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Reducing feedlot nitrogen-based greenhouse gas emissions

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Abstract

One of the major issues facing the Australian feedlot industry into the future is the issue of rations supplying N contents greater than those required by the animal to maintain high levels of productivity. This practice can lead to increase emissions of ammonia and nitrous oxide from manures. Feeding elevated levels of crude protein in the ration did not alter the N efficiency of the animal but did lead to manures with higher labile N content. Direct ammonia emissions from feed pads may not differ between manures from animals offered contrasting N contents, but losses from the stockpile are greater from manures of animals offered elevated levels of crude protein in the feed. Lignite can be considered as a cost effective mitigation technology that can be applied to feedlot systems. The emissions of ammonia can be reduced substantially from the feed pad by using a rate of between 3 to 4 kg/m2 (based on economic analysis of a range of addition of lignite from 3 to 6 kg/m2 over a period of 56 day manure harvesting periods). In contrast to the manure harvested from pens managed under differing crude protein contents, lignite retained N in the stockpile and its subsequent use in cropping systems achieved good yield increases (approximately 63% physiological N response in relation to urea controls). This retention of N inevitably reduces overall ration costs as increased yields of forages grown on land receiving lignite amended manure can be recycled into the feed formulation at a lower cost than purchased-in feeds.

Executive summary

The global emissions of the two important nitrogen-based gases (ammonia and nitrous oxide) from agriculture amount to 27 to 38 Tg NH3-N.yr-1 and 4.2 to 7.1 Tg N2O-N.yr-1 respectively. Agricultural emissions of ammonia are predominately from two sources: synthetic N fertiliser and livestock manures. If the ammonia emissions from livestock systems are further subdivided, it is estimated that 31 to 55% are derived from housing and storage of manure, 23 to 38% from spreading of manures to agricultural land and a further 17 to 37% from grazing animals through direct voiding of faeces and urine to pasture and rangeland. The estimates of emissions of nitrous oxide from agricultural systems are predominately (>90%) from agricultural soils receiving synthetic N fertilisers or livestock manures with the balance being derived from burning of biomass, for instance stubbles and crop wastes, and savannah. One criticism of the current information on global emissions is the lack of segmentation into ruminants, monogastric and avians, leading to considerable uncertainties in estimates of the impacts of individual sectors on global or regional emissions.

The important drivers of emissions of nitrogen-based gases from agricultural systems are the quantity of reactive N entering the agricultural system, losses, including excretion, of reactive N from the agricultural system, the inter-conversion of one species of reactive N to another – for instance the conversion of ammonia to nitrous oxide after atmospheric redistribution of the former, and the type and extent of agricultural systems. By far the most important drivers of emissions are the inputs and efficiencies of use of those inputs by the farming system. It is general acknowledged that ruminant (beef cattle) production systems are inefficient converters of N from feed to N in edible product (estimates range from 7 to 10% of N fed is retained in product consumed) and therefore gains in efficiency are essential to reduce the overall losses as well as the potential impacts of those feed and rearing systems on the environment. However, attempts to manipulate N use efficiency in growing beef cattle have demonstrated little scope to improve N capture in products and therefore management of N inputs and waste are the key focus of improving the overall systems efficiency.

Arguably, the approach of managing beef cattle through feedlot systems, an example of a concentrated animal feeding operation, has a medium to long-term positive impacts on the environmental management of input (diet) energy and nitrogen. Feedlot systems are designed to increase growth rates of animals through the delivery of diets that contain the optimal levels of energy, protein and minerals to maximize the growth rate of the animal. In turn, this approach leads to an improved life cycle efficiency of energy and nitrogen utilization per product produced reflecting an effective reduction in maintenance requirements of the animal during production.

In Experiment 1 of this project, the impact of feeding elevated levels of crude protein in ration on ammonia and nitrous oxide emissions was examined. The direct emissions losses of N (as NH3N + N2O-N) from the feed pad were consistent across the two studies (61.2 and 62.7% for low and high protein feeding systems respectively). These emissions were 54.4 and 58.9% of feed N intake respectively. These losses are consistent with those reported by Todd et al. 2004). However, there was an increase in estimated manure stockpile losses (calculated from mass balances) from the high protein manure management system of more than 50%, thereby posing a significant N loss from the whole farm system as opposed to losses directly from the pen.

Lignite (or brown coal) is a low rank, high moisture coal that is mainly used to produce electricity. Lignite has been used as a soil amendment for many years with the aim of building soil organic matter and humic acid content. Lignite posesses several promising

characteristics that make it a suitable amendment for manure, and potentially a good method of reducing ammonia and nitrous oxide emissions. Lignite is characterised as having low pH, high humic acid content, high CEC and adsorption capacity and high labile carbon content. making it a potential manure amendment in terms of reducing manure NH3 emissions. Lignite was demonstrated as an effective method of retaining N from artificial urine – urea hydrolysis test system. Urea added in the artificial urine was completely hydrolysed (reflecting the 0 to 0.3% of retained urea in the lignite and biochar treatments). The hydrolysis of urea liberates ammonia and carbon dioxide. Ammonia losses were minimal (0.5% of N added to the test system) in the lignite amended system with 82.1% of N liberated retained as ammonium rather than being immobilised on to the surface of the lignite (14.4%). These data suggest that the underlying mechanism for ammonia mitigation using lignite is driven by chemical and adsorbent (cation exchange) processes rather than incorporation of N into the organic matter phase.

The three studies conducted on the potential mitigation of ammonia and nitrous oxide emissions from feed pads clearly demonstrated that direct emissions of ammonia could be reduced by up to 67%. The average over the three studies was ca. 40%. Nitrous oxide abatement was more variable and no clear pattern of abatement was noted. This level of abatement is promising as it demonstrates than ammonia can be sequestered in the lignite in the form of ammonium and hence a potential retention of N in manure. The N retained in manure from feed N intake in Experiment 3 were estimated as 34.5% for control in Period 1, 41.1% in lignite treated in Period 1 (or 19.1% increased retention), and in Period 2 control 21.4% with 31.5% retained in lignite (or 47.3% increase in retained N). On average, the physiological N responses (kg DM/kg N fertiliser) of sorghum to urea fertiliser and lignite amended manure were 26.8 kg DM/kg N and 16.8 kg DM/kg N. These physiological N responses were typical for forage sorghum (range from 12 to 40 kg DM/kg N).

An economic analysis of the use of lignite in feedlot systems was undertaken. The analysis is sensitive to transport costs (especially diesel and freight distance) of lignite to the feedlot and fluctuations in the price of synthetic fertilisers and other inputs to the farming system. It was estimated that when manure valuations (on the basis of nutrient retained in the farming systems), a net income per capita of $4.72 to $6.67 was possible reflecting the rate of lignite application. Further it was estimated (using average abatement of ammonia and nitrous oxide emissions from these studies) that the abatement potential of lignite is 2.04 kg CO2e/kg lignite added to the feedlot systems (without liabilities for transport or fugitive emissions). If a CFI method was to be developed for the abatement technology the net abatement return for the 13750 head feedlot system modelled would be equivalent to approximately $104,150 in credits or $7.57 per capita (exclusive of transaction and compliance costs).

The work undertaken in this project has clearly demonstrated that lignite can be considered as a cost effective mitigation technology that can be applied to feedlot systems. The emissions of ammonia can be reduced substantially from the feed pad by using a rate of between 3 to 4 kg/m2 (based on economic analysis of a range of addition of lignite from 3 to 6 kg/m2 over a period of 56 day manure harvesting periods). In contrast to the manure harvested from pens managed under differing crude protein contents, lignite retained N in the stockpile and its subsequent use in cropping systems achieved good yield increases (approximately 63% physiological N response in relation to urea controls). This retention of N inevitably reduces overall ration costs as increased yields of forages grown on land receiving lignite amended manure can be recycled into the feed formulation at a lower cost than purchased-in feeds.

The project has clearly demonstrated that there is a potential abatement strategy (lignite) that is cost effective. The research conducted has provided good evidence that the addition of lignite to the feed pad is an effective strategy for mitigating ammonia emission from cattle feedlot and recycling nitrogen in agricultural systems. The findings have major economic and environmental implications for effective nitrogen management in agriculture, especially in intensive feedlot systems.

Table of contents

1 Background

1.1 Background and description

The Australian Feedlot industry has well developed, mature environmental policies. One area of considerable concern is the lack of management options available to lot feeders to abate methane and N based gas emissions from growing cattle, feedlot pens and manure stockpiles. If attractive management options can be developed to reduce losses or sequester emissions, there are considerable net benefits to the industry. These include improved environmental outcomes (air & water quality), reduced inputs of N fertiliser to on-farm forage and grain production, increased C sequestration in soils receiving lignite-manure co-products and potential increases in animal performance and welfare. These benefits assist in building profitability within the feedlot sector. The work conducted in FLT.1048 is part of an on-going research program conducted by the University of Melbourne to understand the quantum and management of greenhouse gas emissions (methane & nitrous oxide) and other N based gases (ammonia).

1.1.1 Policy context

The work commissioned by the Australian Lot-feeders Association through Meat & Livestock Australia (grain fed levy), and in conjunction with the Commonwealth Department of Agriculture (Reducing Emissions from Livestock Research Program), has to be viewed in relation to the policies concerned with abatement of greenhouse gas emissions from concentrated animal feeding operations. In brief, much of the work undertaken during the period 2006 to 2015 has aimed to ensure the livestock industries were better positioned to address the increased business risks and opportunities due to the contribution of greenhouse gases (and in particular, methane), to climate change and government policies related to the contribution of those greenhouse gases to climate change.

The MLA Feedlot Program Livestock Production Research & Development Strategic Plan 2006-2011, through commissioning environmental research underpinned this aim, and in particular developed strategic initiatives to address the issue of greenhouse gas (GHG) emissions (methane and nitrous oxide) from lot feeding systems. During the preparation phase of the Strategic plan, it was recognised that there was a lack of data on actual emissions from cattle managed under feedlot conditions in Australia or the whole feedlot system (cattle, pens, manure management and adjacent land managed by the feedlot). A number of studies were commissioned (B.FLT.331, B.CCH.1020 & most recently B.FLT.0148 – this report) to address this situation and provide accurate information for the industry to understand any potential impacts of greenhouse gas regulatory controls on the industry.

The MLA Livestock Production Research & Development Strategic Plan 2010-2015 represented a subtle change in the focus of feedlot environment research (Imperative 3.1, 4.1 & 4.2). The focus of research was changed to manage and report accurately the greenhouse gas emissions across the supply chain thereby enabling analysis of impacts across the supply chain to be determined. The emphasis on management of greenhouse gas emissions led to B.CCH.1020 (2009-2012) focussing on manure management strategies for reducing greenhouse gas emissions from a beef cattle feedlot. Further, the work commissioned in FLT.0148 (2012-2015 – this report) evaluated options to manage greenhouse gas emissions from feedlot systems and aimed to develop a range of practical options that could be adopted by the feedlot industries to mitigate emissions

1.1.2 Science

The Australian beef feedlot industry is an example of a concentrated animal feeding operation and an example of one of the intensive livestock industries. The scale of the Australian Industry is highlighted by the most recent ALFA/MLA coordinated National Accredited Feedlot Survey (NAFS: March – June 2014). The NAFS reported that approximately 846,000 head were on feed utilising 79% of capacity. In 2013, 2.62 million head were turned-off from feedlots into a number of processing supply chains. Census data for the cattle industry suggests the national herd stands at ca. 28.5 million (ABS, 2013) with about 2 to 3% of the national herd under feed. The approach of managing cattle through feedlot systems has a medium to long-term positive impacts on the environmental management of all feed resources, and in particular, nitrogen. As the feedlot system is designed to increase growth rate of the animal, there is an improved life cycle efficiency of feed resources utilization per product produced reflecting an effective reduction in maintenance requirements during production. However, there are short-term trade-offs in in terms of elevated ‘point source’ emissions (Baek et al., 2004a; Baek and Aneja, 2004) of methane, N based gases (ammonia and nitrous oxide), volatile fatty acids (a component of odour) and particulates ((McMurry et. al, 2004; Aneja et. al, 2006b, 2008a,b,c; Galloway et. al, 2008).

Recently, there has been considerable interest in the concept of the “footprinting” of livestock and cropping systems – for example UNECE Task Force on Reactive Nitrogen: Options for ammonia mitigation (2014), US Environmental Protection Agency – Reactive nitrogen in the United States: An analysis of inputs, flows, consequences, and management options (2011), Evaluation of the livestock sector's contribution to the EU greenhouse gas emissions (GGELS: 2010) have all identified that the consumer has raised concern over the levels of nitrogen (and other resources) that are offered and apparently wasted by agricultural production systems. In particular, the UNECE report estimated an economic loss of 1 kg N lost from the system was valued at \$2.9/kg or up to \$14.5 per head in a short finishing system. This estimated cost does not consider other losses of N in the production system nor does it consider the potential costs associated with regulatory compliance. It is inevitable that agricultural and food industry policy makers will respond to the apparent wastage of N in food systems as demonstrated by recent studies of Eshel et al. (2014). Eshel et al. highlighted that consumers exercise purchasing decisions for categories of food that provide the most effective management systems that reduce environmental resource burdens (in a similar fashion to existing preference for high welfare products). The authors also suggested that globalization driven rapid diffusion of US customs, including dietary customs, into economies such as China or India identifies the emergence and global significance of resource foot printing of foods. If this is correct, any policy decisions based on the reduction of the environmental costs of food production may result in either scenarios where animal based systems are reduced (posing a problem for global protein supply) or have to comply to more strict standards that proactively sequester or mitigate potential sources of resource loss or wastage.

The Australian feedlot sector has mature, well developed environmental management codes of practice that allow all producers to be accredited through the National Feedlot

Accreditation Scheme. The codes of practice and the accreditation scheme ensure that producers adhere to current environmental legislation, environmental objectives and performance indicators, and that operational practice and monitoring programs are documented (National Beef Cattle Feedlot Environmental Code of Practice, 2012 & National Guidelines for Beef Cattle Feedlots in Australia, 2012). These codes ensure that the industry achieves gains in production under a 'cleaner production' framework that leads to improvements in resource use efficiency and reduces wastage and costs associated with the production cycle. Furthermore, the principles of product integrity need to be aligned to those of the environmental management principles and in particular the AUS-MEAT Minimum Standards for Grain Fed Beef and other feeding Standards are met. This is important in the export of product to markets such as the European Union (access to the EU High Quality Beef Quota: EU 481/2012) where prescriptions for grain-fed, level of feeding and metabolisable energy content (>12.26 MJ/kg DM) are identified. For efficient ration formulation, the guideline of rations containing >12.26 MJ/kg DM leads to the use of feedstuffs that contain moderate levels of fat and are derived from other agricultural sectors (for instance cottonseed meals, pressed canola products etc.). One of the consequences of this approach is that many of these feed resources contain elevated levels of crude protein (CP) thereby increasing the total CP content of ration to levels in excess of animal requirements. Recent analysis of 43 rations feed to beef cattle managed under lot feeding systems in Australia identified a 'baseline' CP content of a ration to be 13.92% or 22.3 g N/kg DM (range min-max = 11.5 to 15.09%: Hill unpublished data, RELRP B.CCH.1086). The 95% CI for CP content for this sample of rations was 13.49 - 14.35%. Theoretically the minimum CP content of a ration to support the lower 95% CI of growth is 10.2% (NRC, 2000), To maintain an acceptable growth rate (1.35 kg/d) the minimum CP content for a finishing ration that a producer is likely to offer is 11.6% or 18.6 g N/kg DM. In practice this situation may lead to a surplus of 35 to 50 g N/head/day being consumed and excreted by the growing animal, or approximately 3.5 to 5 kg N/head over a short feeding cycle.

The research conducted between 2006-2009 (FLOT.331) using open path laser and FTIR techniques represented the first Australian study to measure greenhouse gas emissions from beef cattle feedlots using open-path spectroscopy and atmospheric dispersion modelling. These methods are less intrusive alternative to SF_6 enteric tracer ratio techniques or small surface chambers for emissions of N_2O or CH_4 from soils (Laubach et al. 2013; Redding et al. 2013). They are based on the measurement of concentration of the greenhouse gas as a line average concentration windward of the feedlot pen. The emissions from the pen (as a point source) are calculated using an inverse-dispersion micrometeorological technique. This model calculates the theoretical relationship between a source emission rate, e.g., the animal pen, and downwind concentration, so that a single concentration measurement with known background concentration can establish the total emission rate (McGinn et al. 2009; Flesch et al. 2007; Loh et al. 2008).

The findings of that work suggested that the IPCC methodologies for estimation of emissions from feedlots overestimated nitrous oxide emissions by up to 50% and underestimated ammonia emissions by a factor of 3 times. Methane emissions were with 15% of the IPCC estimates but there were considerable seasonal variation noted that could not be ascribed to livestock numbers. The data also suggested that at the point of measurement, emissions of ammonia N from a typical 15,000 to 20,000 head feedlot could be as high as 4 to 5 tons of urea fertilizer equivalent daily or more than 70% of N fed was lost to emissions. The work in

FLT.331 also identified that the emissions footprint of these systems is approximately 55-60% methane and the balance as nitrogen based greenhouse gases. Project B.CCH.1020 built on the earlier work that identified the magnitude of emissions, by examining whether urease inhibitors could abate ammonia emissions and retain ammonium as urea in the manure. By application of urease inhibitor to the surface of the feedlot pen, 30 to 40% of urea was retained in the manure after the first 3 to 4 days following transport and stockpiling of the manure. However, despite increased retention of urea in urease-inhibitor treated manure, ammonia emissions from manure stockpiles were not consistently reduced. Deployment of this technology led to increased retention of urea in urease-inhibitor treated manure but ammonia emissions from manure stockpiles were not consistently reduced.

The current project reported here builds on this initial work and assist the development of strategies to abate methane and N-based emissions, thereby potentially allowing the industry to develop abatement methodologies that are approved under the Carbon Farming Initiative or conduct further analysis of their individual systems to reduce the losses of feed resources especially nitrogen.

2 Projective objectives

The objectives of the project were:

1. Determine the chemical and biochemical processes that drive the conversion of urea and other nitrogen sources to ammonia in manure derived from beef feedlots.
2. Evaluate a range of co-amendments to manure that could mitigate emissions of ammonia or nitrous oxide thereby reducing total direct and indirect greenhouse gas emissions.
3. Evaluate the impact of recycling manures with elevated concentrations of retained N under cropping systems.
4. Determine the impact of dietary manipulation to mitigate methane emissions from cattle on subsequent manure composition and N emissions.
5. Based on the above, develop abatement methodologies that can be included on the Department of The Environment 'positive' list for abatement of N-based feedlot greenhouse gas emissions.

