/head/day, and 102.0 g NH$_3$/head/day (Table 5.4). The Northern Site pen contained 196 head of *B. Taurus* steers 642 kg, and 116 days on feed. The ration fed contained 39% sorghum, 4.5 liquid supplement, 1% straw, 1% cotton hulls, 10% cotton seed, 5% silage, 39% barley and 0.5% recycled oil; 73% DM, 13.83% CP, 12.67 MIE ME/kg DM. Emissions averaged for the whole site were 63.8 g CH$_4$/head/day and 324.4 g NH$_3$/head/day (Table 5.4).

6.2.3 Behavioural Observations

Video recording was used for make continuous observations of the pen from sunrise (approximately 6 am in summer and 7 am in winter) to sunset (6 pm in winter and summer). Due to the lack of lights observations of feeding behaviour were not possible during the night. Observations were recorded at 10 minute intervals by stopping the video and counting number of cattle at the feed bunk (within one m), water trough (within one m), standing and lying. Walking and “other” were determined by viewing 15 seconds either side of the 10 minute time point and counting the number walking and undertaking activities classified as ‘other’ (grooming, interacting with another animal). Number ruminating was determined using the same interval, by randomly selecting 20 animals and counting the number...
ruminating. The proportion determined using this method was then multiplied to give total number ruminating.

6.2.4 Micrometeorological Measurements

Full details of the micrometeorological and emissions measurements can be found in the General Methodology (Section 3.2). Briefly, a micrometeorological approach based on measurements of gas concentrations with open-path gas analysis (tunable diode lasers and FTIR) and a backward Lagrangian Stochastic (bLS) dispersion model was used to calculate CH$_4$, NH$_3$, N$_2$O and CO$_2$ emissions. Twenty four hour measurements (15 minute averages) were available for all gases. Full emissions profiles of all gases are shown in Appendix 9.5. However, observations from the Northern Site during summer 2008 were limited due to a large number of data points being excluded in the filtering process associated with bLS calculations (primarily related to wind speed and turbulence).

6.2.5 Environmental Data

Environmental conditions were monitored in conjunction with the micrometeorological measurements. The key environmental parameter recorded and evaluated in this study was air temperature, due to its effect on animal behaviour, and on emissions of NH$_3$ and CH$_4$ from manure. Wind speed, direction and turbulence statistics were also monitored, however these are not reported in the context of this study.

6.2.6 Statistical Analysis

Due to lack of light overnight in the feedlots, behaviour observations were recorded during daylight hours, approximately 6 am to 6 pm, with some deviation from this depending on light and environmental conditions. Thirty minute averages of fluxes, temperature and number of animals performing a specific behaviour (feeding, ruminating etc) based on observations, were calculated. Observations were initially undertaken at 10 minute intervals, as intervals of less than 15 minutes were found to be more precise than longer intervals (30 or 60 minutes; Mitlohner et al. 2001). However, flux data was calculated only as 15 minute averages; therefore 30 minute intervals were used to evaluate the relationship between variables.

Each day was divided into four time periods, 6am-9 am, 9am-12 midday, 12 midday-3 pm and 3-6 pm in order to examine if fluxes and feeding behaviour during one part of the day were significantly different to others. One way ANOVA (Genstat v. 11, VSN International, UK) was used to examine the differences in fluxes associated with time of day. Spearman’s
Correlation (Genstat 11, VSN International, UK) was used to examine correlations between variables, focussing on relationships between feeding behaviour and emissions of CH\(_4\) and NH\(_3\), and the relationships between environmental temperature and emissions of these gases.

### 6.3 Results

#### 6.3.1 Northern Site- Winter 2007

Ammonia fluxes (Figure 6.1) increased from early morning to a peak around midday, then slowly decreased towards evening. CH\(_4\) fluxes showed a small peak in the early morning, followed by a decline around 10 am and increased to a large peak, almost double morning emissions, in the late afternoon. There was a significant \((P<0.001)\) difference in CH\(_4\) emissions between time periods. There was also a significant \((P<0.001)\) difference in NH\(_3\) emissions measured during each time period. Daily temperature was also significantly \((P<0.001)\) different between time periods. Temperature was lowest in the early morning \((12.1^\circ C)\) and highest in the early afternoon \((23.3^\circ C)\). Number of cows feeding (Figure 6.2) was low during the morning and increased rapidly when feed was delivered (around 11 am). Feeding activity continued throughout the afternoon. Number of cattle feeding was significantly \((P<0.01)\) different between time periods, on average most cattle were feeding during the late afternoon, despite the rapid increase in number feeding following feed delivery. Emissions of NH\(_3\) were most highly correlated (Table 6.1) with temperature \((0.77, P<0.001)\), followed by feeding \((0.30, \text{NS})\), although the correlation with feeding was not significant. Emissions of CH\(_4\) (Table 6.1) were most highly correlated with feeding \((0.66, P<0.001)\), and negatively correlated with number ruminating \((-0.51, P<0.05)\).

**Table 6.1** Correlation Matrix (and t-probabilities) for animal behaviour and greenhouse gas fluxes measured at the Northern Site during winter.

<table>
<thead>
<tr>
<th></th>
<th>Temperature</th>
<th>CH(_4)</th>
<th>NH(_3)</th>
<th>Feeding</th>
<th>Ruminating</th>
</tr>
</thead>
<tbody>
<tr>
<td>Temperature</td>
<td>1.00</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>CH(_4)</td>
<td>0.43*</td>
<td>1.00</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>NH(_3)</td>
<td>0.77***</td>
<td>-0.07NS</td>
<td>1.00</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Feeding</td>
<td>0.57**</td>
<td>0.66***</td>
<td>0.30NS</td>
<td>1.00</td>
<td>-</td>
</tr>
<tr>
<td>Ruminating</td>
<td>-0.17NS</td>
<td>-0.51*</td>
<td>-0.01NS</td>
<td>-0.75***</td>
<td>1.00</td>
</tr>
</tbody>
</table>

*NS Not Significant, * \(P<0.05\), ** \(P<0.01\), *** \(P<0.001\)
Figure 6.1 Emission of CH₄ and NH₃ (flux rates calculated to g/head/day) and air temperature (°C) during winter at the northern site. Error bars indicate LSD for significance at $P<0.05$ level.
Figure 6.2 Number of cattle feeding and ruminating over 12 hours during winter at the Northern Site. Error bars indicate LSD for significance at $P<0.05$ level.
6.3.2 Southern Site- Winter 2007

Ammonia emissions were significantly \((P<0.001)\) affected by time period, with the lowest flux in the early morning (227 g/head/day) and highest in the early afternoon (442 g/head/day). Similarly, temperature was highest in the early afternoon (15.6 °C) and lowest in the early morning (8.2 °C). Methane flux was significantly \((P<0.001)\) influenced by time of day (time period). Emissions of \(\text{CH}_4\) during winter at the Southern Site (Figure 6.3) are relatively consistent throughout the day, although a small peak is observed between 5 and 6pm (towards sunset). At the Southern Site, the number of cattle feeding had a small peak in the early morning and a larger peak (of longer duration) in early afternoon, probably stimulated by feed delivery (Figure 6.4). Number feeding and ruminating were both significantly \((P<0.001)\) different between time periods.