These objectives were achieved through desktop, laboratory and field based research studies conducted from 2012 to 2015.

3 Methodology

3.1 Experiments

Two major experiments were conducted in B.FLT.0148. These were focussed on (i) the effect of reducing the crude protein content of the ration consumed by beef cattle on direct (methane and nitrous oxide) and indirect (ammonia) greenhouse gases, and (ii) the use of lignite as an cost effective, efficient abatement method to reduce N based gases from feedlot pens and manures. These studies were underpinned by small scale laboratory experiments and a literature review.

3.1.1 Review of literature

A review of the relevent literature concerned with alternative approaches to reduce N loss from feedlot systems was undertaken. The review considered (i) Characterization of N losses from feedlots and manure management systems; (ii) Strategies to reduce nitrogen losses from feedlot manure; (iii) Chemical inhibition of microbial processes (iv) mineral and chemical amendments to reduce N losses from manure and (v) organic amendments to manure management systems.

3.1.2 Nitrogen transformations in an artificial urine/manure system treated with a range of amendments

The experimental design was similar to those conducted in B.CCH.1020. The apparatus consisted of open-topped, plastic bins of dimensions 17cm (L) x 15cm (W) x 15cm (D) with a force draft fan to carry all gaseous emissions through an acid trap. A rockwool media (500 g per box) was used to act as an artificial manure (inert) matrix. Three treatments were installed:

i) Control – inert matrix plus artificial urine
ii) Lignite at a rate of 10% plus artificial urine
iii) Biochar at a rate of 10% plus artificial urine

A blank (inert media only) without artificial urine was conducted. To simulate cattle pen conditions, 41cm^3 of synthetic urine was added to each box every two days (based on 6 L of daily excretion over an area of 12m^2). These rates of addition were similar to those used in B.CCH.1020. Water was added to all the boxes to keep the moisture content near constant over the period of experiment. Synthetic urine was prepared fresh before each application. The synthetic urine preparation was prepared as described in Parker et al. (2005). Flow rates (0.3, 0.6, 1.0 litres/minute) were evaluated to optimise ammonia capture from the test matrix. Ammonia was released from a solution of $(NH_4)_2SO_4$ (18.9 mmol/litre) treated with NaOH and MgO (1 mmol/litre) and the test solution pH was maintained above 8.0. Ammonia gas evolved was trapped with 125 ml of 0.5 M sulphuric acid and the acid solutions were analysed for NH_4^+-N, in the Skalar auto analyser. Ammonia released from a 14-day incubation was captured in a similar fashion to that noted previously.

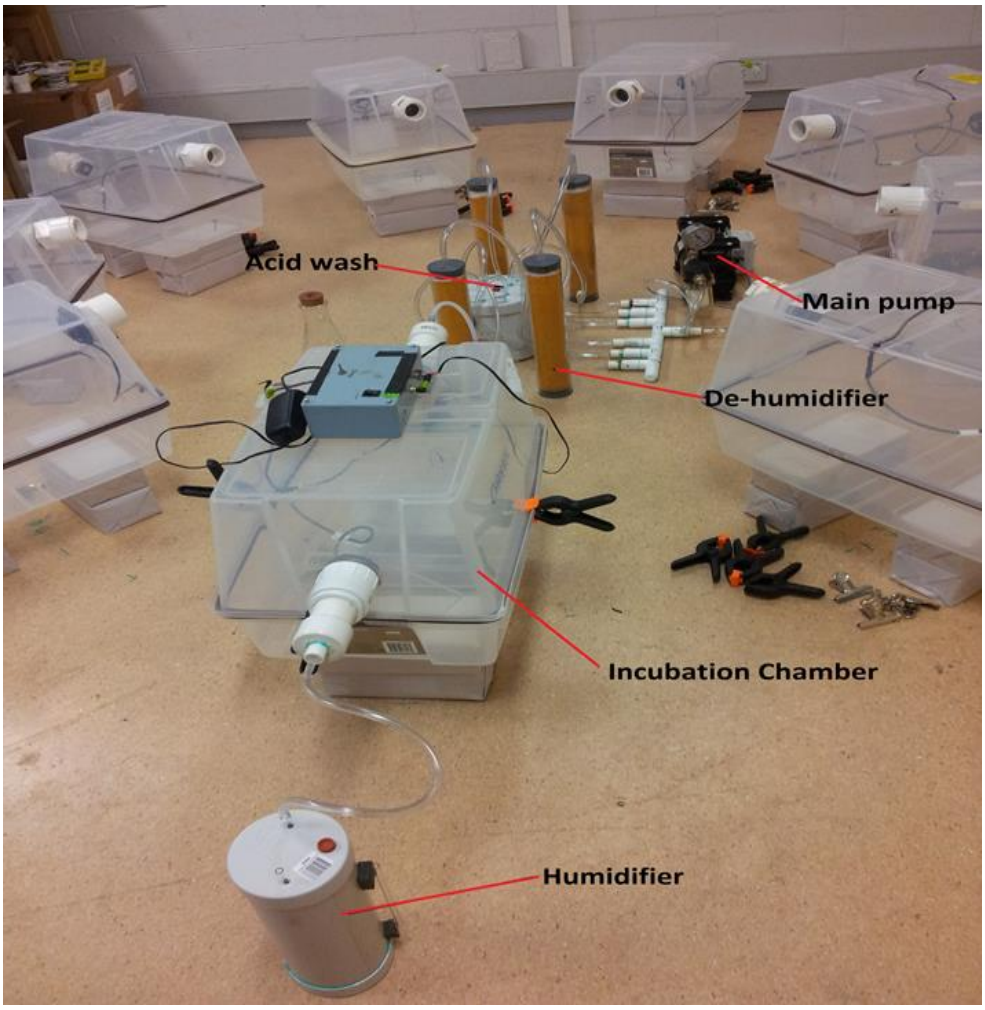


Figure 1 Construction of new incubation chambers allowing 9 treatments to be run in parallel at the same time. Gas sampling from all chambers can be achieved through sequential switching of a gas solenoid after the acid wash.

3.1.3 Effect of altering the crude protein content of the ration on direct and indirect greenhouse gas emissions

Ammonia (NH_3) emissions from intensive ruminant production systems are major environmental and economic issues reflecting their contribution to odour and total greenhouse gas emissions as well as the loss of nutrients. Recent work by Denmead et al. (2013) has demonstrated that ammonia is a potent indirect greenhouse gas with measured emissions of greater than 5000 kg/day from 20000 head beef feedlots in Queensland and Victoria. The impact of these emissions, through re-distribution of ammonia in the environment and subsequent conversion to nitrous oxide (direct greenhouse gas), is approximately 19t CO_{2e}/day. The recent introduction of the Australian Government Carbon Farming Initiative has led the industries to focus on novel abatement strategies to reduce total direct and indirect greenhouse gas emissions. These strategies include the use of adsorbents in the construction of the feedlot (e.g. lignite, diatomaceous earth, kaolin), manipulation of the ration to reduce nitrogen excretion through the urine and faeces (e.g. reducing crude protein content of the ration, substitution urea with nitrate, feeding plant extracts – tannins). The work proposed in this project will evaluate the impact of reducing the crude protein content of the ration on total nitrogen excretion in co-mingled urine and faeces (waste) and emissions of nitrogen-based greenhouse gas emissions (nitrous oxide and ammonia).

The two outputs proposed for the study were:

1. Actual measurement of N based greenhouse gas emissions (including the indirect GHG ammonia) from manure voided from beef cattle offered two levels of crude protein.
2. Biophysical modelling of feed N to N voided to quantify the theoretical N based GHG emissions and identify if there are any deviations from current model approaches and actual measurements.

3.1.3.1 Animals, rations and feed management (Experiment 1)

3.1.3.1.1 Animals

Sixty Angus steers (age 9 – 12 months; initial live weight 289 *s* = 19.6 kg; range 251 to 328 kg LW) were purchased from Seymour, Victoria and transported to Dookie Agricultural College (University of Melbourne). All animals were vaccinated for the five common clostridial diseases (tetanus, malignant oedema, enterotoxaemia, black disease and blackleg) with a '5-in-1' vaccine, and respiratory disease (infectious bovine rhinotracheitis). The animals were backgrounded on pasture and grain supplementation for two weeks in mid-June 2013. The herd was then divided into two equal groups of thirty animals (according to live weight) on 18 June 2013 and introduced into two grain rations formulated to contain nominally high (HP) or low protein (LP) contents. The original modelling of rations was reported in Milestone 3. The initial grain content of each diet was 5% and the level of grain was increased incrementally by 10% per week to 60%.

3.1.3.1.2 Rations and feeding

The two rations were formulated according to best industry practice (NRC, 2007; LRNS 2012) and designed to fulfil energy, protein and mineral requirements of growing beef cattle.

- High Protein Ration (HP) – high crude protein content comprised crushed barley grain, sunflower, lupin and cottonseed as the protein and energy source with ryegrass hay and mixed clover hay chaff as the fibre supplements. The estimated CP content was 17.5% and the ME content was 10.8 MJ/kg DM (Table 1).
- Low Protein Ration (LP) – low crude protein diet comprised crushed barley grain, lupin and sunflower as the protein and energy sources with ryegrass hay and mixed clover hay as the fibre supplements (NRC, 2007). The estimated CP level was 14.5% and the ME was 10.8 MJ/kg DM (Table 1).

Table 1 Ration composition (mixed ration)

Feed Analysis	LP	HP
Grain content (% of total diet)	60.0	60.0
Dry Matter (%)	83.6	83.9
Moisture (%)	16.4	16.1
Crude Protein (% of dry matter)	14.5	17.5
Acid Detergent Fibre (% of dry matter)	26.0	27.1
Neutral Detergent Fibre (% of dry matter)	43.2	45.4
Digestibility (DMD) (% of dry matter)	69.8	68.7
Metabolisable Energy (MJ/kg DM)	10.8	10.8
Fat (% of dry matter)	1.4	1.5
Ash (% of dry matter)	4.8	6.7

Original feeding plan of the trial was to maintain the grain content of the mixed rations closer to 70% of total intake, a level lower than typical feedlot rations used under commercial conditions where the grain contribution is greater than 75%. Initial modelling using LRNS (2012) suggested that rumen pH would not fall below 6.3 for the mixed ration indicating both rations would pose little risk of acidosis. Acidosis occurred on two occasions (leading to animal death) reflecting poor mixing or the ration, poor intake characteristics of the forage component and animals selective feeding. An acidosis risk management strategy was implemented leading to: increased moisture content of the ration, addition of sodium bicarbonate (1.6%) and limiting the grain portion of the diet to 60% of total intake. All feeds were delivered using a mixer wagon feeding system.

The feeding trial was 132 days duration with 41 days measurement of N based gases from pens and manure management system. Feeds were made up twice weekly in batches. A known allocation according to a predetermined feeding schedule was allocated twice daily to individual pens. Refusals (if any) were measured twice weekly. Feed intakes of individual animals were estimated from group intakes. The animals were weighed 6 times during the 132 day feeding period and the estimate of live weight gain was determined by regression. The measurement period was split into two discrete periods (17/09/2013 to 7/10/2013 and 8/10/2013 to 28/10/2013) reflecting the changes in feed allocation through the trial.

3.1.3.2 Feedlot pad, manure handling and sampling

The cattle were be held in two pens (dimensions 20 x 20 m = 400 m^2 or stocking density of 14.3 m^2/animal) and offered one of two feeds at an ad libitum (15% above estimated DM intake) level of intake using commercial feed troughs (27 m access) with standard head space Water (ad libitum) was provided through water troughs with 10 m access (Figure 2).

Figure 2 Feed pad design at Dookie Agricultural College, Victoria

3.1.3.2.1 Manure harvesting and sampling

Co-mingled faeces and urine were collected from the feedlot pen surface using a mini-bobcat) and stockpiled about 1.5km away from site. Process followed the conventional commercial beef feedlot practice (SCARM 47; CSIRO, 2012). Each pen was divided into four sections (quadrants) for manure sampling. Fresh faeces, co-mingled faeces and urine were collected from 5 locations within each quadrant and combined into one bulk sample (four bulked samples/pen). Urine samples were taken from fresh urine pools immediately after voiding using a clean syringe. Samples were transferred to a freezer immediately and stored under -20^{o}C until analysis. Analysis will include, Total N, total C, Urea N, Nitrate-N and NH_4-N contents.

3.1.3.3 Measurement of gas emissions

3.1.3.3.1 Trace gas analysis system

The general plan for the measurement of gas emissions from the feed lot systems is shown in Figure 3.

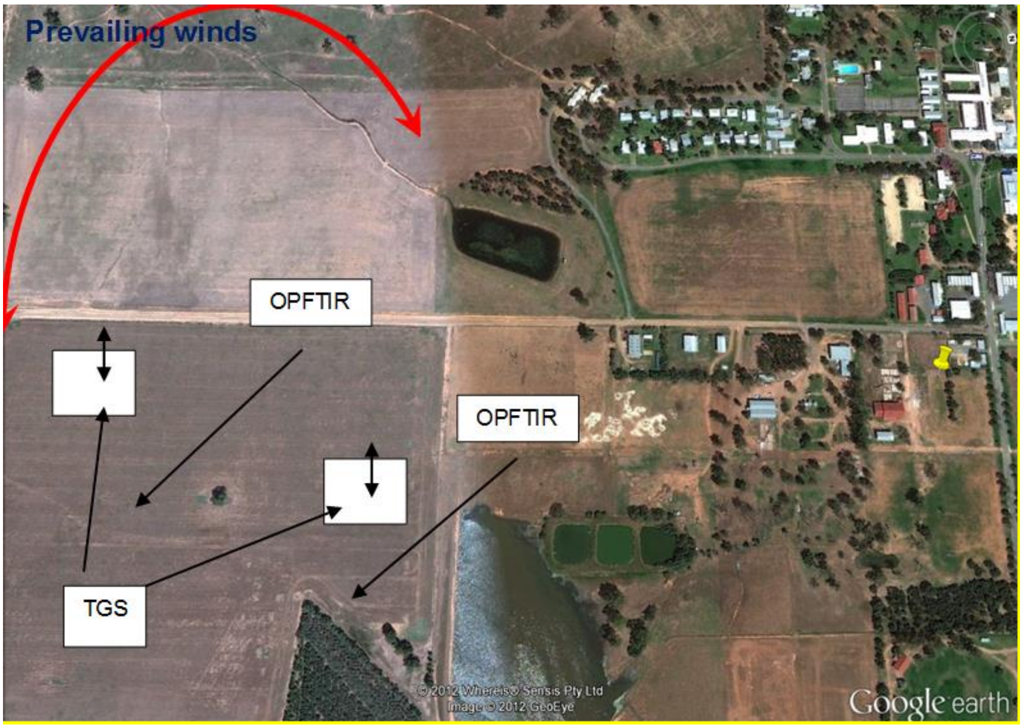


Figure 3 Generalised plan of emissions measurements from feed lots

ECOtech trace gas systems (TGS) with NH_3 and CH_4 chemiluminescence analysers were deployed at each site (15m from each pen). The sampling and analysis system was an air sampling tower, air sample delivery tubing, suction pump, automated switching solenoids/manifold (with data logging system) and the TGS (Figure 4). TGS automatically sampled and analysed air samples drawn from five height levels (0.25m, 0.5m, 1m, 2m and 3m) in the centre of each pen. Five separate air lines (sample delivery tubing) for each height delivered air samples from the centre of the pen to the TGS. Teflon tubing was used reflecting its non-absorbance of NH_3. A custom-made hot water circulation system was used to maintain the temperature above 30^{o}C thereby preventing NH_3 or condensation build-up in the sample tubing. Five Teflon air lines were tied around the hot water supply pipe and the whole structure was then wrapped with industrial grade outdoor thermal insulation.

The switching solenoid supplied air samples to the TGS from each sampling height in a known sequence and measurement of trace gas concentrations were made at every 6 minute intervals. This data was used to calculate the vertical profile of individual trace gas concentrations to 3m. A 2D wind sensor was mounted on the sampling tower at each sampling height and wind speed and direction was recorded at one minute intervals. Trace gas fluxes were calculated as 30 min averages.

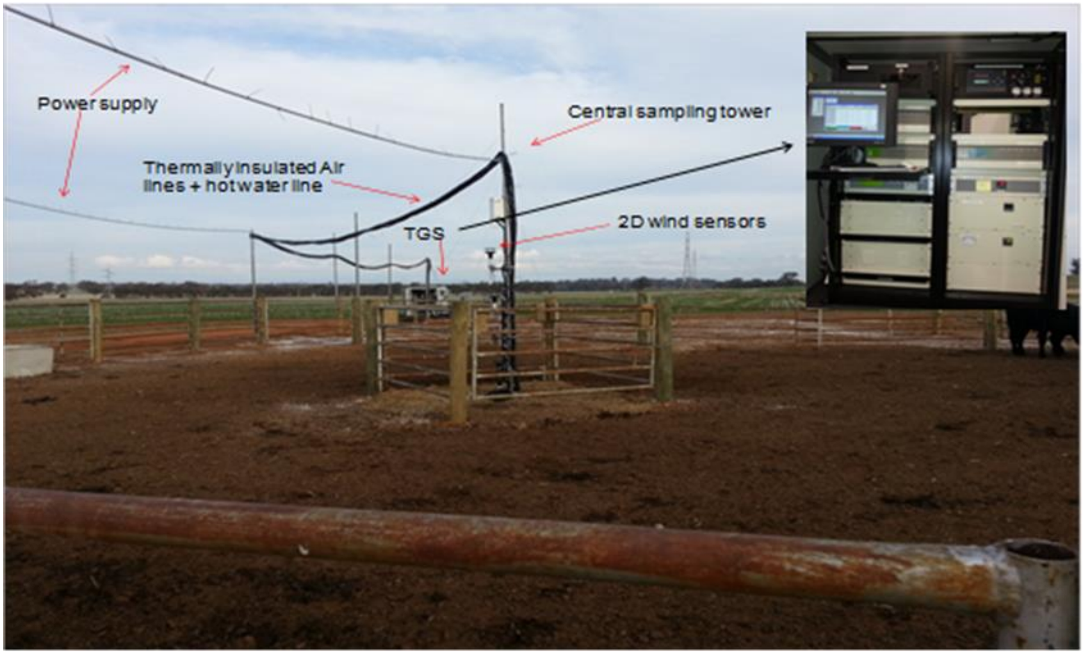


Figure 4 Central gas sampling tower and air lines that delivered air samples to the trace gas analysis system

3.1.3.3.2 FTIR and open path laser measurements

An OP-FTIR spectrometer was installed on the southern side of paddock aligned with each experimental feedlot. The OP-FTIR path-length was 104.56 m and 100.93 m for high protein (HP) treatment feed paddock and low protein (LP) control feed paddock, respectively. OP-FTIR was set up at the height of 1.18 m and 1.37 m above the ground at the HP and LP paddock, respectively. Line-averaged concentrations of CH_4, CO_2, N_2O and NH_3 were measured at 3-min intervals simultaneously. A weather station was set up in the eastern side of experimental sites to avoid the interference from animals and instruments. A 3-D sonic anemometer (CSAT3, Campbell scientific, USA) was mounted on a mast at 2 m above the ground. Three temperature sensors (TCDirect, Australia) were also mounted at 1.0, 2.0 and 3.0 m above the ground. Data was averaged to 15-min periods. Wind speed, wind direction, ambient temperature and the variance and covariance of winds were recorded at 10Hz by a data logger (CR23X, Campbell Scientific, Australia). The coordinates of all sensors and OP-FTIR spectrometers were measured using the global positioning system (GPS) (Garmin, GPSMAP 62S, USA) (Figure 5).