Ammonia flux (g/head/day) was most strongly correlated (Table 6.2) with temperature (0.87, \(P<0.001\)). Ammonia emissions were moderately associated with number feeding (0.60, \(P<0.01\)). Number feeding was also strongly correlated with temperature (0.67, \(P<0.01\)). Methane flux (g/head/day) was negatively correlated with number ruminating (-0.76, \(P<0.001\)).

Table 6.2 Correlation matrix (and t-probabilities) for animal behaviour and greenhouse gas fluxes measured at the Southern Site during winter.

<table>
<thead>
<tr>
<th></th>
<th>Temperature</th>
<th>(\text{CH}_4)</th>
<th>(\text{NH}_3)</th>
<th>Feeding</th>
<th>Ruminating</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Temperature</strong></td>
<td>1.00</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td><strong>(\text{CH}_4)</strong></td>
<td>0.13(^{\text{NS}})</td>
<td>1.00</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td><strong>(\text{NH}_3)</strong></td>
<td>0.87(^{***})</td>
<td>-0.59(^{\text{NS}})</td>
<td>1.00</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td><strong>Feeding</strong></td>
<td>0.67(^{**})</td>
<td>0.49(^{*})</td>
<td>0.60(^{**})</td>
<td>1.00</td>
<td>-</td>
</tr>
<tr>
<td><strong>Ruminating</strong></td>
<td>0.23(^{\text{NS}})</td>
<td>-0.76(^{***})</td>
<td>0.48(^{\text{NS}})</td>
<td>-0.17(^{\text{NS}})</td>
<td>1.00</td>
</tr>
</tbody>
</table>

\(^{\text{NS}}\)Not Significant, \(^{*} P<0.05\), \(^{**} P<0.01\), \(^{***} P<0.001\)
Figure 6.3 Emission of CH$_4$ and NH$_3$ (flux rates calculated to g/head/day) and air temperature (C) during winter at the southern site. Error bars indicate standard deviation in flux rate (g/head/day) or temperature. Error bars indicate LSD for significance at $P<0.05$ level.
Figure 6.4 Number of cattle feeding and ruminating over 12 hours during winter at the Southern Site. Error bars indicate LSD for significance at $P<0.05$ level.
6.3.3. Northern Site- Summer 2008

Due to the filtering criteria applied to flux data, CH$_4$ fluxes were calculated for only 48% of the observed time period (6 am to 6 pm). NH$_3$ fluxes were calculated for 64% of the observed time period. Fluxes of both CH$_4$ and NH$_3$ (Figure 6.5) were not significantly ($P>0.05$) different between time periods. CH$_4$ was highest during the early afternoon (73 g/head/day) and lowest in the late afternoon (43 g/head/day). NH$_3$ was highest in the early morning (403 g/head/day) decreasing to 281 g/head/day in the late afternoon. However the limited flux data available for this period makes it difficult to examine possible diurnal patterns (Figure 6.5).

Temperature was significantly ($P<0.001$) higher during the late morning and early afternoon (26.5 and 26.9 °C). Neither number feeding or ruminating (Figure 6.6) were significantly ($P>0.05$) different between time periods. Number feeding tended to be highest in the late afternoon and lowest in the late morning and early afternoon when temperatures were highest.

In general, there were no strong correlations (Table 6.3) between emissions of CH$_4$ and NH$_3$ and behavioural parameters. Methane was moderately (but significantly) correlated with NH$_3$ (0.63, $P<0.05$), but this is likely to be due to the limited data set available. Number feeding was strongly negatively correlated with temperature (-0.83, $P<0.001$).

Table 6.3 Correlation matrix (and t-probabilities), for animal behaviour and greenhouse gas fluxes measured at the Northern Site during summer 2008

<table>
<thead>
<tr>
<th></th>
<th>Temperature</th>
<th>CH$_4$</th>
<th>NH$_3$</th>
<th>Feeding</th>
<th>Ruminating</th>
</tr>
</thead>
<tbody>
<tr>
<td>Temperature</td>
<td>1.00</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>CH$_4$</td>
<td>-0.34 $^{NS}$</td>
<td>1.00</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>NH$_3$</td>
<td>-0.36 $^{NS}$</td>
<td>0.63 $^{*}$</td>
<td>1.00</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Feeding</td>
<td>-0.83 $^{***}$</td>
<td>0.38 $^{NS}$</td>
<td>0.34 $^{NS}$</td>
<td>1.00</td>
<td>-</td>
</tr>
<tr>
<td>Ruminating</td>
<td>-0.28 $^{NS}$</td>
<td>0.38 $^{NS}$</td>
<td>0.08 $^{NS}$</td>
<td>0.02 $^{NS}$</td>
<td>1.00</td>
</tr>
</tbody>
</table>

$^{NS}$Not Significant, $^{*}P<0.05$, $^{**}P<0.01$, $^{***}P<0.001$
Figure 6.5 Emission of CH₄ and NH₃ (flux rates calculated to g/head/day) and air temperature (°C) during summer at the northern site. Error bars indicate LSD for significance at $P<0.05$ level.
Figure 6.6 Number of cattle feeding and ruminating over 12 hours during summer at the Northern Site. Error bars indicate LSD for significance at $P<0.05$ level.
Chapter 6. Behaviour and Emissions

6.3.4 Southern Site- Summer 2008

Methane emissions from the Southern Site during summer demonstrate a small peak during the morning, and increase from midday (Figure 6.7). Methane flux was significantly \((P<0.001)\) higher during the late afternoon (93 g/head/day) and lowest during late morning (63 g/head/day). Ammonia flux increased, reaching a peak around midday and decreasing towards evening. Significantly differences in \(\text{NH}_3\) flux were observed between time periods, peaking in the early afternoon (173 g/head/day). At the Southern Site, during summer, number feeding exhibits a peak in the early morning, and again in the mid to late afternoon (Figure 6.8). Number ruminating exhibits less distinct peaks in the late morning and the late afternoon. Number feeding and ruminating did not significantly \((P>0.05)\) vary by time of day.

Emissions of both \(\text{CH}_4\) and \(\text{NH}_3\) were not strongly correlated with animal behaviour during summer at the Southern Site (Table 6.4).

Table 6.4 Correlation matrix (and t-probabilities), for animal behaviour and greenhouse gas fluxes measured at the Southern Site during summer

<table>
<thead>
<tr>
<th></th>
<th>Temperature</th>
<th>(\text{CH}_4)</th>
<th>(\text{NH}_3)</th>
<th>Feeding</th>
<th>Ruminating</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Temperature</strong></td>
<td>1.00</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td><strong>(\text{CH}_4)</strong></td>
<td>0.18*</td>
<td>1.00</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td><strong>(\text{NH}_3)</strong></td>
<td>0.02 NS</td>
<td>0.32**</td>
<td>1.00</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td><strong>Feeding</strong></td>
<td>0.29 NS</td>
<td>0.17 NS</td>
<td>0.05 NS</td>
<td>1.00</td>
<td>-</td>
</tr>
<tr>
<td><strong>Ruminating</strong></td>
<td>-0.34 NS</td>
<td>-0.12 NS</td>
<td>0.36 NS</td>
<td>-0.60**</td>
<td>1.00</td>
</tr>
</tbody>
</table>

*NS Not Significant, * \(P<0.05\), ** \(P<0.01\), *** \(P<0.001\)
Figure 6.7 Emission of CH₄ and NH₃ (flux rates calculated to g/head/day) and air temperature (°C) during summer at the southern site. Error bars indicate LSD. for significance at $P<0.05$ level.
Figure 6.8 Number of cattle feeding and ruminating over 12 hours during summer at the Southern Site. Error bars indicate LSD. for significance at $P<0.05$ level.
6.5 Discussion

Anecdotal evidence suggests that greenhouse gas fluxes (CH$_4$ and NH$_3$) from grazing and lot fed livestock (sheep and cattle) are associated with animal behavioural patterns (Flesch et al. 2007; Lockyer 1997; Loh et al. 2008). From the two feedlots measured, stronger relationships between behaviour and emissions were observed during winter, than during summer, when poor correlations were observed. Distinct diurnal patterns were observed for NH$_3$ emissions, and were consistent between sites and seasons. In contrast, CH$_4$ emissions exhibited much more variable daily patterns. Environmental conditions are likely to have contributed both to the different flux patterns observed between summer and winter, and the changes in animal feeding behaviour.

6.5.1 Cattle Behaviour

Animal feeding behaviour was more different between seasons than between sites. The delivery of fresh feed has been found to be a stimulant of feeding behaviour in housed dairy cattle, and feedlot cattle (DeVries et al. 2003; Fell and Clarke 1993). During winter at the Northern Site, number of cattle feeding increased rapidly following feed delivery, and continued at a similar level during the afternoon. Similarly, at the Southern Site number of cattle feeding also increased following the delivery of fresh feed. In winter number feeding peaked at around 2 pm, and then decreased back to similar levels to the early morning by the time observations concluded at 6 pm. However, in summer (at both sites) feeding behaviour exhibits two distinct peaks associated with the early morning and late afternoon. Ray and Roubicek (1971) noted two distinct peaks of eating activity, cattle started eating around sunrise, but feeding ceased by mid morning. They noted that the major period of consumption occurred in the afternoon and early evening. These two peaks were more pronounced during summer than in winter. Ray and Roubicek (1971) also noted that eating between 6pm and midnight was increased in steers during summer. Two distinct peaks of feeding activity were also noted by Gonyou and Stricklin (1981) who also observed a lesser peak during the night and by Hoffman and Self (1973). Gonyou and Stricklin (1981) noted that feeding behaviour appeared to be associated with sunrise and sunset and relatively independent of feed management. These observations are consistent with the summer observations of this study.

It appears environmental conditions and heat load have a significant effect on the feeding behaviour of feedlot cattle (Castaneda et al. 2004). During winter, when air temperatures are moderate, timing of feed delivery during the late morning stimulates feeding. During summer, cattle choose to feed during periods of cooler air temperature. In winter, the highest frequency of number eating appears to be associated with the time the feed in placed in the...
bunk however in summer, placing feed in the bunk during the afternoon does not appear to illicit a response (Ray and Roubicek 1971). We would therefore expect that there would be a difference in diurnal emissions pattern, particularly of CH$_4$, between summer and winter.

### 6.5.2 Fluxes

Over the periods observed, both within day and within season, a distinct diurnal pattern was observed for NH$_3$ flux. Fluxes of NH$_3$ increased from the early morning, peaking during the early afternoon, NH$_3$ fluxes were significantly different between time periods (excluding the Northern Site summer 2008). The diurnal pattern of NH$_3$ emissions recorded here are similar to those reported by Flesch et al. (2007) and Loh et al. (2008). At each site NH$_3$ flux peaked during the early afternoon, despite a significantly higher magnitude of emissions during winter at the Southern Site. Methane fluxes were more variable; however (excluding the Northern Site summer 2008) fluxes were consistently significantly ($P<0.05$) different between time of day, tending to be lowest during the late morning, and highest during the late afternoon. However the timing of depressions and peaks is more variable than the consistency observed for NH$_3$. The variability of daily emission patterns for CH$_4$ is consistent with the results of Lockyer (1997) who observed very different patterns, particularly in the period between sunset and sunrise. However, Lockyer (1997) observed a tendency for CH$_4$ emissions to increase towards sunset, similar to the observations reported here.

### 6.5.3 Correlations

Ammonia emissions follow a consistent diurnal pattern at both sites and in both seasons. The same pattern is reported by a number of other sources- including (Flesch et al. 2007). Although there appears to be an association between number feeding and NH$_3$ during the Winter measurement campaigns, this relationship is likely to be primarily due to the association of both emissions and feeding behaviour with temperature and the timing of feed delivery. During winter, feed delivery stimulates feeding, which continues throughout the afternoon, this coincides with the peak in NH$_3$ emissions (and temperature). During summer, feed delivery does not stimulate feeding to the same extent (behaviour is affected by temperature). Despite these differences in feeding behaviour, the diurnal pattern of NH$_3$ emissions is consistent between seasons and sites. For all seasons the highest correlation is observed between temperature and NH$_3$ emissions. As reported by Van Haarken et al (2008) a 24 hour pattern exists for NH$_3$, but this is not closely related to the feeding schedule (for all seasons). They attribute this to the fact that emissions of NH$_3$ are not a direct animal process. These results suggest that in contrast to the suggestion of Flesch et al. (2007) and Loh et al. (2008) who suggest that the increase in emissions is associated with increased animal activity
and decreases with decreasing activity in the afternoon. Flesch et al. (2007) suggest that time of day is the predominant determinant of NH$_3$, as the pattern is consistent (with some modulation with temperature and wind speed) between days and seasons. The observation that time of day is the predominant determinant of NH$_3$ is supported by the low correlation between temperature and NH$_3$ emissions during summer at the Southern Site, despite NH$_3$ flux maintaining a similar diurnal pattern to winter measurements. However, the lack of animal activity during the early afternoon during these summer measurements suggests that environmental conditions are more likely to be the primary determinant of diurnal patterns of NH$_3$ emissions.

The low correlations between emissions of CH$_4$ and rumination are unexpected. Increased number of animals ruminating (and therefore releasing CH$_4$ via eructation) would be expected to be associated with increased emissions. However, moderately negative correlations were observed between CH$_4$ emission and number ruminating during winter at both sites, and a weak negative correlation at the Southern Site during summer. In general, a positive correlation was observed between CH$_4$ flux and number feeding. The correlation between CH$_4$ and number feeding during winter (CH$_4$ increasing with feeding) is consistent with the observations of Lockyer (1997) and Murray et al. (1999); this was attributed by both authors to accumulation of ingesta into the rumen and displacement of CH$_4$ with the introduction of feed material. Murray et al. (1999) also observed that an increase in ruminating (or idling) after sunset, as the majority of feeding occurs during daylight hours, was associated with a decrease in emissions (as rumen contents decreased). This observation would be consistent with the lowest measured emissions occurring just prior to feed delivery.

6.5.4 Other Considerations

Behavioural observations were not able to be conducted over night; therefore definite conclusions cannot be drawn about the patterns of emissions during this period. However, in both winter and summer emissions tend to peak in the early evening (slightly later in summer). During winter these emissions drop back to “daytime” levels over night, however in summer emissions continue to be high until the early morning. Although, based on the results of this study, this is unlikely to be related directly to feeding behaviour it is possible that alterations in the timing of feeding bouts associated with seasonal differences in temperature and photoperiod may be contributing to these differences. Gonyou and Stricklin (1984) observed a significant period of feeding near midnight during winter in lot fed cattle, however this decreased in duration (and intensity) as photoperiod increased. The longer day length in summer will have resulted in delayed timing of the feeding bout associated with sunset. This
would potentially increase emissions during the early evening, but is unlikely to explain the maintenance of higher emissions until early morning.