Backward Lagrangian stochastic dispersion model (WindTrax 2.0, Thurder Beach Scientific, Canada) was used to simulate gas fluxes emitted from the source (feedlot) by computing concentrations and winds turbulent statistics into flux. Wind conditions are key factors modelling the data through WindTrax. The criteria for obtaining the valid data are: $u^* > 0.15$ m/s, surface roughness is < 0.9 m and atmospheric stability length is $|L| > 5$ m. Furthermore, the touchdown of simulated particles at source area is > 10% of the total area.

A control gas release (recovery) trial was conducted 10 days after the end of the feeding trial (4 November 2013) once animals and manure were removed from the sites. Three gases of CH_4 (> 89%), N_2O (> 99%) and NH_3 (> 99%) (Coregas, Australia), were released in the centre of the southern paddock, and the flow rate was controlled by a mass flow controller (Alicat MCS10SLPM, MC20SLPM, USA). The flow rate of CH_4, N_2O and NH_3 was 8, 8 and 5 standard L/min for an hour for each gas.

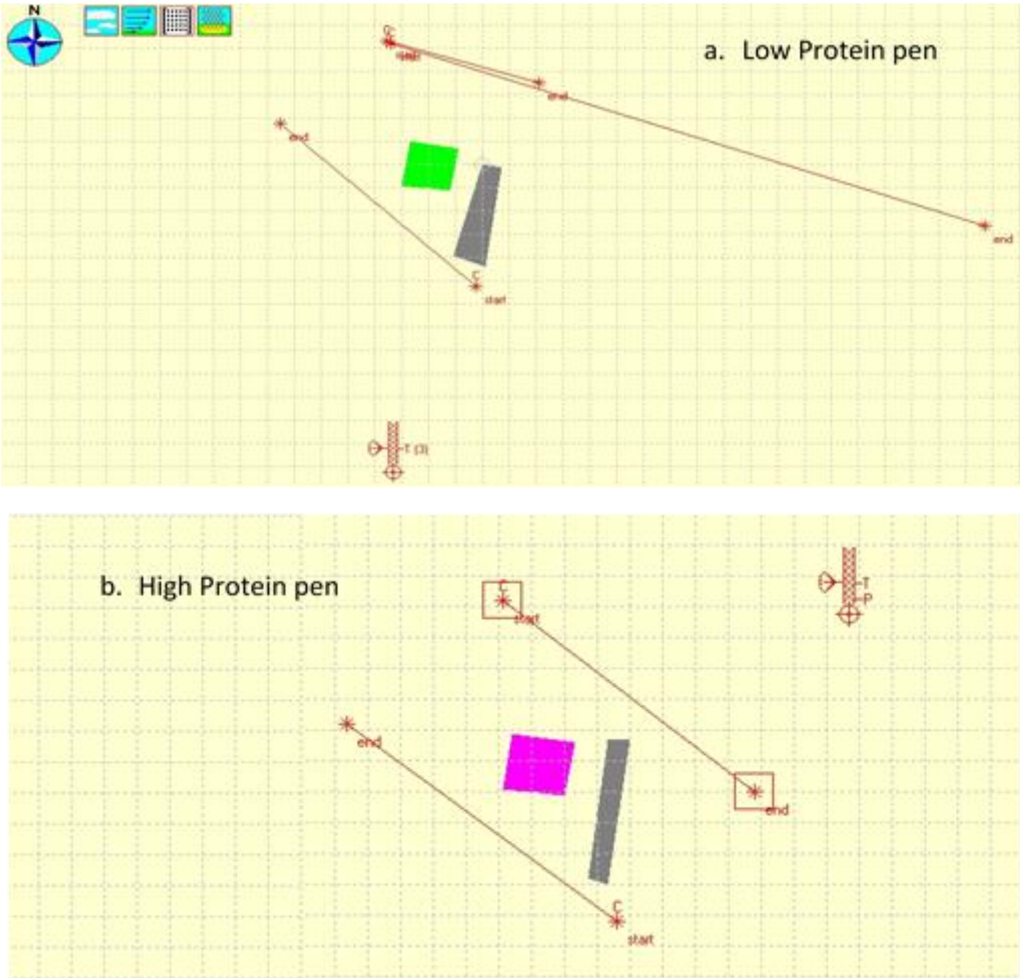


Figure 5 WindTrax map with locations of all sensors and OP-FTIR at high and low protein pens

3.1.3.3.3 Manual N2O sampling

N_2O was sampled twice a week (every Tuesday and Friday) using 50 x 50 x15 cm pre-fabricated thermally insulated sampling chambers. Four chambers were placed across the pen covering the variability in manure/urine distribution on the pen surface and the sampling followed a protocol developed based on the DAFF methodology guidelines (http://www.daff.gov.au/__data/assets/pdf_file/0003/2276490/aotg-guidance-for-on-farm-measurement-final.pdf).

3.1.3.4 Crop trial

A field experiment was conducted at the University of Melbourne, Dookie Campus (36.37°S, 1450.7°E), 220 Km North of Melbourne, Victoria, Australia from 6th December 2013 to April 21st of 2014. The topography of the experimental site was a flat underlain with a loamy clay (Nalinga loam) and well-drained soil. The prevailing winds at the site during the experiment period were NE and SE. The average daily wind speed was 2.35ms-1 and the mean monthly maximum and minimum temperature varied in the range of 21.9-33.5 9.9-15.5and ºC. The average annual rainfall of the area was 550 mm. The field site was previously under annual rye grass and grazed by sheep.

3.1.3.4.1 Treatments and management

Manure was collected from the pen surfaces of both treatments at the end of the measurement period (23/12/2013). A replicated plot design (randomised block design with 11 treatments, 4 blocks and three replicates per treatment) was installed with the aim of detemining soil N transactions from a range of treatments measured under growing conditions. A forage sorghum crop was established in late December 2013 and harvested after 6 and 12 weeks of growth. The treatments were:

1. lignite treated cattle manure:98t/ha,
2. high protein ration fed cattle freshly collected and stockpiled (98t/ha)
3. high protein ration fed cattle aged manure (126t/ha)
4. low protein ration fed cattle aged manure (126t/ha)
5. urea fertiliser (228 kg/ha – 105 kg N/ha)
6. half rate of standard urea only (114 kg/ha- 52.5 kg N/ha)
7. lignite: urea treatment 34t/ha lignite + 114kg/ha urea (52.5 kg N/ha)
8. lignite: urea treatment: 34 t/ha + 228t/ha urea (105 kg N/ha)
9. lignite: urea treatment : 98 t/ha + 228kg/ha urea (105 kg N/ha)
10. lignite only (98t/ha)
11. control – no amendments

3.1.3.4.2 Soil and manure analysis

Soil was sampled before cultivation and the end of the crop growth with a hydraulic auger for analysis. Soil samples were tested for pH (water 1:5 w/v), EC (water 1: 10 w/V), ammonia nitrogen, nitrate nitrogen, total nitrogen and total phosphorus. Available Calcium, Magnesium, Potassium, Ammonium, Exchangeable Sodium, Potassium, Calcium, Magnesium, Aluminium, and available micronutrients such as Zinc, Manganese, Iron, and Copper were analysed using inductively coupled plasma mass spectrometry. Cation exchange capacity, total carbon (TC) and soil organic matter were also determined. Soil moisture and temperature were monitored by using multi-sensor profile probes. All of the manure samples including lignite-treated manures, new manures, fresh manures and old manures were measured for moisture and nutrient contents.

3.1.3.4.3 N_2O Flux measurements

N_2O flux was measured using the closed chamber technique. A cylindrical stainless steel chamber (30 cm in diameter, 20 cm in height) was placed on the soil surface and then inserted to a depth of 5 cm in the soil. A thermometer inserted into the chamber was used to record the temperature when gas samples were taken. Rubber seals ensured the gas-tightness of the joints of the chamber and frame when the chambers were closed. Flux measurements were initially conducted after fertilization. The subsequent flux measurement was conducted at least once a week. The chambers could cover the crop seedlings. When the crop height was above 20 cm, chambers and frames were moved to the space between rows and measured the emissions from the soil. Samples of air from the chamber headspace were withdrawn by using a polypropylene syringe and stored in a gas-tight evacuated glass vial (10 mL). Air samples were collected from each chamber at 0, 15, 30 and 45 min after the chambers were set up. Flux measurements were performed between the hours of 10am and 11am. 11 chambers were used per block for each treatment. In this experiment it was not possible to take measurements by directly connecting the equipment to the chambers due to the flooding conditions that were created in the field after irrigation. Under the existing circumstances, an error in measuring the flux would have been possible if the rate of air sample withdrawal had not been compensated by pressure venting (Livingston and Hutchinson, 1995). However, the potential error associated with this is assumed to be small, firstly because of the relatively small sample volume compared to the total volume of the inside of the chamber. Secondly, because the time required obtaining the sample was about 30 s and the tube vent partially compensated for the difference in pressure during sampling. As the same static approach was used for all chambers, a comparison between the different treatments is valid. The N_2O from the vials was quantified by gas chromatograph.

N_2O flux rates were calculated using the slope of the temporal change of the concentration within the closed box, based on the following equations (Ruser, 1998):

$$FN_2O=KN_2O(273/T)(V/A)(dc/dt)$$

where FN_2O is the flux rate of N_2O (μg N_2O-N/ m^2 /h), kN_2O (1.25 ug N/μL), is unit conversion factor for calculating the N_2O flux rates, T is the air temperature within the chamber (K), V is the volume of the chamber (L), A is the covered soil area (m^2), and *dc/dt* is the rate of change in concentration of N_2O in the chamber.

3.1.3.4.4 Sorghum plant height measurement, biomass sampling, and analyses of N recovery

The plant height was measured at 7, 15, 22, 29, 60 and 90 days after sowing (DAS). Dry biomass was measured at 45 and 100 DAS by oven-drying at 60℃ for 7 days. The head grain yield was measured at the 100 DAS. Heads were also sampled from three random lines in each plot (1m for each) at 100 DAS. The grain subsamples selected were ground and then used for plant nutritional analysis. The analysed plant nutritional parameters were calcium, chloride, copper, iron, magnesium, manganese, nitrate, phosphorus, potassium, sodium, total nitrogen, and zinc.

3.1.3.4.5 Data analysis

Data were subjected to analysis of variance (ANOVA) using Minitab 16 statistical software package (http://www.minitab.com/en-us/) to determine the significance of treatment effects. Least significant differences (LSD) were used to test for differences among the treatment means. Differences were assigned based on a significance of $P \leq 0.05$.

3.1.4 Effect of using lignite amendments to the feed pad surface to reduce direct and indirect greenhouse gas emissions

Nitrogen use efficiency in intensive beef (feedlot) production systems is inherently low, resulting significant NH_3 & N_2O emissions from manure. A recent study (Kagimbo et al. 2013) was conducted at the University of Melbourne to evaluate the effectiveness of lignite coal and biochar in reducing nitrogen losses from feedlot cattle manure. Forty (40) kg of manure was mixed with either 6kg of biochar or lignite and the other 40kg of manure were used as control. These were put in 220 drum replicated 3 times where 50g of N in form of urea were sprayed in them. The concentration of urea, ammonium (NH_4) and nitrate (NO_3) emissions of N_2O and NH3 were monitored for 21 days in each drum to assess nitrogen loss. At the end of the experiment (3 weeks), biochar and lignite reduced ammonia loss by 38% and 72% respectively compared to the control. However biochar increased N_2O flux 10 fold as compared to that of the control. The broader application of these technologies is not evident in the intensive animal industries reflecting the current practice of stockpiling of manure in the pen and outside prior to field application of manure with little or no pen amendment/mitigation strategy. Furthermore, manure management operations are highly diverse depending on the industry type, scale of operations and biochemical & physical conditions and each stage of manure management cycle require specific considerations in terms of developing GHG mitigation and nutrient recovery methods.

During the period of September to November 2013, the effects of lignite application rates on minimising N gaseous loses from feedlot were tested at Dookie Campus, The University of Melbourne. Two small feedlots (20m x 20m) were used for the experiment by surface dressing one with a target lignite rate. Experiment 2 applied 4 kg/m2 lignite to the feedpad surface and was conducted from November 2013. Experiment 3 had two periods – the first period from 4 September 2014 to 2 October 2014 (3 kg/m^2 applied lignite) and a second period conducted from 7th October to 17th November 2014 in which 6kg/m2 lignite was applied to the feedlot pen surface. Both lignite application rates were based on dry mass. The pH value of lignite applied at each trial was 3.6 and 4.5 respectively.

The objective of Experiments 2 & 3 was to evaluate the impact of using lignite as an amendment to the manure pad on N based gas emissions (nitrous oxide and ammonia). The two outputs proposed were:

1. Actual measurement of N based greenhouse gas emissions (including the indirect GHG ammonia) from manure voided from beef cattle offered two levels of crude protein.
2. Biophysical modelling of feed N to N voided to quantify the theoretical N based GHG emissions and identify if there are any deviations from current model approaches and actual measurements.

3.1.4.1 Animals, rations and feed management (Experiments 2 & 3)

3.1.4.1.1 Animals

Experiment 2
Fifty-eight Angus steers (age 12 months; initial live weight 508 s = 26.5 kg; range 404 to 615 kg LW) were allocated to the experiment. All animals had been used in Experiment 1 of B.FLT.0148 (See Section 3.1.3.1.2) and were vaccinated for the five common clostridial diseases (tetanus, malignant oedema, enterotoxaemia, black disease and blackleg) with a '5-in-1' vaccine, and respiratory disease (infectious bovine rhinotracheitis). The animals were already acclimatized to the feedlot rations (See Milestone 5). The herd was then divided into two equal groups of 28 animals (according to live weight) on 2 November 2013 and introduced to the high protein grain ration.

Experiment 3
Fifty-four Angus steers (9 months: initial live weight 260 *s*=24 kg; range 214 - 300 kg LW) were allocated to the experiment. All animals were vaccinated for the five common clostridial diseases (tetanus, malignant oedema, enterotoxaemia, black disease and blackleg) with a '5-in-1' vaccine, and respiratory disease (infectious bovine rhinotracheitis). The animals were backgrounded on pasture and grain supplementation for 35 days until were divided into two groups (25 each) and introduced into pens on 4th September 2014.

3.1.4.1.1 Rations and feeding

Experiment 2
The ration was formulated according to best industry practice (NRC, 2007; LRNS, 2012) and designed to fulfil energy, protein and mineral requirements of growing beef cattle.

- High Protein Ration (HP) – high crude protein content comprised crushed barley grain, sunflower, lupin and cottonseed as the protein and energy source with ryegrass hay and mixed clover hay chaff as the fibre supplements. The estimated CP content was 17.5% and the ME content was 10.8 MJ/kg DM (see Section 3.1.3.1.2).

Feeds were made up twice weekly in batches. A known allocation according to a predetermined feeding schedule was allocated twice daily to individual pens. Refusals (if any) were measured twice weekly. Feed intakes of individual animals were estimated from group intakes. The animals were weighed 3 times during the feeding period and the estimate of live weight gain was determined by regression.

Experiment 3
All animals were fed a single ration formulated on normal Australian industry feeding practices for growing beef cattle (13.5% crude protein; 12 MJ/kg DM).

- Feeder Ration – moderate crude protein content comprised crushed barley grain, sunflower, lupin and cottonseed as the protein and energy source with ryegrass hay and mixed clover hay chaff as the fibre supplements. The estimated CP content was 13.5% and the ME content was 12.0 MJ/kg DM.

Feeds were made up twice weekly in batches. A known allocation according to a predetermined feeding schedule was allocated twice daily to individual pens. Refusals (if any) were measured twice weekly. Feed intakes of individual animals were estimated from group intakes. The animals were weighed 3 times during the feeding period and the estimate of live weight gain was determined by regression.

3.1.4.2 Treatments, feedlot pad, manure handling and sampling

Experiment 2
Two pen treatments were installed - control (no lignite) and a lignite treatment (40t/ha or 4 kg/m^2) to the pen surface. The trial lasted from 4 November to 23 December 2013 (49 days).

Experiment 3
The first period of a 3kg/m^2 lignite application rate was conducted from 4th September to 2nd October 2014; a second period evaluating 6kg/m^2 lignite was conducted from 7th October to 17th November 2014.

The feedlot pens were nominally 20 x 20 m = 400 m^2 and held animals at a stocking density of 14.8 m^2/animal) and offered one of two feeds *ad libitum* (15% above estimated DM intake) level of intake using commercial feed troughs with standard head space. Water (ad libitum) was provided through water troughs with 10 m access (see Section 3.1.3.2).

3.1.4.2.1 Manure harvesting and sampling

Experiment 2
Co-mingled faeces and urine were collected from the feedlot pen surface using a mini-bobcat) and stockpiled about 1.5km away from site. Process followed the conventional commercial beef feedlot practice (SCARM 47; CSIRO, 2012). Each pen was divided into four sections (quadrants) for manure sampling. Fresh faeces, co-mingled faeces and urine were collected from 5 locations within each quadrant and combined into one bulk sample (four bulked samples/pen). Urine samples were taken from fresh urine pools immediately after voiding using a clean syringe. Samples were transferred to a freezer immediately and stored under -20^{o}C until analysis. Analysis will include, Total N, total C, Urea N, Nitrate-N and NH_4-N contents.

Experiment 3
Co-mingled faeces and urine were collected from the feedlot pen surface in four transects, with each transect as one sample. 10 samples were collected on each transect by using a soil sampling corer (3cm diameter). Each sample was dried at 40ºC for moisture (background information for NH_3 and N_2O emissions) and preserved for N content analysis. Analysis will include Total N, total C, Urea N, Nitrate-N and NH_4-N contents. In addition, each flux chamber covered area were sampled using the same corer after each flux chamber measurement, providing moisture and N content for N_2O emissions study. Each pile was divided into four sections for manure sampling and flux chamber measurements (Figure 6).