However, the increase in CH$_4$ emissions during the late afternoon and evening may also be related to micrometeorological conditions. When using micrometeorological measurements, a commonly observed phenomenon is a period of atmospheric boundary layer stability after sunset (low turbulence and wind movement; Griffith et al. 2002). These results in concentration build up under the inversion layer, and can result in very low accuracy in calculations of fluxes using dispersion modelling (Flesch et al. 2007). Laubach and Kelliher (2004) examined concentrations and CH$_4$ emissions estimates from various periods over the day (measured using micrometeorological methods) and determined increased concentrations (from 250 g/head/day to 400 g/head/day) associated with the change from unstable day conditions to stable night time conditions. This trend appears to be more clearly demonstrated during summer than winter (see appendix), and is potentially associated with differences in environmental conditions, such as wind speed and turbulence. Yi et al. (2001) report that the boundary layer is often not clearly defined during heavy cloud or rainfall, and demonstrates differences between the height of the stable boundary layer between seasons. Similarly, Dayan and Rodnizki (1999) observed the depth and frequency (of occurrence) of the boundary layer to be greater in summer. Both these factors are possible explanations for the differences in “over night” emissions patterns between seasons.

This is additionally a potential source of the diurnal emissions pattern observed for NH$_3$. Temperature and air inversion create a boundary layer, which may hold NH$_3$ near the surface over night reducing volatile NH$_3$ in the atmosphere until air temperature warms and more turbulence is created by less stable conditions. Further, high concentrations of NH$_3$ near the surface of the manure pad will impact rate of diffusion of NH$_3$ from the manure pad to the atmosphere. However further examination would be required to draw conclusions about sources of diurnal variation in NH$_3$ emissions.

Errors associated with this methodology (as evidenced by the reduction in flux data from the Northern summer campaign) may influence reported diurnal patterns as well as average emissions estimates. Further, although video recording and assessment of behaviour at ten to 30 minute intervals has been found to be relatively accurate for longer term behaviours (Hao et al. 2001a). However, short term behaviours may not be observed. Additionally, video recording techniques are commonly used for housed animals. Although feedlot animals are confined to a relatively small area the placement of the camera resulted in incomplete coverage of the pens (roughly 95% of the pens could be seen on the video at any time). The
ability to easily distinguish between idling and ruminating cattle in the video recording was limited at times. Further development of the methodology, perhaps utilising smaller numbers of cattle and smaller areas in an artificial feed yard situation may enable increased ability to draw conclusions.

6.6 Conclusion

Diurnal pattern of NH₃ emissions is more strongly associated with environmental conditions than animal activity, as evidenced by the similarity in emissions despite seasonal differences in animal feeding behaviour. In contrast, CH₄ emissions are more closely associated with feeding behaviour (increases in number feeding) than ruminating; however the correlation is greater in summer than winter. Feeding behaviour explains only a small amount of observed diurnal patterns of emissions. If “snap-shot” sampling methods are going to be utilised, consideration needs to be given to changes in emissions with time of day. When unfavourable wind conditions result in limited data points which can be used in the estimation of emissions, calculated values may be significantly impacted by the time of day in which these data points are centred. Clustering of usable data in periods of high or low emissions may result in artificial inflation/deflation of daily average emissions. This is potentially a significant limiting factor for the use of micrometeorological methods over short term measurement campaigns.
Chapter 7. General Discussion

7.1. Introduction

This thesis examined greenhouse gas emissions from beef feedlot operations in Australia, including investigating differences in magnitude of emissions between sites and seasons, role of intake behaviour in influencing diurnal emissions fluctuations and the application of a modelling methodology for prediction of emissions. But why are emissions from feedlot systems important or different from other livestock production systems? Why do we need to account for emissions, and what is the role of predictive equations?

In this Chapter the research findings are contextualised in the current and future policy situation regarding greenhouse emissions from agricultural systems and from high to low level; in order to address the above questions. Opportunities for mitigation and quantification of emissions are outlined and areas for future exploration in modelling approaches discussed.

7.2 Feedlot Production

The principle of feedlot operation is the ability to produce a consistent product efficiently. Although the majority of Australian beef production occurs on pasture (particularly at the breeding, cow/calf pre finishing stages) cattle are being increasingly finished using feedlot operations in response to consumer demand. The ability to control quality and quantity of finished cattle throughout the year and produce faster growth rates and finishing makes feedlot operations very efficient (Charmley et al. 2008; Pritchard 2006; Stackhouse et al. 2011). Approximately 65% of Australian beef is exported annually, although domestic production is still the largest single market (ALFA 2008). Another significant output of feedlot systems is nutrients (N and phosphorus) and greenhouse gases (CH\textsubscript{4}, N\textsubscript{2}O and the indirect greenhouse gas NH\textsubscript{3}); which effectively represents an inefficiency or leakage from the system.

Characteristics of feedlot production mean that the emissions profile is considerably different to other cattle feeding operations, including grass fed, rangeland and pastoral beef and dairy systems. The concentrated nature of a feedlot, with large numbers of cattle confined in a relatively small area makes the site a significant point source of enteric CH\textsubscript{4}. Built up manure, in pens, effluent systems and compost piles is a source of CH\textsubscript{4}, N\textsubscript{2}O, NH\textsubscript{3} and volatile compounds. A considerable part of the efficiency of feedlot operations results from the ability to supply rations balanced for energy, protein and other dietary requirements in the quantities
required for growth. Feedlot rations are typically higher grain and higher energy than a pasture based ration, resulting in not only faster growth rates, but improved efficiency.

In Australian feedlots, cattle are typically fed to meet domestic or export markets (which can be long or short fed), cattle will be yarded within the feedlot based on breed, age, weight and likely market. Cattle spend between 50 and 120 days on average in a feedlot, dependent on market. The concentrated nature of feedlot operations results in a build up of manure, which, like an individual dung or urine patch, can provide a significant source of CH$_4$, N$_2$O and NH$_3$. However, emissions from the manure pad differ from those from a single dung pat or urine patch due to built up manure and manure of varying ages (Miller and Berry 2005; Miller et al. 2003; Stackhouse et al. 2011).

Additionally, feedlot rations differ from grazing systems with the addition of products (e.g. oils, rumen modifiers) which are commonly reported to reduce emissions, and proposed as mitigation options. These options can be applied more easily under the controlled feeding environment of the feedlot, where feed is provided daily, than in a grazing (intensive or extensive) system. Lipid based supplements (commonly recycled vegetable oil) are commonly utilised in feedlot rations, which may contribute to lower emissions. However, there may be limited potential for further emissions reduction, due to the higher basal efficiency of the system (compared with grazing operations).

There may also be some differences in the feeding behaviour of feedlot cattle, particularly in winter where timing of feed delivery has a greater impact on timing of feeding (number of cattle feeding immediately following feed delivery declines sharply in summer, but remains higher during winter). However, similar to grazing systems, feedlot cattle demonstrate feeding bouts near sunrise and sunset in both seasons. However, this may influence diurnal emissions profile in relation to a grazing animal, and have implications for snapshot measurements of enteric emissions, and for micrometeorological measurements, where large amounts of data can be filtered based on environmental conditions.

7.3 The Need for Accounting
Quantification of emissions from feedlot systems is necessary due to requirements for national auditing and reporting of emissions, under the Kyoto Protocol and international agreements (administered by the IPCC) and for use in a potential emissions trading scheme. Despite the current debate around the application of a Carbon Tax and/or price for carbon credits in Australia, consensus has been to exclude Agriculture from a direct Carbon Tax in the immediate future, although the Carbon Farming Initiative will allow agricultural
producers to utilise mitigation practices to generate carbon credits. There is increasing public awareness and demand for low emissions, low pollution products, which is likely to continue in the future.

The increased requirement for accounting due to government requirements notwithstanding, accurate accounting for emissions may be a useful tool in benchmarking the efficiency of the system in terms of energy and protein use. CH₄ is a loss of energy, which could potentially be utilised as energy for growth or production (although there is little concrete evidence that decreasing CH₄ emissions increases growth or milk production). Whilst N/protein is commonly one of the most expensive ingredients in feeds, and often slightly overfed (to avoid limitations to growth). High levels of N, in excess of requirements, result in the animal expending energy to process the excess as well as excretion of significant quantities of N. The ability to accurately account for emissions could have significant benefits for operators in terms of examining potential changes to their production system.

**7.3.1 Measurement as an Accounting Method**

The ideal way of accounting for emissions is direct measurement. Measurements of enteric CH₄ were originally undertaken using calorimetric methods, in order to examine the loss of feed energy from CH₄. Calorimetric approaches continue to be used for experimental purposes; however they have limited applicability under commercial conditions due to requirements for enclosure of animals (and availability of appropriate calorimetric facilities). Traditional measurements of nitrogenous gases has utilised chamber measurements, static or dynamic, or various sizes. These chambers are used to contain subsamples of manure/soil, placed over artificial feed yard surfaces, or for *in situ* measurements. More recently techniques which enable animals to remain under grazing/unrestrained, the SF₆ technique and measurement using tuneable diode lasers (or similar) accompanied by calculations based on wind statistics (commonly known as micrometeorological methods).

It can be considered that the primary difference between measurement approaches is between measurement of individual components or the whole system. The application of component measurement methods, such as calorimetry and SF₆ under commercial situation is impractical, given the requirements for infrastructure, training of animals, and in the case of SF₆ dosing with a sulfur-hexafluoride bolus, which has a withholding period prohibitive for commercial meat production. The use of small chambers and short term measurements for nitrogenous gases can result in inability to capture spatial and temporal variation which can significantly impact on estimated emissions. For NH₃ volatilisation, due to its nature and reversible physical-chemical process, micrometeorological techniques, are needed to measure losses.
from the whole system. From this perspective, whole of system approaches which can provide indirect measurements (limiting impacts on animal behaviour and intake and encompassing spatial variability within soil or manure) should enable more accurate accounting.

Although not a primary aim of this thesis, the use of micrometeorological measurements in the comparison between modelled and measured emissions highlighted one of the major issues with micrometeorological measurements - the loss of useable observations under unfavourable environmental conditions. In contrast to a calorimetric chamber, where conditions are closely controlled, micrometeorological measurements rely on wind speed, direction and turbulence to be between set parameters to enable accurate estimation of emissions from measured concentration. This effect is demonstrated clearly in the comparison between the numbers of data points used in the calculation of average emissions from the Northern Site summer 2008 measurement campaign, with the remainder of the campaigns. The reduced data points resulted in much more variable average emissions (SE is more than 20 times higher) for all reported gases, making it difficult to draw conclusions when attempting to quantify emissions. Increased variability in measurement using these types of approaches may also make the assessment of mitigation options difficult. Where expected reductions are less than the variability observed, it may be difficult to determine either a statistically or practically significant value. However, this approach remains the most viable for measuring whole system emissions, provided the limitations are understood.

The effect of reduction in useable data under unfavourable environmental conditions is compounded when emissions show a distinct diurnal pattern. Although this study demonstrated low correlations between animal behaviour and emissions there was observed to be a distinct diurnal pattern in emissions of both NH₃ and CH₄. The pattern of NH₃ emissions was more consistent between seasons and sites, but CH₄ was more variable. The variation in CH₄ emission could potentially be associated with differences in feeding times due to climatic conditions and timing of feed delivery. However, where the usable data points are clustered around periods of peaks or troughs in emission (as they were during the Northern Summer 2008 campaign) artificial inflation or deflation of average emissions can occur.

Although measurements of emissions from agricultural systems would enable accurate inventories the cost can be prohibitive. Further, the challenges associated with all measurement methods makes application difficult, particularly under commercial situations where value for money (in terms of accurate emissions obtained for investment in measurement equipment/ expertise) is likely to be as important to the enterprise as obtaining accurate estimates of emissions.
7.3.2 Modelled Emissions Estimates for Accounting

The impracticality of direct measurement of emissions from each individual feedlot operator means an indirect method of estimating emissions is required for accounting purposes. A standard methodology is prescribed by the IPCC (for various industries); however, where agriculture is a key source category (such as Australian and New Zealand) the use of a country specific methodology is recommended. For feedlot cattle in Australia, CH$_4$ estimates are based on the equation developed by (Moe and Tyrrell 1979) and on volatile solid excretion (combined with a country specific manure CH$_4$ emissions factor). Emissions of nitrogenous gases are predicted utilising an N excretion estimate based on SCA guidelines and country specific emission factors (for N$_2$O and NH$_3$). A similar approach is taken for N emissions under IPCC guidelines, and the same emissions factors are used in both the IPCC and Australian methodologies.

A key aspect of the evaluation was to examine the ability to utilise data available from commercial feedlot operations, reducing requirement for additional recording for accounting purposes. Feedlot cattle are managed primarily on a pen or lot basis, weights are estimated on arrival based on the weight of the transport vehicle and number of cattle, feed is delivered and rationed on a pen basis and growth is estimated by a pen basis. There appears to be a difference in the reliability of emissions estimates between individually measured values and measured values from the whole system. The more detailed CHO based equations Moe and Tyrrell (1979) and Ellis et al. (2007) show a closer relationship to emissions measured on an individual basis, presumably due to increased specificity compared with the Tier II approach. However, for emissions measurements based on the whole feedlot the Tier II approached showed (overall) the closest relationship between measured and predicted emissions.

However, over-estimation with the CHO equations and under estimation with the Tier II equation was consistent between published and measured data (Chapters 4 and 5). In terms of Tier II, this is likely to be due to the range of grain concentrations observed in feedlot rations, which can limit the applicability of a single emissions factor. Kebreab et al. (2006) suggests that with ration specific emission factors the Tier II equation may provide reasonable estimates of emissions. Although this equation demonstrated the closest relationship (overall) with measured emissions the differences observed in accuracy between sites and seasons suggest that it needs to be modified in order to improve emissions estimates.

An issue with all the current approaches for estimating enteric CH$_4$ is that despite the ability of the CHO based equations to account for differences in ration composition, there is limited ability to account for the use of mitigation options, such as lipids or dietary additives. In
Chapter 4, an equation developed by Ellis et al. (2009) which included fat as a parameter was evaluated; however accuracy proved to be reduced compared with the other options. Future accounting methodologies need to be not only accurate, but able to account for reductions associated with mitigation. It is clear further development is required if an empirical modelling approach is to be used in preference to a dynamic/ rumen simulating approach.