Figure 6 Flux chamber measurements on manure piles

3.1.4.1 Measurement of gas emissions

Experiment 2

3.1.4.1.1 Trace gas analysis system

The general plan for the measurement of gas emissions from the feed lot systems is shown in Section 3.1.3.3.1. ECOtech trace gas systems (TGS) with NH_3 and CH_4 chemiluminescence analysers were deployed at each site (15m from each pen). The sampling and analysis system was an air sampling tower, air sample delivery tubing, suction pump, automated switching solenoids/manifold (with data logging system) and the TGS (Figure 4). TGS automatically sampled and analysed air samples drawn from five height levels (0.25m, 0.5m, 1m, 2m and 3m) in the centre of each pen. Five separate air lines (sample delivery tubing) for each height delivered air samples from the centre of the pen to the TGS. Teflon tubing was used reflecting its non-absorbance of NH_3. A custom-made hot water circulation system was used to maintain the temperature above 30°C thereby preventing NH_3 or condensation build-up in the sample tubing. Five Teflon air lines were tied around the hot water supply pipe and the whole structure was then wrapped with industrial grade outdoor thermal insulation. The switching solenoid supplied air samples to the TGS from each sampling height in a known sequence and measurement of trace gas concentrations were made at every 6 minute intervals. This data was used to calculate the vertical profile of individual trace gas concentrations to 3m. A 2D wind sensor was mounted on the sampling tower at each sampling height and wind speed and direction was recorded at one minute intervals. Trace gas fluxes were calculated as 30 min averages.

3.1.4.1.2 FTIR and open path laser measurements

An OP-FTIR spectrometer was installed on the southern side of paddock aligned with each experimental feedlot (see Section 3.1.3.3.1). The OP-FTIR path-length was 104.56 m and 100.93 m for high protein (HP) treatment feed paddock and low protein (LP) control feed paddock, respectively. OP-FTIR was set up at the height of 1.18 m and 1.37 m above the ground at the HP and LP paddock, respectively. Line-averaged concentrations of CH_4, CO_2, N_2O and NH_3 were measured at 3-min intervals simultaneously. A weather station was set up in the eastern side of experimental sites to avoid the interference from animals and instruments. A 3-D sonic anemometer (CSAT3, Campbell scientific, USA) was mounted on a mast at 2 m above the ground. Three temperature sensors (TCDirect, Australia) were also mounted at 1.0, 2.0 and 3.0 m above the ground. Data was averaged to 15-min periods. Wind speed, wind direction, ambient temperature and the variance and covariance of winds were recorded at 10Hz by a datalogger (CR23X, Campbell Scientific, Australia). The coordinates of all sensors and OP-FTIR spectrometers were measured using the global positioning system (GPS) (Garmin, GPSMAP 62S, USA). Backward Lagrangian stochastic dispersion model (WindTrax 2.0, Thurder Beach Scientific, Canada) was used to simulate gas fluxes emitted from the source (feedlot) by computing concentrations and winds turbulent statistics into flux. Wind conditions are key factors modelling the data through WindTrax. The criteria for obtaining the valid data are: $u^*> 0.15$ m/s, surface roughness is < 0.9 m and atmospheric stability length is $|L| >5$ m. Furthermore, the touchdown of simulated particles at source area is > 10% of the total area. A control gas release (recovery) trial was conducted 10 days after the end of the feeding trial (4 November 2013) once animals and manure were removed from the sites. Three gases of CH_4 (> 89%), N_2O (> 99%) and NH_3 (> 99%) (Coregas, Australia), were released in the centre of the southern paddock, and the flow rate was controlled by a mass flow controller (Alicat MCS10SLPM, MC20SLPM, USA). The flow rate of CH_4, N_2O and NH_3 was 8, 8 and 5 standard L/min for an hour for each gas.

Experiment 3

3.1.4.1.3 Close path NH3 sampling scheme

Aerodyne Quantum Cascade Laser Mini Monitor (QCL-NH3) and ECOTECH 9842 NH_3 chemiluminescence analyser were deployed at each site (approx. 30m from each pen: Figures 7 to 10). The sampling system consists of an 4-m sampling mast located in the centre of animal pen, 5 sampling inlets (virtual impactors) on each mast (0.25m, 0.5m, 1m, 2m and 4m), ¼ inch OD Teflon sampling tubing, high capacity dry scroll pump (Agilent Triscroll 600), and automated air pneumatic switching manifold. A 5-m post was put up on each TGS to provide a background source. A custom-made hot water circulation system was used to maintain the temperature of sampling lines at 40^oC constantly thereby preventing NH_3 condensation or build-up in the sample tubing. The Teflon sampling lines were tied around the hot water supply pipe and the whole structure was then wrapped with industrial grade outdoor thermal insulation.

Sampling frequency of the mast was at 5-minute interval of each sampling height, rounding to one hour each cycle: background (1-5 min); 4 meter (6-10 min); 2 meter (11-15 min); 1 meter (16-20 min); 0.5 meter (21-25 min); 0.25 meter (26-30 min and 31-35 min); 0.5 meter

(36-40 min); 1 meter (41-45 min); 2 meter (46-50 min); 4 meter (51-55 min); background (56-60min).

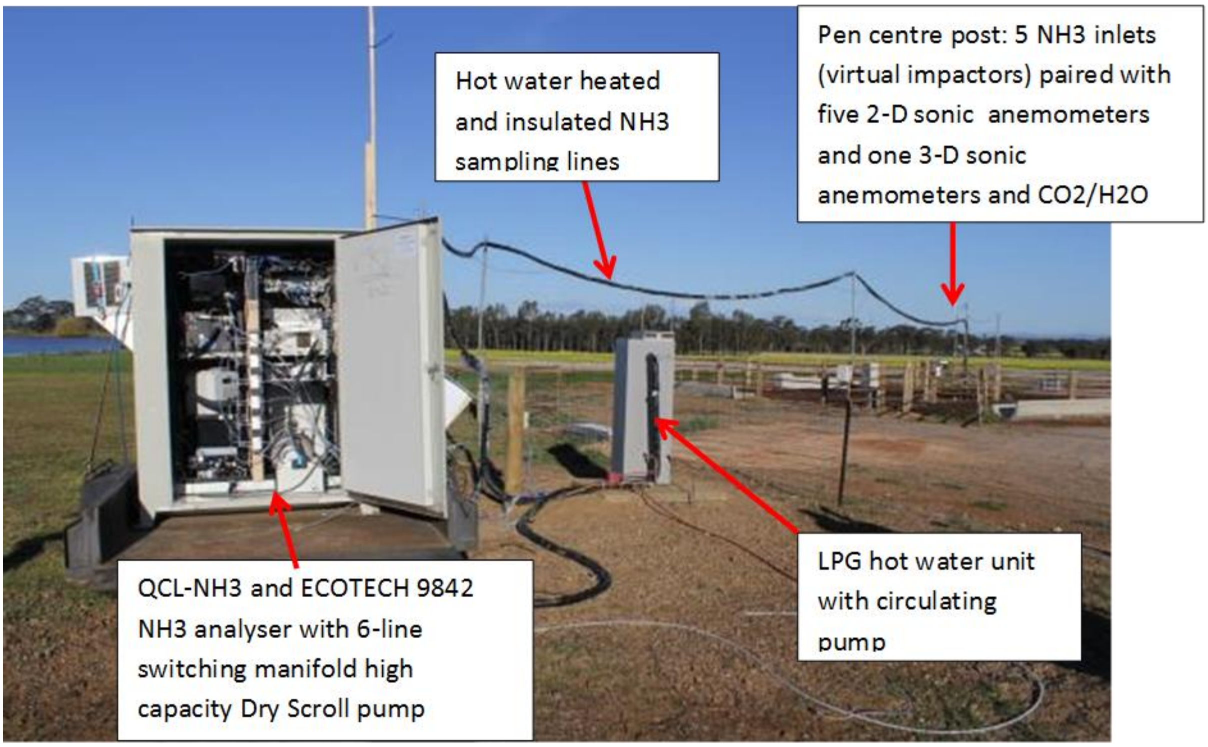


Figure 7 Close path NH3 sampling system

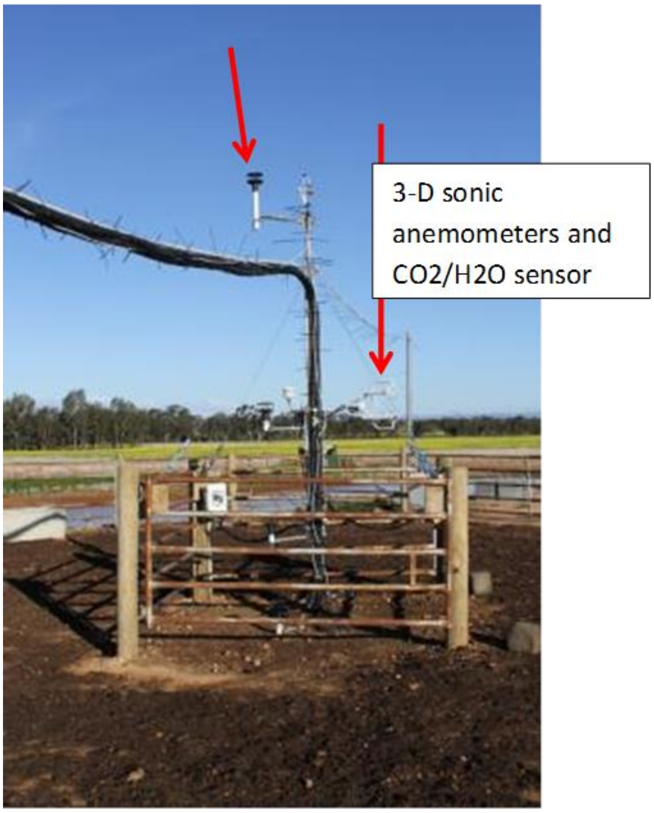


Figure 8 Sampling mast at the centre of pen

3.1.4.1.4 Close path N2O/CH4 sampling scheme

Four sampling inlets (no virtual impactor or pressure reducer) were installed at the same NH3 sampling mast (at heights of 0.5 m, 1m, 2m and 4m) via 3/8 inch OD nylon tubing (non-insulated) to sample N2O/CH4 into site office. One CPFTIR and one QCL-N2O/CH4 were deployed to take the task for N2O/CH4 measurement. One 5-m height background is sampled at site office, which is located in the middle between the two pens. The sampling schedule was: first half hour – background, 4 m, 2 m, 1 m, 0.5 m at 6 minutes interval for south pen (lignite), and second half hour same sequence for control pen. Flow restriction was applied only at instrument inlet therefore flow rate in tubing is low at near ambient pressure, which works as ballast.

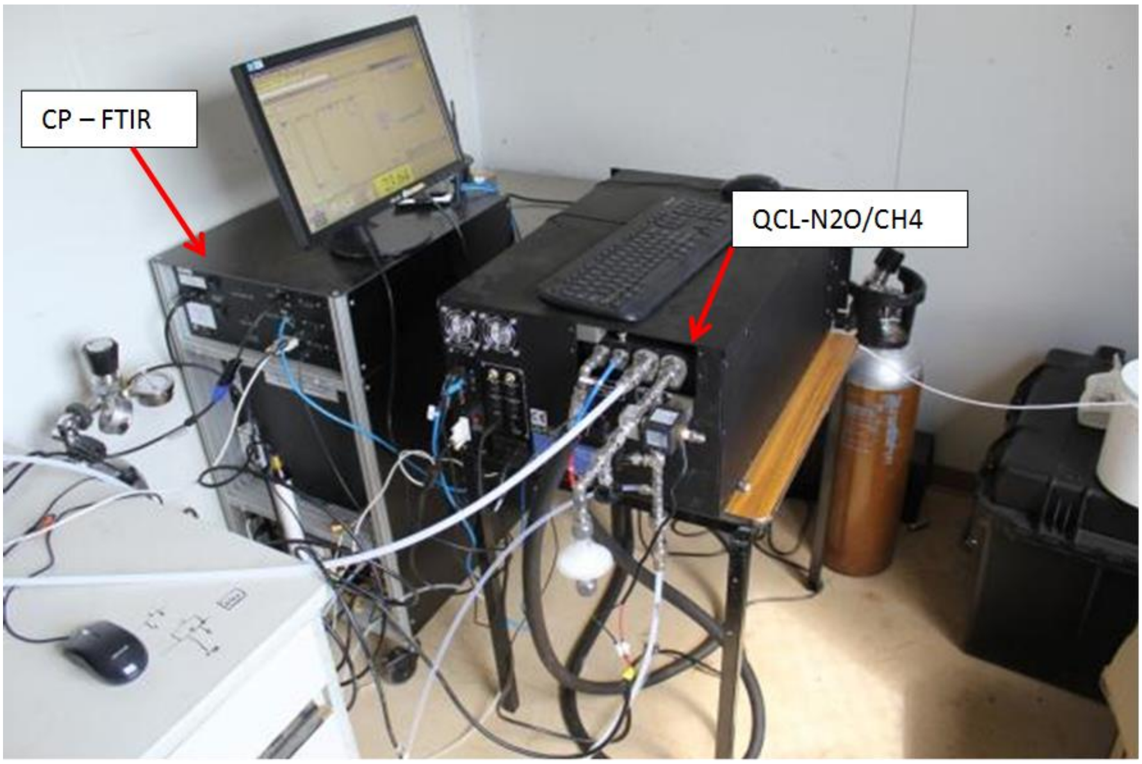


Figure 9 Close path N2O and CH4 analysers

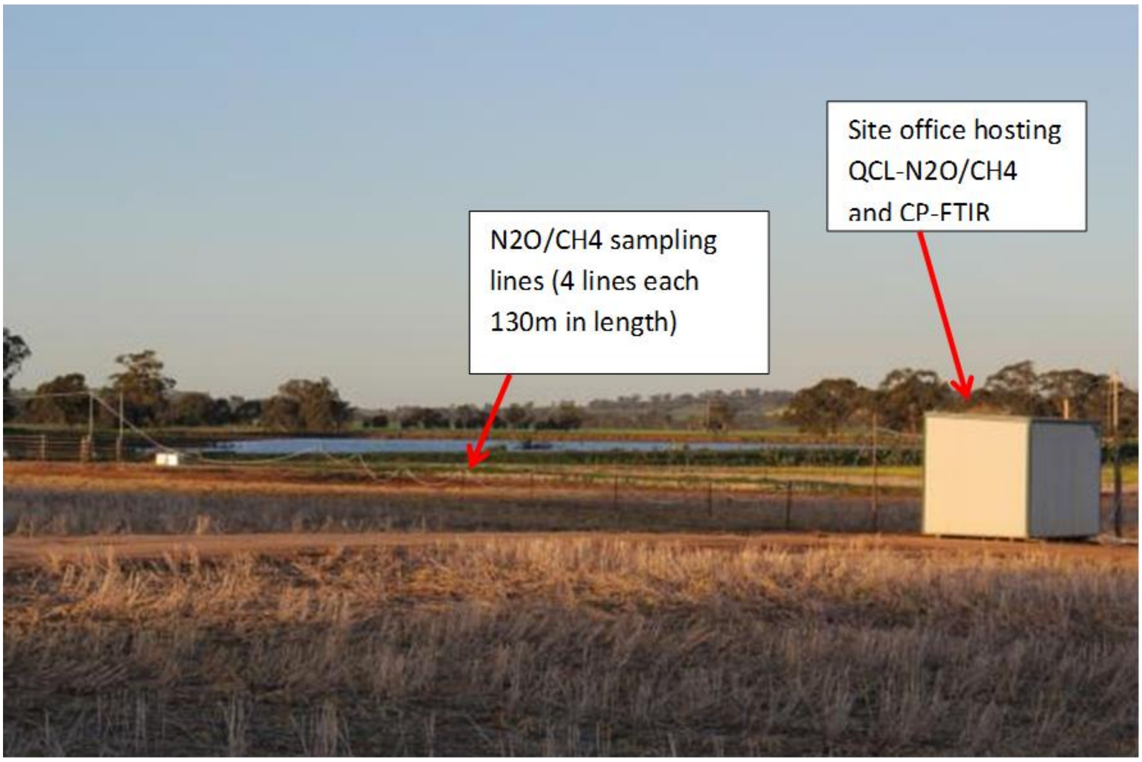


Figure 10 Close path N2O and CH4 sampling system

3.1.4.1.5 Open-path FTIR (NH_3/N_2O/CH_4) sampling scheme

Two open-path Fourier transform infrared spectrometers (OP-FTIR) (Matrix-M IRcube, Bruker, Germany) were located at the south of cattle pen to measure line-average N_2O, NH_3 and CH_4 concentrations simultaneously downwind from cattle pens (Fig. 2). A motorised tripod head (University of Wollongong), for the first time, was utilized to scan in the vertical and horizontal planes which makes OP-FTIR measurements on two different paths (at different heights). In this study, OP-FTIR with motorised scanner was pointed to retro reflectors located at the east and west. Two and half-min line-average concentrations of N_2O, NH_3 and CH_4 were measured at each path, with path length of 100 m. This employment enabled to detect the difference between two paths during northerly wind periods. A weather station coupled with a 3-D sonic anemometer (CSAT3, Campbell Scientific, Logan, UT, USA) was located in the eastern side of experimental sites to avoid the interference from animals and instruments. Fifteen-minute intervals of wind speed, wind direction, ambient temperature and the variance and covariance of winds were recorded at 10Hz by a datalogger (CR23X, Campbell Scientific, Australia). The coordinates of all instruments were measured using the global positioning system (GPS) (Garmin, GPSMAP 62S, USA).

Inverse-dispersion technique (Backward Lagrangian stochastic dispersion model, BLS) using WindTrax 2.0 (Thurder Beach Scientific, Canada) was employed to simulate gas fluxes emitted from the source (cattle pen) by computing line-averaged concentrations downwind and winds turbulent statistics. Background concentration of the cattle pen was also provided. Wind conditions are key factors modelling the data through WindTrax. The criteria for obtaining the valid data are: $u^* > 0.05$ m/s, surface roughness is < 0.5 m and atmospheric stability length is $|L| > 5$ m (Bai et al., 2015). Furthermore, the touchdown of simulated particles at source area is > 30% of the entire cattle pen assuming cattle evenly distributed in the pen.

3.1.4.1.6 Manual chamber measurements of N2O and CH4 surface emissions

Experiment 2

N2O was sampled twice a week (every Tuesday and Friday) using 50 x 50 x15 cm pre-fabricated thermally insulated sampling chambers. Four chambers were placed across the pen covering the variability in manure/urine distribution on the pen surface and the sampling followed a protocol developed based on the DAFF methodology guidelines (http://www.daff.gov.au/__data/assets/pdf_file/0003/2276490/aotg-guidance-for-on-farm-measurement-final.pdf).