The failure of predictive approaches for N (NH₃ and N₂O) gas emissions is due in the most part to the lack of consideration of the impact of environmental conditions on emissions. Emissions of nitrogenous gas from the manure pad is influenced not only by quantities of N excreted and the form of N (urine or faeces) of the current cattle, which is accounted for in the modelling approach, but accumulated manure, substrate pH, moisture content, organic matter and temperature. However, it should be noted that the measurement campaigns reported here and used in the application of the model are short term measurements. It is possible that the N emissions factors (0.3 for NH₃ and 0.02 for N₂O) may be appropriate when considered annually (or over the time which an individual pen of animals spend on feed), although they are inaccurate for short term measurements.

7.4 Implication of Inaccurate Accounting

This thesis demonstrates that there are issues with over estimation of both enteric (and manure pad) CH₄ and N₂O, and underestimation of NH₃ under the current National Inventory approach (AGO 2006). Despite agriculture being excluded from a Carbon Trading scheme indefinitely, there will still be pressure on agricultural industries (political and social) to produce low emissions products. However, this will need to occur within a carbon constrained economy. The RIRDC (Jiang et al. 2009) recently examined the impact of an emissions trading scheme on agricultural industries, under 2 prices for carbon ($25 and $50/ tonne), and scenarios of full agricultural inclusion and indirect impacts (agriculture not included). Key aspects of the analysis which may impact the productivity of feedlot operations included a projected increased in the cost of purchasing cattle of between 1.28 and 2.35 % for a full inclusion scenario (at $25 and $50/ tonne respectively) or 0.13 and 0.25% for a non inclusion scenario. They determined the primary increase in cost to be related to the introduction of abatement methods, or purchase of permits. Under a full inclusion scenario a beef farm is suggested to have the biggest fall in farm income under both carbon prices (63 and 125% respectively), compared to grain, sheep, horticulture and mixed farming. Under a non inclusion scenario, agricultural industries would still be affected due to changes in prices of inputs including fuel, fertiliser, transport and other goods and services. The RIRDC (Jiang et al. 2009.) suggest that the cash income of a beef operation would decrease by 2.7% under a
non-inclusion scenario. The impacts of increased costs of fuel and transport would be likely to have a greater impact on a feedlot operation than an extensive beef operation, given the high proportion of fuel and transport costs in operating costs.

Over-estimation of emissions will increase the pressure to reduce emissions, substantially easier to achieve actual reductions. Under estimation of emissions will actually make it harder to achieve actual reductions, whilst will not encourage the application of mitigation practices, or changes to current practice, particularly if this adds an additional cost to the system.

### 7.5 Opportunities

The nature of the feedlot system means that application of mitigation strategies (such as lipids and rumen modifying additives) can be easily utilised in the production system; indeed lipids and fats as well as additives are commonly already added to feedlot rations to increase energy density. This thesis does not aim to fully explore mitigation options; however the need to include reductions in emissions associated with the use of mitigation options in methodologies should be highlighted.

Of equal importance to the reduction of emissions is accurate accounting for emissions from the system. As described, the cost of measurements using calorimetric and micrometeorological methods can limit use under commercial conditions. This results in short term measurement periods, limiting the ability to evaluate emissions over whole growth periods and examine seasonal differences. Denmead et al. (2008) report on emissions measured using a fixed trace gas station (Ecotech Pty Ltd), which utilises a point measurement rather than a path length measurement of concentration. The benefit of this approach is that once set up, the unit can be left in place for long periods enabling long term measurements and assessment of changes in emissions. The unit utilised by (Denmead et al. 2008) contained a CH₄/non CH₄ hydrocarbon analyser, an NH₃ chemiluminescence analyser and a NO/NO₂/NOₓ chemiluminescence analyser allowing measurement of the greenhouse gas of interest. As with other micrometeorological measurements, this equipment is expensive and requires some expertise to apply and use correctly. However, this may provide a tool for researchers to monitor long term emissions and correlate these with animal production, rations and climatic conditions. This is particularly relevant for measurements of the nitrogenous gases, as long term measurements with automatic dynamic chambers are impractical in occupied pens. New high sensitivity, low maintenance and more reliable sensors are emerging which will make long term measurement more feasible.
More precise measurement of enteric \( \text{CH}_4 \) can be obtained from direct measurements of rumen gas production; however this is difficult without the use of fistulated cows in an intensively managed situation. More recently wireless sensors (boluses) have been utilised in ruminant nutrition, measuring pH, temperature and pressure telemetrically and in real time (Laporte-Uribe et al. 2010) and can be used in non fistulated cattle (Lin 2009). Lin (2009) evaluated the Kahne bolus (which is commercially available) for effectiveness in monitoring rumen pH under both indoor and outdoor feeding situations and determined they were effective. They also discuss the use of a pressure sensing bolus for measurement of changes in rumen pressure (Lin 2009). Rumen boluses which measure pH are commercially available in New Zealand (Kahne Ltd) and the United Kingdom (WellCow\textsuperscript{TM}) targeted at primarily dairy farmers for the management of subclinical acidosis, however with further development they may have application for \( \text{CH}_4 \) production. The recording span of the bolus is up to 100 days (WellCowLtd 2009), making it suitable for monitoring cattle over the finishing period in a feedlot.

Montanholi et al. (2008) draw on the concept that more efficient dairy cattle demonstrate both lower heat loss and lower \( \text{CH}_4 \) production and examined the use of infrared thermography (which has also seen recent evaluation in heat stress research) in prediction of \( \text{CH}_4 \) production. Infra-red measurements of the difference in temperature between the left and right flank showed correlations of between 0.53 and 0.77 dependent on time of day (time following feeding). In the periods after feeding temperature difference explained 60% of the variation in \( \text{CH}_4 \) emissions. This approach would be non-invasive, relatively inexpensive and easily applied, which would enable easy adoption under commercial feedlot operations. However, significant further research and demonstration of effectiveness is required before it could be applied to large numbers of cattle or for the purposes of accounting.

For a feedlot system, where the majority of “measurements” are conducted on a pen basis (estimates of weight, intakes) modelling emissions on a pen basis (whilst still accounting for the potential variability between individual animals) would have more relevance in terms of the production system. Further, drawing links between cattle class and emissions would provide a useful tool in terms of accounting. Within the feedlot system cattle are fed for a specific market (generally domestic, short fed export or long fed export). If a greenhouse gas emission value could be applied to each of these classes numbers in each category reaching the processor (detailed are reported back to the feedlot) could be used as a proxy for detailed inventory calculations. The increase in accuracy in the validation study (Chapter 4) for the carbohydrate based equations when using the class-based-intake rather than the measured intake indicates the possibility for further development in this area.
7.6 Implications

This study highlights the contributions of the various gases to total feedlot emissions. In contrast to expectations, feedlot emissions are dominated by nitrogenous gases, in the form of $\text{NH}_3$, rather than $\text{CH}_4$. Based on the feedlots measured, an “average” feedlot steer in a feedlot is predicted to emit 183 g $\text{CH}_4$, 62 g $\text{NH}_3$ and 6.5 g $\text{N}_2\text{O}$ daily. However measured emissions reported here and by Chen et al. (2009) indicate $\text{CH}_4$ emissions closer to 119 g/head/day (94.0 to 130.3), $\text{NH}_3$ emissions closer to 154 g/head/day (94 to 305 g/head/day) and $\text{N}_2\text{O}$ emissions of 3 g/head/day (0.1 to 5.7 g/head/day). On an individual site basis (Northern Australian feedlot compared with Southern) an average animal (and manure contribution) at the Northern Site is predicted to emit 187 $\text{CH}_4$, 62 g $\text{NH}_3$ and 6.5 g $\text{N}_2\text{O}$ daily, whilst measured emissions from this site suggest each animal contributes 132 g $\text{CH}_4$, 123 g $\text{NH}_3$ and 3.6 g $\text{N}_2\text{O}$. From the Southern Site, predicted emissions were 181 g $\text{CH}_4$, 61.2 g $\text{NH}_3$ and 6.4 g $\text{N}_2\text{O}$ per head daily. Whereas measured emissions are closer to 110 g $\text{CH}_4$, 178 g $\text{NH}_3$ and 2.6 g $\text{N}_2\text{O}$. From both sites, $\text{NH}_3$ volatilisation is considerably higher than predicted, however daily emissions of $\text{CH}_4$ and $\text{NH}_3$ are more similar from the Northern Site (132 and 123 g/head/day) compared with the Southern Site, where emissions of $\text{NH}_3$ are considerably higher than $\text{CH}_4$ (178 and 110 g/head/day).

Denmead et al. (2008) assumed potential emissions of indirect $\text{N}_2\text{O}$ from deposition of $\text{NH}_3$ to be equal in magnitude to direct $\text{N}_2\text{O}$, amounting to approximately 3 g/head/day. Using the same approach as (Denmead et al. 2008) indirect $\text{N}_2\text{O}$ is calculated at 1.25% of deposited N, and that all NH$_3$ is assumed to be deposited, measured emissions of NH$_3$ account for an extra 1.9 g indirect $\text{N}_2\text{O}$/head/day. Nitrogen is predicted to contribute an average of 40% of total CO$_2$-e and up to 52% of total CO$_2$-e (predicted contribution range from 34 to 39% total CO$_2$-e). Based on average values, the contribution of N to total emissions is slightly greater at the Northern Site (where NH$_3$ emissions are included as indirect $\text{N}_2\text{O}$). N gases contribute 40% to total CO$_2$-e at the Northern Site, compared with 39% at the Southern Site. This similarity arises from the higher direct $\text{N}_2\text{O}$ loss measured at the Northern Site, resulting in 1.8 kg/head/day CO$_2$-e from N and 2.8 kg/head/day from $\text{CH}_4$. At the Southern Site, the lower direct $\text{N}_2\text{O}$ emissions (2.6 g/head/day) and indirect $\text{N}_2\text{O}$ (from NH$_3$) contribute 1.4 kg/head/day CO$_2$-e, whilst $\text{CH}_4$ contributes 2.3 kg/head/day to total CO$_2$-e.

Although not a considerable difference in terms of contribution of CO$_2$-e, the higher NH$_3$ emissions on a daily basis recorded at both sites, but in particular at the Southern Site, indicate that mitigation strategies should be evaluated and applied not only for $\text{CH}_4$ emissions. This contribution of N to total emissions demonstrates that whilst mitigation strategies could
be applied effectively to CH₄, between 40 and 50% of total CO₂-e is contributed by N, and a substantial amount of this through indirect emissions of N₂O, coming from NH₃. Therefore management strategies to reduce excess N excretion and volatilisation could have a significant impact on total CO₂-e emissions from feedlots. However, this is based on the assumption (Denmead et al. 2008) that all volatile NH₃ eventually becomes a source of N₂O. Volatilisation, deposition and potential indirect N₂O emissions require further investigation from concentrated animal feeding operations.

As discussed in the literature review, feeding a high digestibility high grain ration has been observed to reduce emissions by a number of sources (Beauchemin and McGinn 2005; Boadi et al. 2004b), additionally additives such as oils and products which modify rumen fermentation (e.g. monensin) are commonly added to feedlot rations already. The results of this study demonstrate this impact, particularly when the Southern Site is considered, a change in grain concentration (averaged over rations offered) of 9% from 61 to 70% grain was associated with an increase in average CH₄ emissions from 95 g/head/day to 125 g/head/day. Although this demonstrates the potential for further reducing emissions feedlot rations are balanced to produce maximum growth whilst avoid metabolic conditions (sub clinical and clinical acidosis) which may limit more significant reductions in ration forage concentration difficult. Although, rations at the Northern Site comprise up to 89% grain, and Beauchemin and McGinn (2005) fed rations of 90% grain to finishing cattle. Suggesting further increases in grain concentration are feasible.

Despite rationing systems such as the MP (metabolisable protein system, NRC) enabling feedlot rations to be formulated so requirements are met but not exceeded (Klopfenstein and Erickson 2002) and protein being one of the more expensive feed components, over feeding of protein occurs to avoid limitations in growth (Cole et al. 2003). Rations are formulated with a safety margin increasing concentrations above those required (Vasconcelos et al. 2007). Overfeeding of N (in the form of crude protein) is a significant source of volatile NH₃; protein consumed above requirements is excreted, primarily in the urine, its most volatile form. The greater emissions of NH₃ from the Southern Site may potentially be attributed to overfeeding protein, which also indicates a simple approach to reducing NH₃ volatilisation. At the Southern Site, CP % in the rations averaged 14 %, compared with 13.5% at the Northern Site, despite lower live weights (477 kg compared with 598 kg), and lower estimated growth rates (1.3 compared with 1.8 kg/day). Todd et al. (2006) observed decreasing crude protein from 13% to 11.5% decreased daily NH₃ emissions by 28% (on an annual basis) however impact on animal performance was not reported. Although they suggest that “reducing crude protein in beef cattle diets may provide the most practical and cost-effective way to reduce
Chapter 7. General Discussion

NH$_3$ emissions from feedyards” (Todd et al. 2006). Similarly Vasconcelos et al. (2006) determined that reducing CP concentration during the finishing period did not affect feedlot performance but can reduce the N content of manure. However, environmental conditions and pen management can also play a considerable role in regulating NH$_3$ volatilisation and further investigation surrounding (for example) pen cleaning frequency, altering surface pH and the use of urease inhibitors is required (in association with dietary changes) to effectively manage NH$_3$ volatilisation (Cole et al. 2005; Todd et al. 2006).

A number of policy solutions for reduction of carbon emissions have been proposed in the last 3-5 years, associated with the end of Kyoto obligations. The initial approach was the Carbon Pollution Reduction Scheme (CPRS), a cap and trade approach which was intended to effect a reduction in emissions to 60% of 2000 levels by 2050. This approach was intended to be applied by mid 2010; however its introduction was deferred early in 2010, in favour of a fixed carbon price. Due to commence in July 2012, the “Carbon Tax” is a fixed price on C emissions (initially $23/tonne) applied to the top 500 polluters, or about 0.02% of Australian businesses.

The current approach of the Carbon Tax does not include emissions from agriculture, consistently; the CPRS was not proposed to cover agriculture within the first three years, with intended inclusion after the 5$^{th}$ year. However, the recently introduced Carbon Farming Initiative (CFI) will allow agricultural industries to become actively involved in Carbon Trading (Australian Government 2010a) and is a further reason why baseline emissions from feedlots need to be measured. The CFI introduces legislation to establish carbon crediting for offset/ emissions reduction projects. Effectively, businesses with high emitters can purchase carbon credits- units of greenhouse gas abatement achieved (in agricultural industries) by either reducing or avoid emissions, or by removing and storing atmospheric C. This activity needs to be additional to any management strategy which was undertaken prior to the CFI (Australian Government 2010b).