Experiment 3

Static chambers were conducted on (12pm-1pm) 23rd Sept 2014 for the first time, then each Tuesday and Thursday until end of the project. Each chamber has a size of 50cm x 50cm x 50cm. Each sample has 20ml injected into pre-evacuated 12ml Labco vial. The sampling time was 11am-12am on each Tuesday (south half of each pen) and Thursday (north half of each pen) with a sampling frequency of 10 minutes interval after initial sample, 0 min, 10 min, 20 min, 30 min and 40 min for five samples (Figure 11).

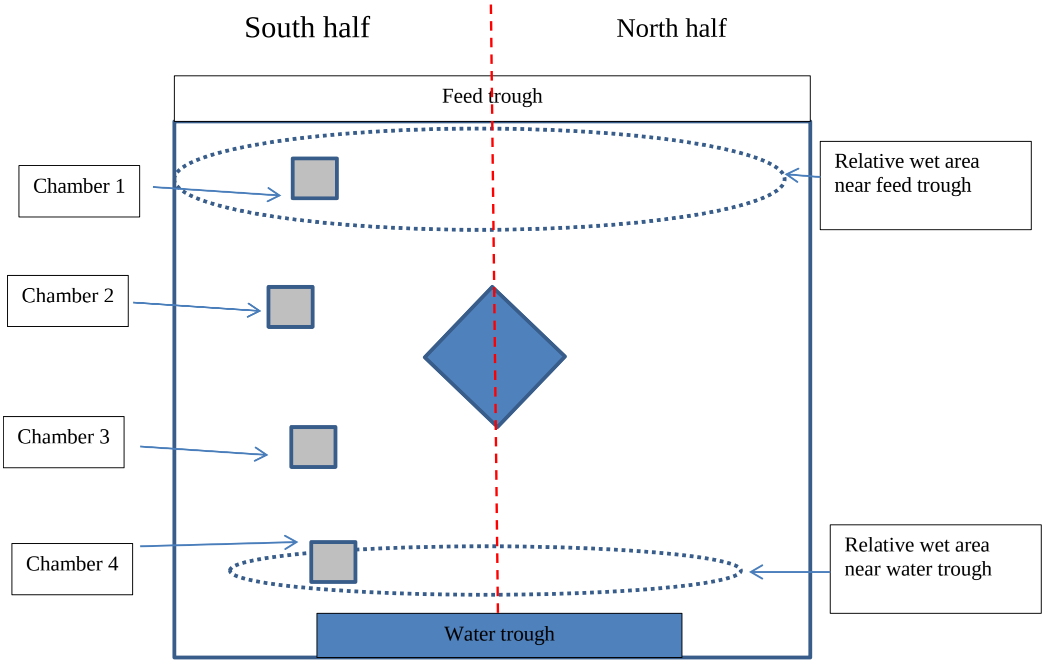


Figure 11 Flux chamber sampling at pen surface

3.1.4.2 Crop trial

A field experiment was conducted at the University of Melbourne, Dookie Campus (36.37°S, 1450.7°E), 220 Km North of Melbourne, Victoria, Australia from 23rd December 2015 to 26th January 2015. The topography of the experimental site was a flat underlain with a loamy clay (Nalinga loam) and well-drained soil.

3.1.4.2.1 Treatments and management

Manure was collected from the pen surfaces of both treatments at the end of the measurement periods (7 October 2014 and 23 December 2014). A replicated plot design (randomised block design with 6 treatments and three replicates per treatment) was installed with the aim of detemining soil N transactions from a range of treatments measured under growing conditions. A forage sorghum crop was established in late December 2013 and harvested after 3, 6 and 12 weeks of growth. The treatments were:

1. Control (no manure of N fertiliser)
2. Urea (98 kg N/ha)
3. Experiment 3 – Period 1 control manure (harvested 7 October 2014)
4. Experiment 3 – Period 1 Lignite manure (harvested 7 October 2014)
5. Experiment 3 – Period 2 control manure (harvested 23 December 2014)
6. Experiment 3 – Period 2 Lignite manure (harvested 23 December 2014)

3.1.4.2.2 Sorghum biomass sampling, and analyses of N recovery

Dry biomass was measured at 21, 49 and 100 DAS by oven-drying at 60℃ for 7 days.

3.1.4.2.3 Data analysis

Data were subjected to analysis of variance (ANOVA) using Minitab 16 statistical software package (http://www.minitab.com/en-us/) to determine the significance of treatment effects. Least significant differences (LSD) were used to test for differences among the treatment means. Differences were assigned based on a significance of $P \leq 0.05$.

3.1.5 Modelling of greenhouse gas emissions from feedlot systems

Model framework and boundaries: The model framework and boundaries for the feedlot environmental foot print were developed using ISO 14040:2006 (Environmental Management – Life Cycle Assessment – Principles and Framework) and ISO 14044:2006 (Environmental Management – Life Cycle Assessment – Requirements and guidelines). Briefly, the crop production boundary to determine fertiliser N inputs and biological N fixation was described as 'off-farm' and considered direct inputs to crops (Leach et al. 2012) that were used to formulate the rations of beef cattle.

Estimation of crop N inputs (after Leach et al. 2012): Crop N inputs were calculated by determining the recommended application of N fertiliser plus potential biological nitrogen fixation minus the N not taken up by the crop as determined by the nitrogen retained in the harvested component of the crop (i.e. average yield of the crop component multiplied by the average N content). The crop N input was then modified (inflated) by the N lost through processing of the crop for animal feed (for instance dehulling of pulses, estimated field losses associated with hay production (Ozkan et al. 2015) or collection of food processing by-products (e.g. wheat middlings used in the lignite study). This approach modifies that of Pierer et al. (2014) but is aligned with production steps 1 to 3 resulting in an outcome of production step 5 (Feed N consumed) for the calculation of virtual N factors.

Feed and animal management: The experiments conducted at Dookie (36.3° S, 145.7° E), Victoria, Australia were modelled using NRC (2007). Feed composition and animal performance (feed intake, live weight, live weight gain and carcass yield) from the feeding trials were used as inputs into the NRC beef cattle model (LRNS, 2012; Fox et al. 2004). Predictions of animal growth, N retained in the carcass, rumen N transactions, faecal and urinary N outputs were made in a comparable fashion to Steps 4 and 5a of Pierer et al. (2014).

Measurement of N containing gas emissions from manure used the approach of Bai et al. 2015 (a & b). Raw emissions data were processed using the backward Lagrangian stochastic (bLS) dispersion model described by Flesch et al. (2004) with mapping capabilities to estimate fluxes of each gas. Prior to processing, data was screened according to Flesch et al. (2004 & 2007). To model the fluxes of each gas during each 15-min emission, 50,000 trajectories were released along the measurement path length to simulate atmospheric dispersion. The fluxes were then calculated on a per capita (animal) basis. Emissions were assumed to occur from the pen surface, however in reality, some gases came directly from the breath of cattle (enteric emissions) that represent elevated point sources. Furthermore, the emissions were assumed to represent a spatially uniform source reflecting pen size and stocking density (McGinn et al. 2014). Upwind concentrations were not measured continuously. Gas samples were taken daily and analysed using GC-MS. Cross-calibration between concentrations measured using GC-MS and open path FTIR measured at the same time was conducted. Consequently, a constant background concentration for CH_4, N_2O and NH_3 of 1721 ± 1.4, 320 ± 0.5 and 1.5 ± 0.9 ppb, respectively was assumed.

Modelling stockpile N emissions (N: P ratio) To estimate dry solids and N volatilization losses from manure stockpiles, 5 samples of manure were collected from each manure harvested from S1, S2, C and L1. Apparent dry solids and N volatilization losses were estimated based on the change in the N: P ratios of diets and air-dried pen manure using Todd et al. (2005):

N volatilization (% of intake) = (N: P of diet − N: P of manure) / (N: P of diet)

Model outputs were calculated on the basis of g N to achieve feed N intakes associated with a single animal in one 24 hour feeding period. Proportionality of a range of indices was calculated to identify the magnitude of inputs and losses of N from each compartment of the model as well as the overall N foot print. Proportional data is represented as % in the text or equivalent of kg/100kg.

4 Results

4.1 Literature review on alternative approaches to N loss mitigation updated to incorporate recent scientific knowledge

A literature review was conducted to identify alternative approaches to reduce N losses from feedlot pens. Four loss pathways were identified (nitrate, ammonia, nitrous oxide and volatile N content of solids. These pathways were characterised and options to abate losses of N from each were identified. These processes were broadly grouped into chemical inhibition of microbial processes, capture of N through chemical reactions, physical adsorbents or organic amendments.

4.1.1 Characterization of N losses from feedlots and manure management systems

Four sources of nitrogen loss from feedlots and manure management systems can be identified: nitrate, ammonia, nitrous oxide, and the so-called volatile N content of volatile solids.

Nitrate and volatile N content will not be considered in this brief review. However it is worth noting that direct nitrate losses through the feedlot pen surface should be minimal if the pad has been constructed according to the codes of practice (National Beef Cattle Feedlot Environmental Code of Practice, 2012 & National Guidelines for Beef Cattle Feedlots in Australia, 2012). The pad therefore has a low infiltration rate to water and hence low migration of nitrate, ammonium or other dissolved solutes. Run-off directly from the feedlot pad is however a potential source of nitrate to surface or ground water. Again, if the feed pad has been designed according to the codes of practice then the losses associated with this pathway are proposed as negligible. It should however be noted that these potential pathways for nitrate loss have not been accurately quantified. Nitrate losses to drainage water only become a problem for feedlot systems if harvested manure is poorly managed or the recycling of manure to agricultural land exceeds the propose crop requirements of the soils receiving manure (GRDC, 2010).

The volatile N components of manure are characterised as ammonia, nitrous oxide and a range of volatile amine and indole compounds. Ammonia and nitrous oxide will be considered separately. There are a number of volatile amine and indole compounds that are a result of degradation of the microbial biomass in manure under anaerobic conditions. These compounds occur in feedlot systems when the pad become saturated or manure and lagoon management is poor. They are a major component of odour but do not contribute substantially to the overall N losses from feedlots.

Ammonia volatilization is highly variable and is influenced by the amount of total nitrogen (TAN), temperature, wind speed, pH, chemical and microbiological processes, diffusive and convective transport in the manure, and gas phase resistance in the boundary layer above the source (Arogo et al., 2006). Little is known about the total annual emissions from Australian beef feedlots, and few robust long-term measurements have been made. US Environmental Protection agencies estimated in 2002 that 6.8 million tonnes of manure N was excreted from all livestock production systems. Approximately 18.3% was recovered and applied directly to cropping systems and 26.4% was transferred from the manure management systems to the environment through ammonia volatilization. Other losses included leaching and run-off during treatment, and storage and transport before application to the soil. These losses are similar to those reported in Europe (10 to 40% losses of total N as direct emissions of ammonia from manure: UNECE, 2014).

There have been many reviews on the emissions of nitrous oxides from organic manures. Nitrous oxide is largely emitted from manure and soils as a result of microbial nitrification and denitrification. Nitrification is an aerobic process that oxidises NH4+ to NO3- with N2O as a by-product, while denitrification is an anaerobic process that reduces NO3 − into N2, with N2O as an obligatory intermediate. Nitrification is often an essential prerequisite for denitrification and plays a dominant role in the conversion of N inputs from urine and urea or ammonium-based fertilisers into NO3–. For instance in animal production systems high loadings of nitrate in the soil (for instance under poorly consolidated feed pads) and low soil aeration (wet or compacted soils) can result in high N2O emissions (Klein et al. 2006). Nitrous oxide has received considerable attention recently and has become a major measured component of the total N budget of agricultural systems reflecting its potent global warming potential. Chen et al. has measured and reported nitrous oxide emissions from feedlot manure managed systems in a number of projects managed through MLA (for instance FLT.331).

There is some conjecture over the correct reporting metric to partition the ammonia emissions from the total N input. The main area of contention is the flux of a gas (for instance ammonia) may reflect that day's manure or a number of the previous days manure. However, for the purposes of this report, the metric will be reported as per head per day. If ammonia emissions are reported as g NH3-N/kg total N consumed, the average emissions from feedlots measured in the experiments outline in Table 1 was 54.9% of total N input or 148.2 g NH3-N/head/day. This exceeds current estimates from NIR (2012) of 22.3% of total N input or 60 g NH3-N/head/day. Measured emissions of nitrous oxide (3.79 g N2O-N/head/day) were comparable to those estimated from NIR (2012) or 4.91 g N2O-N/head/day.

4.1.2 Strategies to reduce nitrogen losses from feedlot manure

There are a number of approaches to reduce the losses of nitrogen from manure. These are broadly grouped into chemical inhibition of microbial processes, capture of N through chemical reactions, physical adsorbents or organic amendments.

4.1.3 Chemical inhibition of microbial processes

There are two important processes in manures and organic wastes that drive the conversion of N held in an organic form to ammonia and/or nitrate. These processes are nitrification and urea hydrolysis (mineralization/ammonification). There are a range of technologies that reduce these processes.

i) Nitrification inhibitors are designed to delay microbial oxidation of NH4+ to nitrite for periods from several weeks to months. They are very effective in preventing microbial nitrification and subsequent denitrification (Weiske et al., 2001; Zerulla et al., 2001). There are a number of nitrification inhibitors known, however very few of these candidate compounds have gained commercial use. Two compounds that have been used in agriculture are dicyandiamide (DCD) and 3, 4-dimethylpyrazol phosphate (DMPP). There are a number of studies that have demonstrated the efficacy of these compounds in the field, for instance in studies that have used DCD, reductions in nitrous oxide and nitrate leaching from soils amended with slurries or receiving N fertiliser have been noted (Weiske et al., 2001; Majumdar et al., 2002; Zaman et al., 2009; Cui et al., 2011; Di and Cameron, 2012; Moir et al., 2012; Pfab et al., 2012). DMPP addition to soils receiving N fertiliser or slurries has been shown to have equivalent or better abatement in nitrous oxide emissions and comparable reductions in nitrate leaching. However, the main advantage of DMPP over DCD is the reduced rate of application of the product to achieve comparable reductions in N losses (Weiske et al., 2001; Belastegui-Macadam et al., 2003; Di and Cameron, 2012). There are however a range of factors that affect the efficacy of DCD or DMPP including rate, time and method of application (Barth et al., 2008; Verma et al., 2008; Zaman and Blennerhassett, 2010; Zaman and Nguyen, 2012); post application management of the cropping area (irrigation, type and method of application of ammonium (or urea) fertilisers (Sanz-Cobena et al., 2012); climate (rainfall and temperature: Shepherd et al., 2012); and soil properties (moisture, pH, texture, organic carbon and mineral N, Barth et al., 2001; Shepherd et al., 2012).

ii) Urease inhibitors are designed to reduce ammonia emissions from the surface of soils immediately after fertiliser application especially if the fertiliser has been surface-applied (Wang et al., 1995; Grant and Bailey, 1999). The inhibitor is designed to inhibit urease activity in the soil microbial community, thereby reducing the rate of urea hydrolysis and allowing more time for the urea and ammonium to be retained in the soil. This reduces ammonia volatilisation from the surface of the soil (Grant et al., 1996). There are a range of compounds that can inhibit microbial urease activity for instance NBPT (N-(n-butyl) thiophosphoric triamide: AgrotainTM), CHPT (cyclohexylphosphorictriamide), PPDA (phenyl phosphorodiamidate), and ammonium thiosulphate. NBPT has been shown to inhibit urease activity at low concentrations under laboratory conditions (Carmona et al., 1990; Gill et al., 1997) by forming stable complexes with the enzyme (McCarty et al., 1989; Manunza et al., 1999). One of the main problems associated with the use of NBPT or any other urease inhibitor is the ubiquitous nature of the enzyme as part of microbial N transactions and hence the low specificity of the compounds in terms of hyper ammoniating microbial communities. This situation may mean that in manure management, the product may have reduced efficacy reflecting the sheer extent, size and diversity of the microbial community present. In recent work by Chen et al. (DAFF-MLA funded B.CCH.1020) the use of NBPT was shown to be limited in the management of urease activity in manures derived from feedlots, have low efficacy within a week of application to the manure pad and not to be cost effective in its ability to reduce N based gases from manure. These observations are in contract to those of Varel et al. (1999) and Varel (1997) who suggest that NBPT, CHPT, and phenyl phosphorodiamidate (PPDA), can be successfully used to inhibit urease activity in cattle feedlot manure, based on laboratory scale and field studies.

4.1.4 Mineral and chemical amendments to reduce N losses from manure

There have been numerous studies using mineral and chemical amendments to reduce ammonia emissions from animal manures. The range of minerals and chemicals used include sources of phosphate, aluminium sulphate, calcium chloride, magnesium sulphate, strong acids and gypsum. All of these options have to be considered from a cost:benefit and an ‘ease’ of management point of view. The trade-off between the cost of adding another chemical or mineral to manure to retain ammonia needs to reflect excellent capture of the ammonia in the manure, the ability of the co-amended manure to re-release N when applied to the soil and any potential adverse environmental effects of the mineral/chemical added to retain the volatile N phase of the manure. For instance, the normal rates of addition of phosphate (single or triple) of 50 kg/tonne manure have been shown to reduce ammonia losses from dairy slurries by 28 and 14% respectively (Iowa State University Extension Service, 2004: http://www.extension.iastate.edu/publications/PM1971a.pdf). In these cases, the supply of P from the amendment would be 4.4 kg P/tonne (fresh weight) and 10.3 kg P/tonne (fresh weight) over and above the rate of addition of P from the manure per se. Similar abatement potentials for super and triple phosphate are noted if the product is surface applied to the manure (33 and 24% reductions in ammonia respectively) however the likelihood that continuous application of the phosphate source to the feed pad is high to maintain the reduction in ammonia emissions. This would lead to an increase in phosphate in the manure pad and may pose a risk to water sources (and the lagoon) under heavy rainfall events. Gypsum and calcium chloride have also been shown to be effective as an abatement technology for N based gas emissions from manure. These products are generally less expensive than phosphate based products but their abatement potential is lower (13 and 8% reduction in emissions at comparable rates of addition, respectively: Iowa State University Extension Service, 2004).