Methodologies for emission reduction, which comprise detail of the implementation of abatement or mitigation approaches need to also detail how these approaches will be monitored (Australian Government 2010a). One of the current submissions (and the only animal based proposal submitted to date, is reduction of CH$_4$ emissions through the management (culling) of feral herbivores (camels). In this approach (Australian Government 2010b), baseline emissions are estimated using an empirical equation- effectively derived from the National Inventory Methodology (2006)). Although camels appear to have little relevance to feedlot cattle, the approach for estimating baseline emissions using the set of
equations in the current methodology could be transferable to other projects proposed under the Carbon Farming Initiative. The results of this thesis demonstrate inaccuracies in the estimation of emissions using these methods, and indicate further measurements are required for emissions benchmarking before a “business as usual” baseline emission can be established.

7.7 Conclusion

This thesis examines emissions from Australian beef feedlot systems, diurnal fluctuations in emissions and the role of animal feeding behaviour in these emissions profiles. The current Australian National Inventory Methodology (AGO 2006) was evaluated with both published data and with measurements from Australian beef feedlots.

Although the equations on which the current methodology (for enteric CH₄) is based performed adequately with the results of published studies, it did not demonstrate the same accuracy when applied to Australian measurements. The results of both the validation (Chapter 4) and application exercises (Chapter 5) indicate that diet specific calculations will be essential in the prediction of enteric CH₄ emissions. Whether this is a more complex approach considering dietary CHO, or a simpler modified Tier II approach. The diurnal pattern of emissions of both CH₄ and NH₃ is not entirely controlled by behaviour, particularly in the case of NH₃, which is influenced more strongly by environmental temperature (which in turn influences feeding behaviour). Diurnal fluctuations can influence emissions measurements using micrometeorological measurements when coupled with poor environmental conditions, and caution needs to be taken in assessing the results from these measurement approaches.

The prediction of nitrogenous emissions using a set emissions factor has significant issues, due to the lack of consideration of the role of environmental conditions in moderating emissions. A more detailed approach will be required to improve these estimates- including long term measurements of not just emissions, but animal and ration characteristics and manure pad composition.

Development of technologies around measurements and increasing social demand for reduced emissions products will increase both ability to measure emissions on farm, and requirement to do so. In either case, further research will only increase accuracy of emissions measurements and enable development of a more reliable model for emissions prediction.
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Chapter 9. Appendices

9.1 Appendix for Chapter 3- Lin’s concordance

\[ C_b = [\nu + 1/\nu + \nu^2]/2 \]
\[ \nu = \sigma_1/\sigma_2 = \text{scale shift}, \]
\[ u = (\mu_1 - \mu_2)/\sqrt{\sigma_1 \sigma_2} = \text{location shift relative to the scale}. \]

Figure 9.1. Calculation of \( C_b \), a bias correction factor that measures how far the best-fit line deviates from the 45° line (measure of accuracy). Reproduced from Lin (1989).

\[
\rho_c = \frac{2 \beta_1 \sigma_2^3}{(\sigma_1^2 + \sigma_2^2) \left[ (\beta_0 - 0) + (\beta_1 - 1) \mu_2 \right]^2}.
\]

This concordance correlation coefficient, \( \rho_c \), possesses the following characteristics:

(i) \(-1 \leq -|\rho| \leq \rho_c \leq |\rho| \leq 1.\)
(ii) \( \rho_c = 0 \) if and only if \( \rho = 0. \)
(iii) \( \rho_c = \rho \) if and only if \( \sigma_1 = \sigma_2 \) and \( \mu_1 = \mu_2. \)
(iv) \( \rho_c = \pm 1 \) if and only if:
   (a) \( (\mu_1 - \mu_2)^2 + (\sigma_1^2 - \sigma_2^2) + 2 \sigma_1 \sigma_2 (1 \mp \rho) = 0, \)
   or equivalently,
   (b) \( \rho = \pm 1, \sigma_1 = \sigma_2, \) and \( \mu_1 = \mu_2, \) or equivalently,
   (c) each pair of readings is in perfect (1) agreement (for example, 1, 1; 2, 2; 3, 3; 4, 4; 5, 5) or in perfect reversed (-1) agreement (for example, 5, 1; 4, 2; 3, 3; 2, 4; 1, 5).

Figure 9.2. Calculation of the concordance correlation coefficient. Reproduced from Lin (1989).
9.2 Appendix for Chapter 3- General Model Structure

<table>
<thead>
<tr>
<th>Data collection</th>
<th>Biophysical and System Inputs</th>
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<th>Output</th>
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<tr>
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- **ENTERIC METHANE**
  1. Blaxter & Clapperton (1965)
  2. Moe & Tyrell (1979)
  3. IPCC Tier II
  4. Ellis et al. (2007)
  5. Ellis et al. (2009)

- **MANURE METHANE**
  Volatile Solids Output
  Conversion of VS to CH₄

- **NITROGEN TRANSACTIONS**
  Nitrogen partitioning to maintenance, growth and excretion
  Conversion of excreted nitrogen to N₂O & NH₃

**Figure 9.3** Diagrammatic representation of the basic modelling approach used to estimate greenhouse gas emissions from feedlot systems. The principal domains of each equation or set of equations is indicated by the dotted lines. Transfers of information between parts of the model are indicated by the arrows.
### 9.3 Appendix for Chapter 4- Measured and Predicted Emissions

Table 9.1 Published and modelled CH$_4$ emissions (g/head/day) used in the model validation

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Table 9.2 Published and predicted values (g/head/day) of nitrogen intake and excretion used in the validation of the nitrogen model

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<td>Cole et al. (2003)^</td>
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<tr>
<td>Adams et al. (2004)^</td>
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<td>T3. Bran</td>
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<td>11.5% CP</td>
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<td>13% CP</td>
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<td>Archibeque et al. (2007)</td>
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Table 9.3 Published and Predicted emissions (g/head/day) of emissions of NH$_3$ and N$_2$O used in the validation of the model for N gas

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<td>29</td>
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<td>Todd et al. (2008)</td>
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<td>118</td>
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<td>Van Haarlem et al. (2008)</td>
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### 9.4 Appendix for Chapter 5- Measured and Predicted Emissions

**Table 9.4**: Measured and predicted emissions (g/head/day) used in the application/evaluation of the model for use in Australian feedlots.

<table>
<thead>
<tr>
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<td>N₂O</td>
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<td>Ellis <em>et al</em> (2007)</td>
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<td>N₂O</td>
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<td>101.3</td>
<td>219.7</td>
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9.5 Appendix for Chapter 6- Full Diurnal Emissions Patterns

Figure 9.4 15 minute average CH₄ (g/head/day) and NH₃ (g/head/day) fluxes from the Northern Site measured during winter 2007
Figure 9.5 15 minute average CH₄ (g/head/day) and NH₃ (g/head/day) fluxes from the Northern Site measured during summer 2008
Figure 9.6 15 minute average CH$_4$ (g/head/day) and NH$_3$ (g/head/day) fluxes from the Southern Site measured during winter 2007
Figure 9.7 15 minute average CH$_4$ (g/head/day) and NH$_3$ (g/head/day) fluxes from the Southern Site measured during summer 2008
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Author/s:
Muir, Stephanie Kate

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Date:
2011

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