Ammonia emissions can be reduced by the chemical amendments that directly acidify the manure pack (for instance alum: ($Al_2(SO_4)_3$) or sulphuric acid (H_2SO_4)). Hydrolysis of alum to yield Al^{3+} ion releases three free protons (H^+) and hence a rapid reduction in pH of the manure. The decline in pH of the manure reduces ammonia emissions as the ammonium (NH_4^+) ion dominates the ammonia – ammonium exchange. Ammonium is retained in the manure through cation exchange liberating more H^+ or other cations resulting in increased acidification (Shi, Parker et al. 2001). Kithome et al. (1999) evaluated the efficacy of a range of mineral salts (reduction in ammonia emissions) that would lead to acidification of poultry manure ($CaCl_2$, $CaSO_4$, $MgCl_2$, $MgSO_4$ and $Al_2(SO_4)_3$. Reductions in ammonia emissions of 10 to 20% were observed after 20% mass/mass of $CaCl_2$ or $MgCl_2$ was mixed into the poultry manure and up to 74% reductions in ammonia emissions at the same rate of addition of alum. $CaSO_4$ and $MgSO_4$ were ineffectively in reducing ammonia emissions. Two issues arise; the rate of application (20% mass/mass) is prohibitively expensive reflecting the inefficient capture of ammonia and the impact of increased loadings of chloride and/or aluminium on the soil.

Two potential chemical methods that are more suited to liquids or wet manure are struvite precipitation or acidification through the addition of sulphur (in the form of iron pyrites). Struvite precipitation relies on the availability of Mg^{2+}, NH_4^+ and $H_2PO_4^-$ being available in the liquid phase of the manure/effluent (Ren et al. 2010; Uladag et al. 2005). Struvite (magnesium ammonium phosphate hexahydrate) is relatively insoluble and can be viewed as a slow release fertiliser. However, it is a significant problem in the management of pumps and pipelines that re-distribute effluents and liquids from lagoons and pumping stations insofar as it blocks or crystallises around valves. The precipitation of struvite is affected by pH, the chemical composition of the wastewater (degree of saturation of magnesium, ammonium and phosphate); presence of other ions (calcium and aluminium); ionic concentration of solution, and the temperature of the solution (Startful et al., 2001; Doyle & Parson, 2002). The addition of iron pyrites (FeS) to manures results in the release of elemental S and hence acidification of the manure mass as a result of sulphuric acid production (Banath and Holland, 1976 and Tiwar & Dwivedi, 1986). There is a lack of evidence on the efficacy of this approach and potential risks of the production of hydrogen sulphide under anaerobic reducing conditions in poorly managed manure stacks.

4.1.5 Adsorbents

There are a range of clay minerals and other products that could be used as adsorbents of ammonium, nitrate and nitrite in manure management systems. Many of the clay mineral adsorbents are classified as zeolites (aluminosilicate) that have moderate charge density (cation exchange capacity) and can liberate significant concentrations of H+. They are classified as temporary and/or repositories of cations and anions and re-release of the cation or anion depends on the chemistry of the soil solution. The most commonly used adsorbent is clinoptilolite (zeolite). Miner et al. observed a 60% reduction in ammonia emissions from dairy slurries after the addition of 1-4% (w/v) of finely ground clinoptilolite. These results were similar to those of Kephart (2004) who studied the effects of adding zeolite to dairy manure at a rate of 6.25% (w/v) and observed a 50% reduction of NH3 emissions from the manure. Zeolites can also be used to alter the pattern of ammonium release in the soil and hence reduce nitrous oxide emissions (Zaman and Nguyen, 2010). Even though these approaches show good potential, one of the main considerations of using zeolite is the impact of fine particles on overall dust burden in the feedlot environment. There are a number of potential implications of using these products for animal health and well-being as well as OH&S requirements for the feedlot labour force.

4.1.6 Organic amendments to manure management systems

Three approaches to reduce ammonia emissions from manure using organic amendments are briefly reviewed. The approaches are (i) manipulation of the C:N ratio, (ii) addition of biochar, and (iii) use of lignite.

(i) Manipulation of the C:N ratio of manure. Typically the C:N ratio of manures from ruminants range from 13 to 20:1. By adding saw dust or straw to the pen surface, an increase in the C:N ratio of the manure occurs resulting in greater N retention (Glendenning & Smith, 1999; Adams et al., 2004; Lory et al., 2012). Dewes (1999) demonstrated that by adding straw as an additional C source to cattle manure, N volatilization losses were reduced from 23.2 to 5.1% over a 14 day period. Aguerre et al. (2012) also observed reductions in ammonia emissions from manures and slurries receiving 2.2% (mass/mass) addition of chopped wheat straw and suggested the causal mechanism was increased acidification of the manure as a result of increased C degradation rates. The use of straws and other products to alter the C:N ratio of manure on feedlots is probably not feasible in practice; however it could be integrated at the point of pen clearance, stockpiling and on-farm compost production (addition to the windrow). The main detraction of using chopped straw or other products is the costs associated with purchase, processing and handling the straw before addition to the manure.

(ii) Biochar is a carbon rich by product that is produced from pyrolysis of organic matter under oxygen limited conditions (Kookana, Sarmah et al. 2011). There have been a number of studies that have demonstrated no, minor and substantial benefits of addition of biochar to soils (for instance Clough et al. 2010; Jeffery et al. 2011). It has been suggested that biochar addition to soils increases crop productivity by altering soil pH, soil organic carbon and total nitrogen content while decreasing the soil bulk density (Afeng et al. 2012; Wang et al. 2012). There is considerable conjecture over the mechanisms involved with abatement of N based gases from manures as a result of biochar addition. Rogovska et al. (2011) suggested that biochar may have an impact on the total availability of C in manure (reduce the availability as biochar C is refractory) but enhance soil aeration thereby reducing N2O emissions from manure co-amended soils (Clough et al. 2010 & 2012; Clough and Condron, 2010; Alho et al., 2012). More recently, this view has been rejected (see for instance Case et al. 2012 ;Knowles et al., 2011 and Yao et al., 2012) and it is thought that the reduction in nitrous oxide reflects increased water holding capacity of the soil and hence improved soil environment to increase microbial or physical immobilisation of nitrate. Yao et al. (2012) also suggests that biochar has considerable nutrient sorption characteristics and can bind ammonium from the soil solution. This is however still open to question. It is known that the feedstock source of biochar has a considerable impact on the chemical composition of the char (Singh et al. 2010; Knowles et al. 2011; Yao et al. 2012). Feedstock type affects microspace volume, surface area and potentially adsorbent capacity (Allen et al. 1997) but it is not known how these parameters affect nitrous oxide emissions or if they have any role in the abatement of ammonia emissions from manures. It has been suggested that biochar has a moderate ammonia adsorption capacity (Steiner et al., 2010). Taghizadeh-Toosi et al. (2010) demonstrated that a 45% reduction in NH3 volatilisation from 15N labelled ruminant urine applied to soil amended with 30 t/ha of low temperature wood biochar. Furthermore, N release to plants from ammonium-biochar complexes occurred rapidly and there was no impact of plant productivity compared to N fertiliser control treatments. These observations suggest that incorporating biochar into the soil can decrease NH3 volatilisation from urine and that the NH3-N adsorbed onto the biochar is available to soil processes that yield ammonium and nitrate (Taghizadeh-Toosi, Clough et al. 2012). The major detraction for the use of biochar as an amendment for feedlot systems is the limited supply of a consistent product (of known provenance) and the cost.

(iii) Lignite (or brown coal) is a low rank, high moisture coal that is mainly used to produce electricity (Ciuk and Piwocki, 1990). Even though lignite is categorized as a low rank coal, it has a high volatile organic content and can be gas and liquid petroleum products relatively easily (www.dpi.vic.gov.au/earth-resources; Geoscience Australia, 2012). Lignite has been used as a soil amendment for many years with the aim of building soil organic matter and humic acid content (Kwiatkowska et al, 2005; Katzur et al. 2002 & 2003). Lignite possesses several promising characteristics that make it a suitable amendment for manure, and potentially a good method of reduction ammonia and nitrous oxide emissions. Lignite is characterised as having low pH (Vorres, 2000; Bernal et al. 2009), high humic acid content (Kalaichelvi (2006), high CEC and adsorption capacity and high labile carbon content. making it a potential manure amendment in terms of reducing manure NH3 emissions (Rhoades et al. 2010). Lignite is well known for its adsorption properties for various industrial applications (Yuliana et al., 2010; Oussa, 1978 and Qi et al. 2011). Berg et al. (2009) showed that ammonia emissions from liquid swine manure could be reduced by 67% with the addition of 3% (mass/mass) of powdered brown coal but did not provide reasons for the abatement. Hwang et al., (2004) also reported similar results when adding 5 to 10% (mass/mass) brown coal to swine manure. Shi et al. (2001) and Abesekara et al. (2012) demonstrated similar retention of ammonia in cattle manure amened with lignite (again 5 to 10% mass/mass). Little is known about the use of lignite in large feedlot systems, however initial examination of laboratory studies (e.g. Kagimbo et al. 2013) have suggested substantial retention of ammonia in the lignite. Nitrogen use efficiency in intensive beef (feedlot) production systems is inherently low, resulting significant NH3 & N2O emissions from manure. These emissions may pose significant risk to the industry in the future reflecting potential changes in environmental protection policies or acceptance of N foot printing as an indicator of sustainability of the industry. The broader application of these technologies is not evident in the intensive animal industries reflecting the current practice of stockpiling of manure in the pen and outside prior to field application of manure with little or no pen amendment/mitigation strategy.

4.2 Nitrogen transformations in an artificial urine/manure system treated with a range of amendments

B.CCH.1020 used a combination of laboratory and field experiments to examine the effectiveness of the urease inhibitor N−(n−butyl) thiophosphoric triamide (NBPT), as “Agrotain”, applied to solid manure from a beef cattle feedlot. This work identified that even though there was substantial temporal and spatial variability in emissions from stockpiled manure, a chemical inhibitor such as NBPT could be used to reduce both ammonia and nitrous oxide emissions *in situ*. However, the main impediments to adoption of this technology were the frequency of application of the inhibitor to pen surfaces or manure stockpiles and the prohibitive costs associated with abatement (labour & operational costs, chemical input cost and potential low returns in N enriched manures/composts). These observations led to the development of pen amendment technologies such as the addition of lignite or potentially biochar. A small scale laboratory assessment of the efficacy of lignite and biochar was conducted (Table 2).

Table 2 Partitioning of retained N and losses from a rockwool test sytem amended with lignite or biochar at a rate equivalent of 4 kg/m^2.

Treatment	NH3 Losses (%)	NH4+ retained (%)	Urea Retained (%)	(%) Immobilised N	N Unaccounted (%)	Total
Control	35.0	6.5	24.1	5.1	29.2	100
Lignite	0.5	82.1	0.0	14.4	3.1	100
Biochar	41.1	2.1	0.3	39.2	17.2	100

Lignite was demonstrated as an effective method of retaining N from an artificial urine – urea hydrolysis test system (see Section 3.1.2). Urea added in the artificial urine was completely hydrolysed (reflecting the 0 to 0.3% of retained urea in the lignite and biochar treatments). The hydrolysis of urea liberates ammonia and carbon dioxide. Ammonia losses were minimal (0.5% of N added to the test system) in the lignite amended system with 82.1% of N liberated retained as ammonium rather than being immobilised on to the surface of the lignite (14.4%). These data suggest that the underlying mechanism for ammonia mitigation using lignite is driven by chemical and adsorbents (cation exchange) processes rather than incorporation of N into the organic matter phase. If this is correct, pH (lignite is acidic), cation exchange capacity, moisture content of lignite and local environmental conditions (especially feed pad surface temperature are critical to the success of lignite as an amendment. The ability of biochar to retain N was disappointing (did not varying substantially from the control – no amendment). Approximately 40% of N liberated from the artificial urine was lost as ammonia, 2% retained as ammonium and 39% immobilised on the biochar. The loss to the ammonia pathway is not surprising as biochar is alkaline and has a relatively low cation exchange capacity.

4.3 Effect of altering the crude protein content of the ration on direct and indirect greenhouse gas emissions

Ammonia (NH_3) emissions from intensive ruminant production systems are major environmental and economic issues reflecting their contribution to odour and total greenhouse gas emissions as well as the loss of nutrients. Recent work by Denmead et al. (2013) has demonstrated that ammonia is a potent indirect greenhouse gas with measured emissions of greater than 5000 kg/day from 20000 head beef feedlots in Queensland and Victoria. The impact of these emissions, through re-distribution of ammonia in the environment and subsequent conversion to nitrous oxide (direct greenhouse gas), is approximately 19t CO_2e (carbon dioxide equivalent)/day. These estimates were used as a blind data set to evaluate the potential of developing a CFI methodology based on manipulation of the ration of the growing beef animal (B.CCH.1086 : Proposed CFI methodology to mitigate greenhouse gas emissions from intensively managed ruminants by nutritional strategies). The findings of B.CCH.1086 were that a reduction in crude protein content of the ration had a marked impact on N voided and hence the production of nitrous oxide from manure management systems. However, the quantification of ammonia (and as a consequence nitrous oxide through deposition of ammonia) was not adequately modelled. The work conducted in Experiment 1 was aimed to (i) quantify N based greenhouse gas emissions (including the indirect GHG ammonia) from manure voided from beef cattle offered two levels of crude protein, and (ii) model those emissions to understand the bioconversion from feed N to N voided and (iii) identify if there are any deviations from current model approaches and actual measurements.

4.3.1 Rations and animal management

The rations offered in this study contained 40% forage (50% Lucerne hay and 50% perennial ryegrass hay) and 60% grain mix (Table 3). The high crude protein content grain mix comprised crushed barley grain (68%), sunflower (13.5%), lupin (13.5%) and cottonseed (5%) as the protein and energy source with ryegrass hay and mixed clover hay chaff as the fibre supplements. The estimated CP content of the whole ration was 17.5% and the ME content was 10.8 MJ/kg DM. The low crude protein diet comprised crushed barley grain (71%), lupin (10%), wheat millings (14%) and sunflower (5%) as the protein and energy sources with ryegrass hay and mixed clover hay as the fibre supplements (NRC, 2007). The estimated CP level was 14.5% and the ME was 10.8 MJ/kg DM.

Table 3 Ration composition (mixed ration)

Feed Analysis	LP	HP
Grain content (% of total diet)	60.0	60.0
Dry Matter (%)	83.6	83.9
Moisture (%)	16.4	16.1
Crude Protein (% of dry matter)	14.5	17.5
Acid Detergent Fibre (% of dry matter)	26.0	27.1
Neutral Detergent Fibre (% of dry matter)	43.2	45.4
Digestibility (DMD) (% of dry matter)	69.8	68.7
Metabolisable Energy (MJ/kg DM)	10.8	10.7
Fat (% of dry matter)	1.4	1.5
Ash (% of dry matter)	4.8	6.7

Original feeding plan of the trial was to maintain the grain content of the mixed rations closer to 70% of total intake, a level lower than typical feedlot rations used under commercial conditions where the grain contribution is greater than 75%. Further, initial modelling reported in Milestone 3 suggested that rumen pH would not fall below 6.3 for the mixed ration indicating both rations would pose little risk of acidosis. Acidosis occurred on two occasions reflecting poor mixing or the ration, poor intake characteristics of the forage component and animals selective feeding. An acidosis risk management strategy was implemented leading to: increased moisture content of the ration, addition of sodium bicarbonate (1.6%) and limiting the grain portion of the diet to 60% of total intake.

4.3.2 Feed intake and live weight change

The feed intakes for the LP and HP treatments are in Figure 12. The average feed intakes of animals managed on the LP and HP rations were 12.4 and 12.45 kg DM/day respectively or 2.84 and 2.77 % LW respectively. These feed intakes were comparable to commercial feed intakes for beef steers offered a moderate/high energy rations containing ca. 15% crude protein.

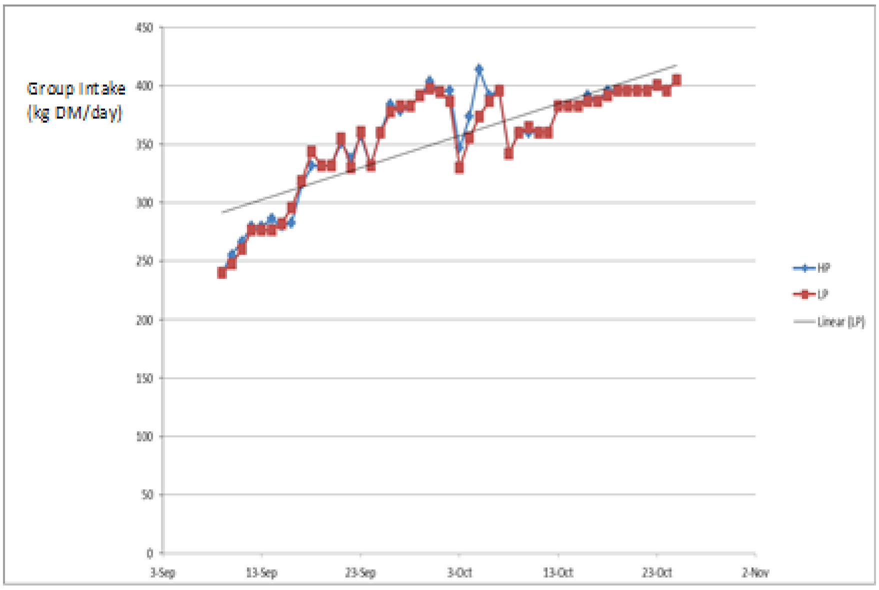


Figure 12 Feed intake (groups) of steers offered rations containing low (14.5%) or high (17.5%) crude protein rations

The gains in live weight over the experiment were estimated as 1.77 and 1.74 kg/day for the animals offered LP and HP respectively (Figure 13). These intakes were greater (0.32 and 0.33 kg/d) than those predicted by NRC (2007) using an energy limiting model but feasible reflecting the oversupply of crude protein in the ration. The mean intakes of N from the ration for LP and HP respectively were 301 and 350 g/day respectively.

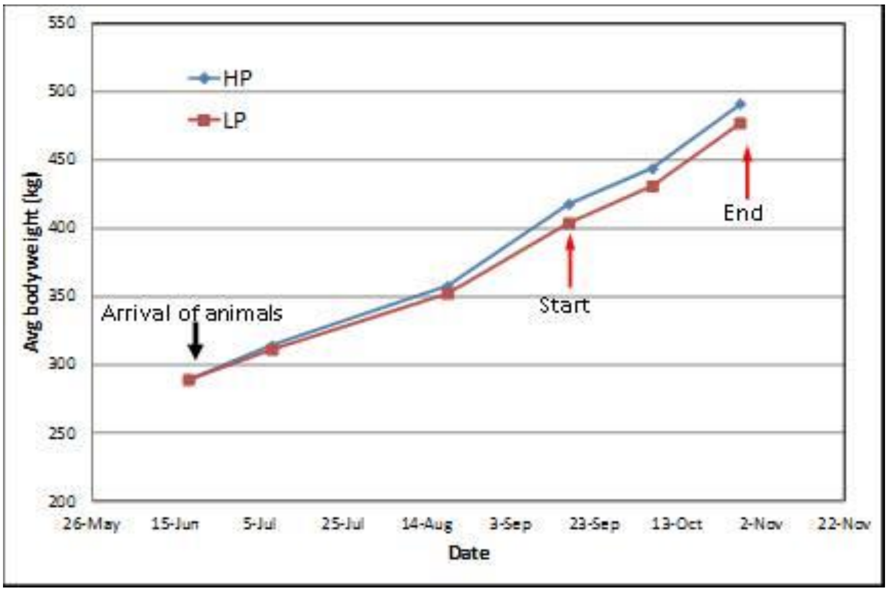
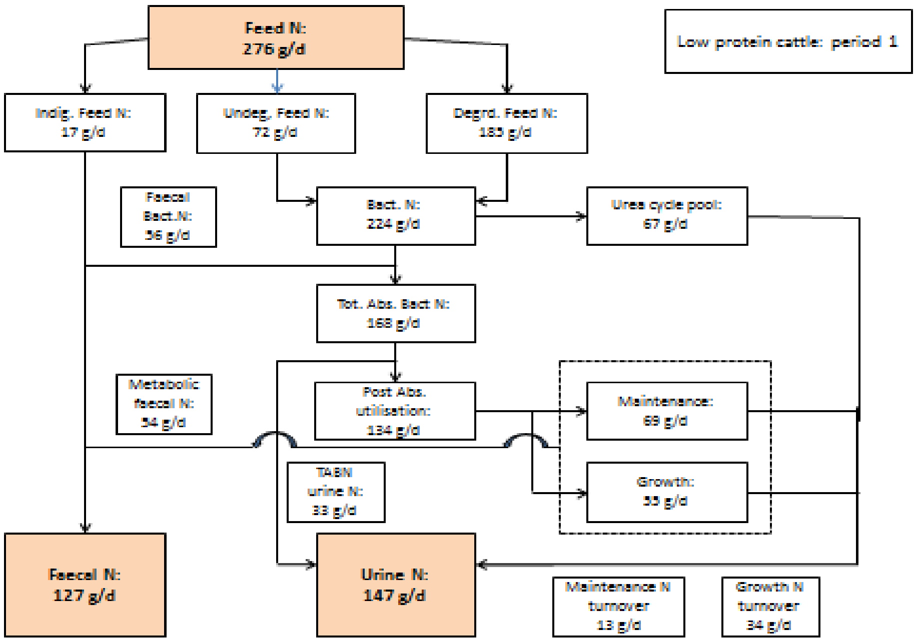
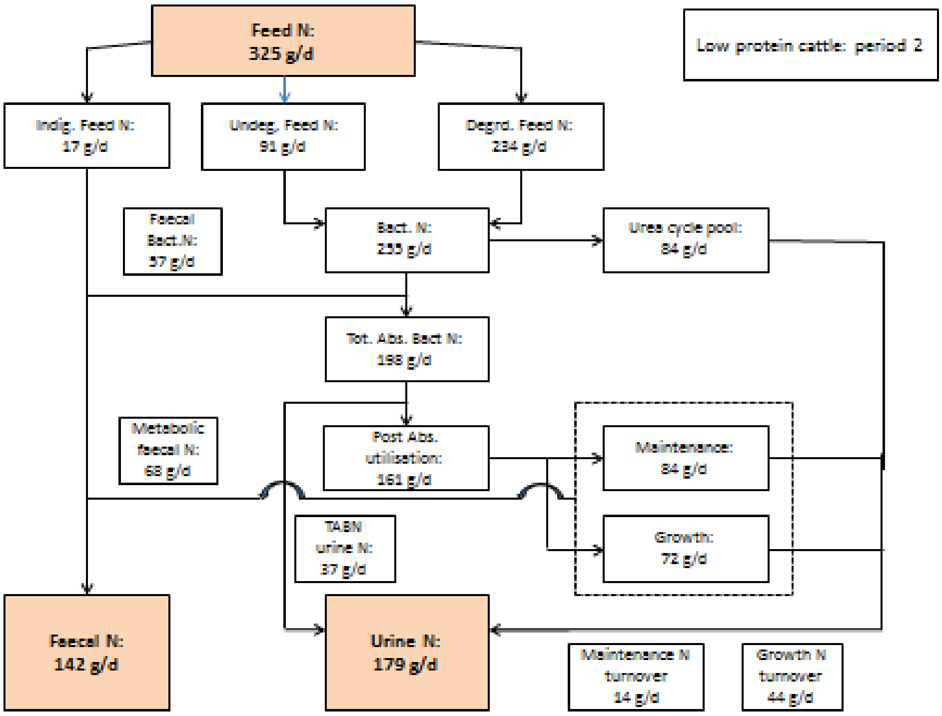


Figure 13 Live weight and live weight change of steers offered LP or HP rations

4.3.3 Manure yield, estimates of faecal and urinary N excretion and N metabolism in the animal

The yields of fresh co-mingled manure from the LP and HP treatments respectively were 21.4 and 23.7 t of fresh manure. Inevitably, when harvesting the fresh manure there is a degree of contamination of manure with material from the pen surface (clay and rocks). This was minimised during harvesting. NRC (2007) estimates of fresh manure (faecal and urine) outputs for LP and HP respectively were 18.5 & 17.2, and 18.1 and 21.1 kg/day. The increased urinary output in the HP treatment is expected reflecting the increased N intake. The estimated urinary and faecal N excretions for the LP and HP rations were 163 & 135, and 180 & 142 g N/day or total N excretion of 298 and 322 g N/day for LP and HP respectively. The data from the feeding trial was modelled using NRC (2007) to determine the N flow through animal metabolism. The data series was split into two periods – the first 21 days where animal intake average ca. 11.2 kg DM/day and the last 20 days where animal intake increased to ca. 13.5 kg DM/day. The outputs of the modelling are represented in Figure 14.




Feed N:
325 g/d
Low protein cattle: period 2
Indig. Feed N:
17 g/d
Undeg. Feed N:
91 g/d
Degrd. Feed N:
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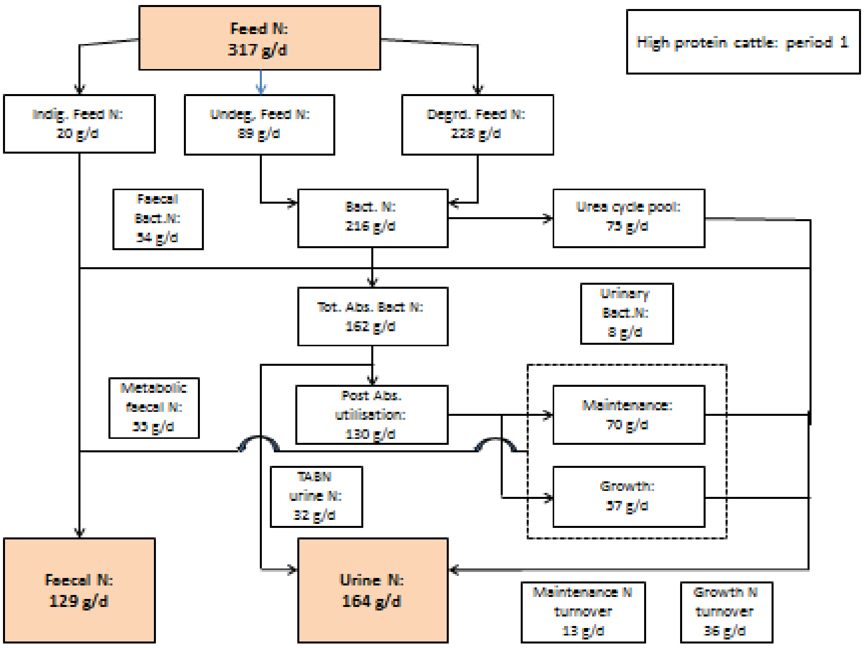
Faecal
Bact.N:
57 g/d
Bact. N:
255 g/d
Urea cycle pool:
84 g/d
Tot. Abs. Bact N:
198 g/d
Metabolic
faecal N:
68 g/d
Post Abs.
utilisation:
161 g/d
Maintenance:
84 g/d
TABN
urine N:
37 g/d
Growth:
72 g/d
Faecal N:
142 g/d
Urine N:
179 g/d
Maintenance N
turnover
14 g/d
Growth N
turnover
44 g/d


Feed N:
317 g/d
High protein cattle: period 1
Indig. Feed N:
20 g/d
Undeg. Feed N:
89 g/d
Degrd. Feed N:
228 g/d
Faecal
Bact.N:
54 g/d
Bact. N:
216 g/d
Urea cycle pool:
75 g/d
Tot. Abs. Bact N:
162 g/d
Urinary
Bact.N:
8 g/d
Metabolic
faecal N:
55 g/d
Post Abs.
utilisation:
130 g/d
Maintenance:
70 g/d
TABN
urine N:
32 g/d
Growth:
57 g/d
Faecal N:
129 g/d
Urine N:
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Maintenance N
turnover
13 g/d
Growth N
turnover
36 g/d

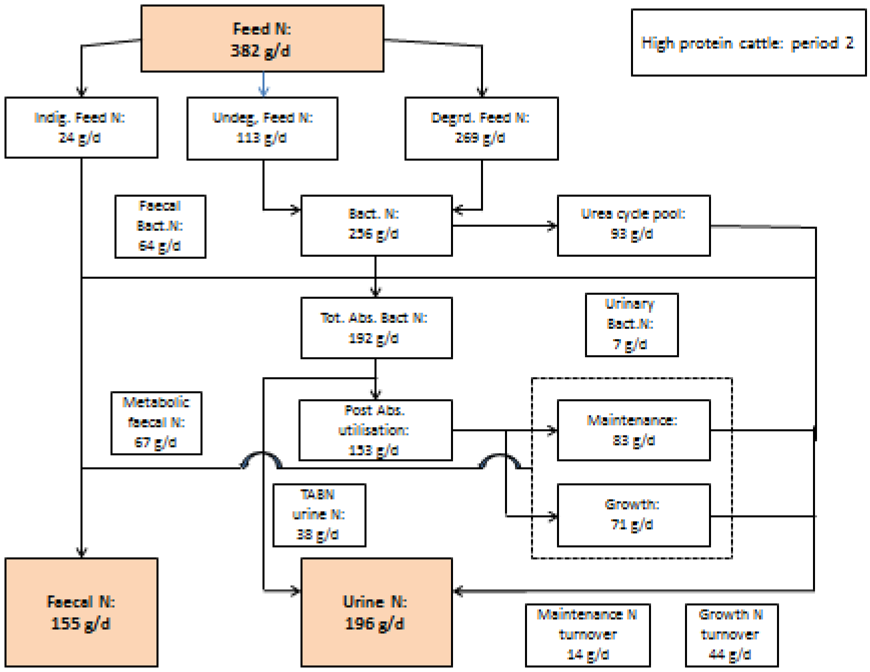
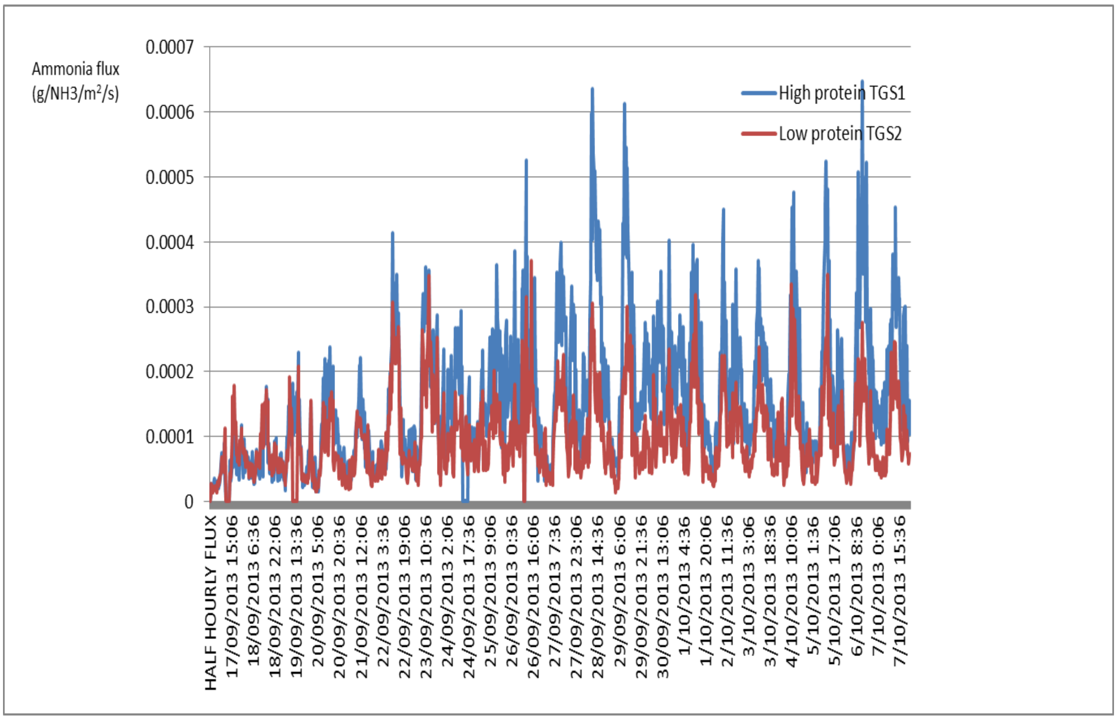


Figure 14 N metabolic modelling conducted using NRC (2007)

4.3.4 Gas emissions from feedlots

Integrated horizontal flux density calculations for ammonia using averaging over 30-minute time period indicated that the loss of NH_3 from HP was consistently greater than that observed for LP (Figure 15). The normal precautionary comments concerning extrapolation of flux to head have to be noted insofar that the estimate assumes that manure and urine is voided equally across the whole feedlot area and that animals utilised the whole area. These losses are comparable to those noted by Chen and co-workers in B.CCH 1020 and FLT.0331.



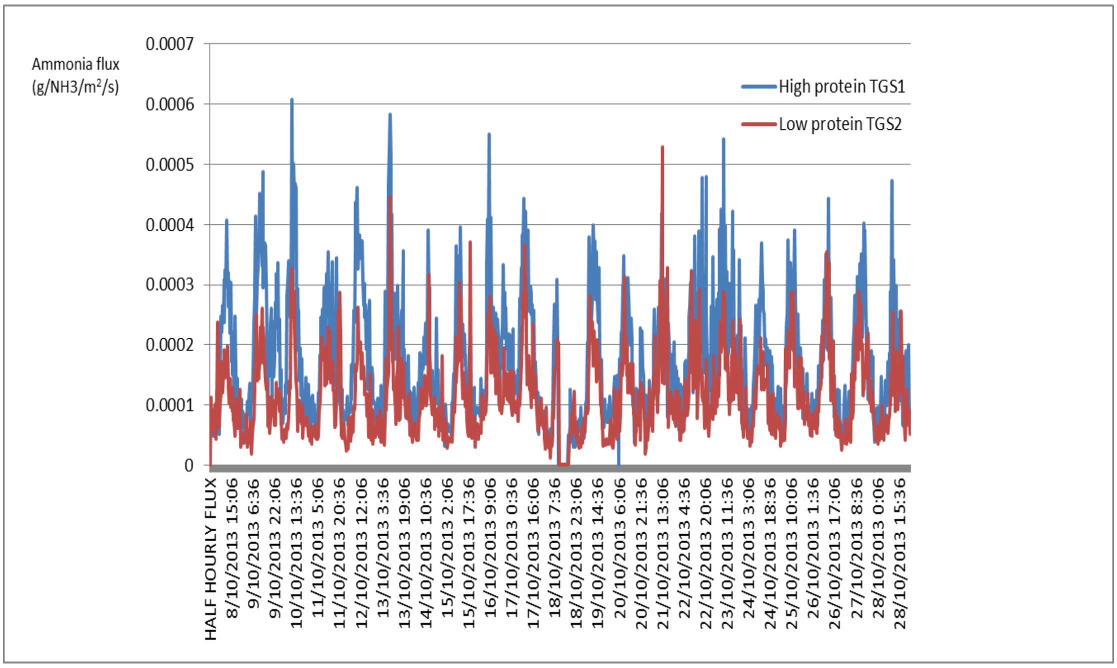


Figure 15 Integrated horizontal flux data calculated for ammonia from LP and HP feedlots.

The average fluxes for the LP treatment for CH_4, N_2O-N and NH_3N were 153.8, 8.9 and 158.1 g/capita d respectively. The HP treatment fluxes of CH_4, N_2O-N and NH_3N were 178.7, 16.5 and 186.1 g/capita.day (Table 4). In terms of ammonia flux, the measured fluxes for LP and HP were 3.2 and 3.1 times that of estimates modelled using IPCC methods. These observations were similar t those reported by Denmead et al. (2012). Measured nitrous oxide fluxes were however 1.63 and 2.55 times that of IPCC predictions. These observations are new as previous studies (e.g. B.FLT.331) had demonstrated that measured emissions were lower than those predicted by IPCC (B.FLT.331: Southern feedlot measured data was 41% of IPCC prediction and QLD data was 71% of IPCC predictions).

N2O, NH3 and CH4 concentrations from the control-release trial were measured by the side-by-side OP-FTIR spectrometers. Wind speed and wind direction during the testing period were more or less stable. The enhanced concentration for CH4, N2O and NH3 at HP experimental site is 66.7 ppb, 44.7 ppb and 55.0 ppb, and 56.3 ppb, 39.3 ppb and 46.5 ppb at LP site, respectively (Figure 11). The discrepancy between two FTIR measurements is less than 10 ppb, which is within the detective sensitivity of the instrument.

Table 4 Synopsis of animal performance from Experiment 1

Feeding system	Notes	Low protein	High protein
Animal performance			
Entry weight (kg)		418	431
Offtake (kg)		454	468
Live weight gain (kg/d)		1.77	1.74
Feed intake (kg DM/d)		12.4	12.45
Estimated faecal output (kg DM/d)		3.98	3.94
NRC estimates of N dynamics			
Feed N intake (g/d)		301	344
Urine N excretion (g/d)		150	181
Faecal N excretion (g/d)		123	142
Greenhouse gas emissions (per capita calculation)			
Manure			
Ammonia g NH3N/d)	measured	158.1	186.1
	model (NIR, 2012)	49.0	60.4
Nitrous oxide (g N2ON/d)	measured	8.9	16.5
	model (NIR, 2012)	5.46	6.46
Methane (g/d)	model (NIR, 2012)	6.16	6.12
Animal			
Methane (g/d)	model (NIR, 2012)	280	270
Methane emission (animal + manure)	measured	153.8	178.7
N efficiency			
Gross N efficiency (animal requirement / feed N intake		0.477	0.417
N retain in product (g/kg N consumed)		0.091	0.078
Losses (% N intake)			
Ammonia		52.5	54.1
Nitrous oxide		2.96	4.80
Total CO2e (kg/d)		6.60	9.55
% reduction in CO2e by reducing CP level from 17.5% to 14.5%			30.9
% reduction in ammonia by reducing CP level from 17.5% to 14.5%			15.0

4.3.5 Modelling the impact of reducing N input through dietary manipulation

Bai et al. (2015a) reported that OP-FTIR spectrometry combined with the inverse-dispersion technique can be used to measure CH_4, N_2O and NH_3 emissions from cattle feedlots. By using a combination of approaches continuous, high temporal resolution measurements over long-term periods of time and provided accurate estimates of emissions from these systems without disrupting normal animal behaviour. Bai's study (LP) focussed on establishment of the methods to measure emissions from growing beef cattle offered a ration that contained a relatively high nitrogen content (24.3 g N/kg DM; 151 g crude protein/kg DM; intake 301 g N/day) compared to NRC (2006) recommendations. Measured emissions of ammonia (NH_3-N) and nitrous oxide (N_2O-N) were 158.1 and 8.9 g/head/day or 0.525 and 0.030 of daily N intake. In a parallel study (HP), a ration that contained excessive levels of dietary N was used to understand if there was any change in proportion of N lost from the feedlot system as ammonia + nitrous oxide compared to total N lost. A ration containing 27.6 g N/kg DM (173 g crude protein/kg DM) was offered to growing beef cattle at a rate of 12.45 kg DM/day resulting in intakes of N of 344 g N/day. Measured emissions of ammonia (NH_3-N) and nitrous oxide (N_2O-N) were 186.1 and 16.5 g/head/day or 54.4% and 4.8% of daily N intake; similar proportions to N losses through gaseous emissions reported by Bai et al.

Analysis of each compartment of the modelling framework is shown in Table 5. The proportion of fertiliser N required to produce the feeds consumed in the animal's rations were lower than expected (63% to 72%) for Australian feedlot rations reflecting the relatively high intakes of lucerne and lupins in the mixes. Therefore estimates of biological N fixation were high (being 28 to 37% of total systems N). The retention of feed N in the carcass were predicted using the equations of NRC (2007) and estimated as 9.1% and 7.8% of total feed N intake reflecting growth rates of the animals of 1.77 and 1.64 kg/day. Faecal and urine N outputs were calculated as mass balances of N intake and N requirements of the growing cattle. These outputs were used to calculate the direct emissions losses of N (as NH_3N + N_2O-N) from the feed pad and were consistent across the two studies (61.2 and 62.7% for S1 and S2 respectively). These emissions were 54.4 and 58.9% of feed N intake respectively. Estimates of stockpile N losses (using the nitrogen: phosphorus ratio in feed and manure) ranged from 15 to 19% of manure N across studies. Approximately 13.7 and 10.3 of estimated total systems N in S1 and S2 was retained in the system and recycled to agricultural land. This suggests the total systems N harvested (carcass N offtake + manure N recycled) for S1 and S2 were 19.5 and 15.2% respectively.

Table 5 Modelling the impact of reducing crude protein content in rations of growing beef cattle.

Model	Input/output	Study (gN/capita.day)		Model outputs	
		LP	HP	LP Proportion of N input to different compartments of model	HP Proportion of N input to different compartments of model
Systems	Fertiliser N input	295.7	404.6	0.63	0.72
	Biological N fixation	176.0	154.2	0.37	0.28
	Total	***471.7***	***558.8***	***1.00[a]***	***1.00[a]***
Feeding	Feed intake	301.0	343.9	0.638[b]	0.615[b]
	Net carcass N	27.5	26.8	0.091[c]	0.078[c]
Manure	Faecal N	123.0	142.0	0.409[d]	0.413[d]
	Urine N	150.0	181.0	0.591[d]	0.587[d]
Losses	Aerial (NH_3N +N_2O-N)	167.0	202.6	0.612[e]	0.627[e]
	N retained in stockpile (incl. NO_3^-) after aerial loss	106.0	120.4	0.388[e]	0.373[e]
	Stockpile N losses	41.5	62.8	0.152[f]	0.194[f]
Manure resource	N applied to field	64.5	57.6	0.236[g]	0.178[g]

[a] fertiliser or biological N fixation as a proportion of total systems N
[b] feed intake N represented as proportion of systems N requirement
[c] net carcass N represented as the gross carcass N – growth N turnover as a proportion of measured feed N intake (NRC, 2007; Garrett, 1987)
[d] faecal and urine N outputs calculated using NRC (2007) as a proportion of measured feed N intake
[e] measured emissions of ammonia (NH_3-N) and nitrous oxide (N_2O-N) from feed pad surface as a proportion of manure N
[f] estimate of losses of N from stockpiled manure using the N:P ratio method as a proportion of manure N
[g] manure N potentially available for soil amendment as proportion of manure N

4.3.6 Crop trial

A cropping experiment was conducted at the University of Melbourne, Dookie Campus (36.37°S, 1450.7°E), 220 Km North of Melbourne, Victoria, Australia from 6th December 2013 to April 21st of 2014. The crop grown was forage sorghum. The topography of the experimental site was a flat underlain with a loamy clay (Nalinga loam) and well-drained soil. The mean monthly maximum and minimum temperature varied in the range of 21.9-33.5 & 9.9-15.5ºC. The average annual rainfall of the area was 550 mm. During the experiment 148.6 mm rainfall fell (of which 84 mm was in April 2014). The crop was maintained through irrigation. The field site was previously under annual rye grass and grazed by sheep. The design of the experiment considered both Experiments 1 & 2. Experiment 1 treatments are reported:

2 high protein ration fed cattle freshly collected and stockpiled (98t/ha)
3 high protein ration fed cattle aged manure (126t/ha)
4 low protein ration fed cattle aged manure (126t/ha)
5 urea fertiliser (228 kg/ha – 105 kg N/ha)
6 half rate of standard urea only (114 kg/ha- 52.5 kg N/ha)
11 Control – no fertiliser

4.3.6.1 Initial soil sampling

The soil characteristics before manure application are typical of loamy clays in northern Victoria (Table 6). The soils have good drainage, relatively low organic carbon content and are slightly acidic in nature. The cation exchange capacity of the soil is low with low base saturation. The soils are not particularly fertile reflecting the low to moderate concentrations of nitrate, phosphate and potassium: a comment that should be considered in light of the previous management of the site (pasture and grazing).

Table 6 Soil characteristics before manure application

Characteristics	Soil depth (cm)		
	0-10 (n=5)	10-30 (n=5)	30-60 (n=5)
pH(H2O)	6.0±0.4	6.3±0.3	6.9±0.6
EC (H2O) dS/m	0.08±0.02	0.05±0.01	0.06±0.02
Organic carbon (%)	1.60±0.31	0.50±0.21	ND
NH4+(KCl) mg/kg	3.3±0.9	1.8±0.5	1.0±0.0
NO3- (KCl) mg/kg	16.3±9.4	4.0±1.4	2.0±0.0
P (Colwell) mg/kg	58.7±14.4	16.5±0.69	10.7±6.8
K(Colwell) mg/kg	278.7±28.7	216.5±40.7	217.3±50.2
S mg/kg	7.52±0.81	6.42±2.72	ND
Al (cmol+/kg)	0.04±0.03	0.03±0.02	ND
ECEC(cmol+/kg)	7.62±0.87	7.62±1.22	ND
Exch. Ca meq/100g	4.72±0.51	3.91±0.71	ND
Exch. Mg meq/100g	1.64±0.32	2.53±0.52	ND
Exch. K meq/100g	0.62±0.13	0.45±0.15	ND
Exch. Na meq/100g	0.63±0.23	0.82±0.26	ND

Means (n=5) were calculated with five replicates.

4.3.6.2 Manure composition and rates of N application for the field trial

The chemical composition of the manures collected from the first experiment is shown in Table 7. The dry solids content for the freshly stockpiled HP manure, aged HP manure and aged LP manure were 67, 81 and 82% respectively. The nutrient composition of the harvested, stockpiled manures was typical of beef feedlot systems. The C:N ratio varied from 13.4: 1 (aged LP) to 16.0:1 (fresh HP manure). The manure N loadings for treatments 2, 3 and 4 were 117.5, 169.4 & 113.3 kg N/ha respectively. The total soil N reserves (top 10 cm) are estimated as 1300 kg N/ha.

The total nitrogen use efficiency ((N harvested in crop – control N yield) / (Total N reserve in soil + added fertiliser or manure N) was estimated for each treatment (Table 8). The NUE of sorghum grown under the various manure treatments was higher than the urea fertiliser treatments only. This reflects the more sustained release of N from the organic substrate. Interestingly, even though the total N addition of HP aged was higher than that of HP fresh (+52 kg N/ha), the N use efficiency of the crop was lower even though the N yield (kg N/ha) was comparable (366 vs. 362 kg N/ha). The lack of difference between the two treatments can be ascribed to the higher N content of dry matter from the second harvest (1.81% vs. 2.20% for HP fresh and HP aged respectively). Furthermore, even though the N inputs of HP fresh and LP aged were comparable, there was a substantial difference in NUE between the two crops. The main reason for the differences were the lower N content of the first and second cuts in LP aged compared to HP fresh and the lower yield in harvest 2 in LP aged. This suggests that N supply was limiting after the first cut in LP aged compared to that of HP fresh.

Table 7 Manure composition (reported on a dry solids basis)

		Total Nitrogen	Boron	Calcium	Carbon	Copper	Iron	Magnesium	Manganese	Phosphorus	Potassium	Sodium	Sulphur	Zinc
		%	mg/Kg	%	%	mg/Kg	mg/Kg	%	mg/Kg	%	%	%	%	mg/Kg
HP fresh manure	Mean	1.79	17.37	0.44	28.71	15.57	9379.33	0.39	150.85	0.52	2.08	0.42	0.27	48.33
	SD	0.06	1.13	0.02	3.91	0.48	947.30	0.03	3.84	0.04	0.13	0.03	0.03	2.10
HP aged manure	Mean	1.66	15.46	0.40	25.49	17.13	12666.00	0.34	140.61	0.45	1.82	0.37	0.23	44.89
	SD	0.02	1.31	0.09	2.62	2.62	1854.09	0.08	29.80	0.16	0.25	0.04	0.03	8.99
LP aged manure	Mean	1.11	18.99	0.54	14.85	23.68	14858.67	0.35	245.09	0.34	1.27	0.27	0.20	66.25
	SD	0.16	8.46	0.23	2.20	7.60	1960.63	0.06	82.63	0.12	0.43	0.08	0.01	34.61

Table 8 Sorghum yield (t DM/ha), total N content (%) and N yield (kg N/ha)

	Harvest 1 yield (t DM/ha)	Harvest 2 yield (t DM/ha)	Total Yield (t DM/ha)	Harvest 1 fraction yield (t DM/ha)		Harvest 2 fraction yield (t DM/ha)		Total N content (%)		Total N yield (kg N/ha)		
				Leaf	Stem	Leaf	Stem	Harvest 1	Harvest 2	Harvest 1	Harvest 2	Total
HP fresh	6.8	12.4	19.2	3.8	3.0	4.2	8.2	2.08	1.81	141.3	224.4	365.8
HP aged	6.1	10.4	16.5	3.2	2.9	4.4	6.1	2.17	2.20	132.5	228.8	361.3
LP aged	6.4	9.9	16.3	3.4	3.0	3.5	6.4	1.89	1.35	121.2	133.7	254.8
Urea 105 kg N	4.9	6.6	11.5	2.4	2.5	2.6	4.0	2.49	1.33	122.1	87.8	209.9
Urea 52.5 kg N	4.7	6.3	11	2.7	2.0	2.3	4.1	2.14	1.43	100.5	90.1	190.6
Control	2.4	3.6	6	1.5	0.9	1.6	2.1	1.38	1.44	33.0	51.8	84.9

4.4 Effect of using lignite amendments to the feed pad surface to reduce direct and indirect greenhouse gas emissions (Experiment 2)

In the first experiment conducted in B.FLT.0148, a 20% reduction in crude protein content (a comparison of rations containing 17.5% or 14.5% CP) of the ration led to an estimated 35% reduction in ammonia emissions from the feed pad surface. The estimated total reduction in greenhouse gas emissions (excluding indirect GHG such as ammonia) was approximately 6%. These data clearly demonstrated that any method to reduce the overall losses of N (in the form of ammonia, N based GHG or manure) would improve the nitrogen use efficiency in intensive beef (feedlot) production systems as well as reducing costs associated with purchased-in N to the farm. Kagimbo et al. (2013) recently demonstrated that N losses from manure co-amended with lignite reduced ammonia losses by 72% compared to non-amended controls. The application of these technologies is not evident in the intensive animal industries reflecting the current practice of stockpiling of manure in the pen and outside prior to field application of manure with little or no pen amendment/mitigation strategy.

The two outputs proposed for the study were:

1. Actual measurement of N based greenhouse gas emissions (including the indirect GHG ammonia) from manure voided from beef cattle offered two levels of crude protein.

2. Biophysical modelling of feed N to N voided to quantify the theoretical N based GHG emissions and identify if there are any deviations from current model approaches and actual measurements.

4.4.1 Feeds, feed intake and live weight change

The design of the experiment was to offer rations containing iso energetic and iso nitrogenous concentrations at a fixed level of feeding. This was achieved by restricting feed intake to 11.7 kg DM/day for each group. The experimental factor was pen treatment with lignite rather than a diet manipulation. Furthermore, the ration offered was the HP ration in Experiment 1. This ration contains surplus rumen degradable protein and therefore an opportunity to increase both urinary and faecal N outputs.

The level of voluntary feed intake and animal performance (control vs. lignite: 1.05 vs. 1.28 kg/d: $P>0.05$) was expected and similar to that predicted by NRC (2007). Nitrogen intake (crude protein intake / 6.25) was high (358 g N/day) reflecting NRC recommendations (ca. 230 g N/day) for animals of similar live weight and live weight gain (Table 9). Figures 16 & 17 outline the partitioning of N in beef cattle offered the high protein ration and managed on the feed pads (control vs. lignite amended). Subtle differences in N retention and turnover of N pool from tissue to urine are noted. These small changes reflect the difference in growth rate between each group, however they are not significant.

Table 9 Animal performance and greenhouse gas (direct and indirect) emissions in Experiment 2

	No lignite	Lignite
Animal performance		
DM intake (kg/d)	11.7	11.7
N intake (g/d)	358	358
Live weight (kg)	508	514
Live weight gain (kg/d)	1.05	1.28
N Excretion (NRC, 2007)		
Faecal (g/d)	133	133
Urinary (g/d)	212	215
Inventory (NIR, 2012)		
CH_4 (enteric: g/d)	267.2	267.2
CH_4 (manure: g/d)	17.6	17.6
Total CH_4 (enteric + manure)	284.8	284.8
N_2O (g/d)	10.1	10.1
N volatilisation (NH_3+NO_x: g N/day)	96.9	98.2
Measured emissions		
CH_4 (enteric + manure)	218	264
NH_3-N	127.6	42.8
N_2O-N	24.2	7.6

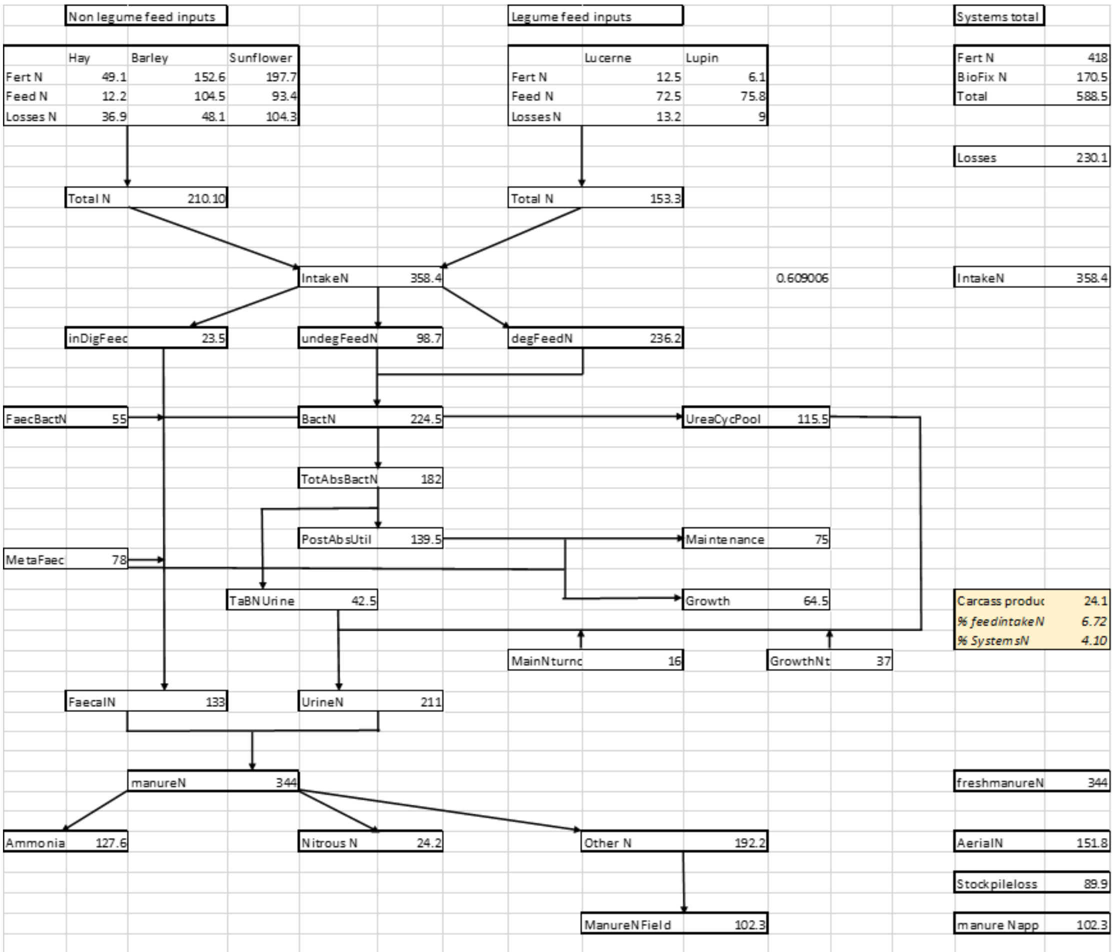


Figure 16 **N partition model of beef cattle consuming 11.7 kg DM/day of the high protein ration (358 g N/d) managed on a feed pad not amended with lignite**

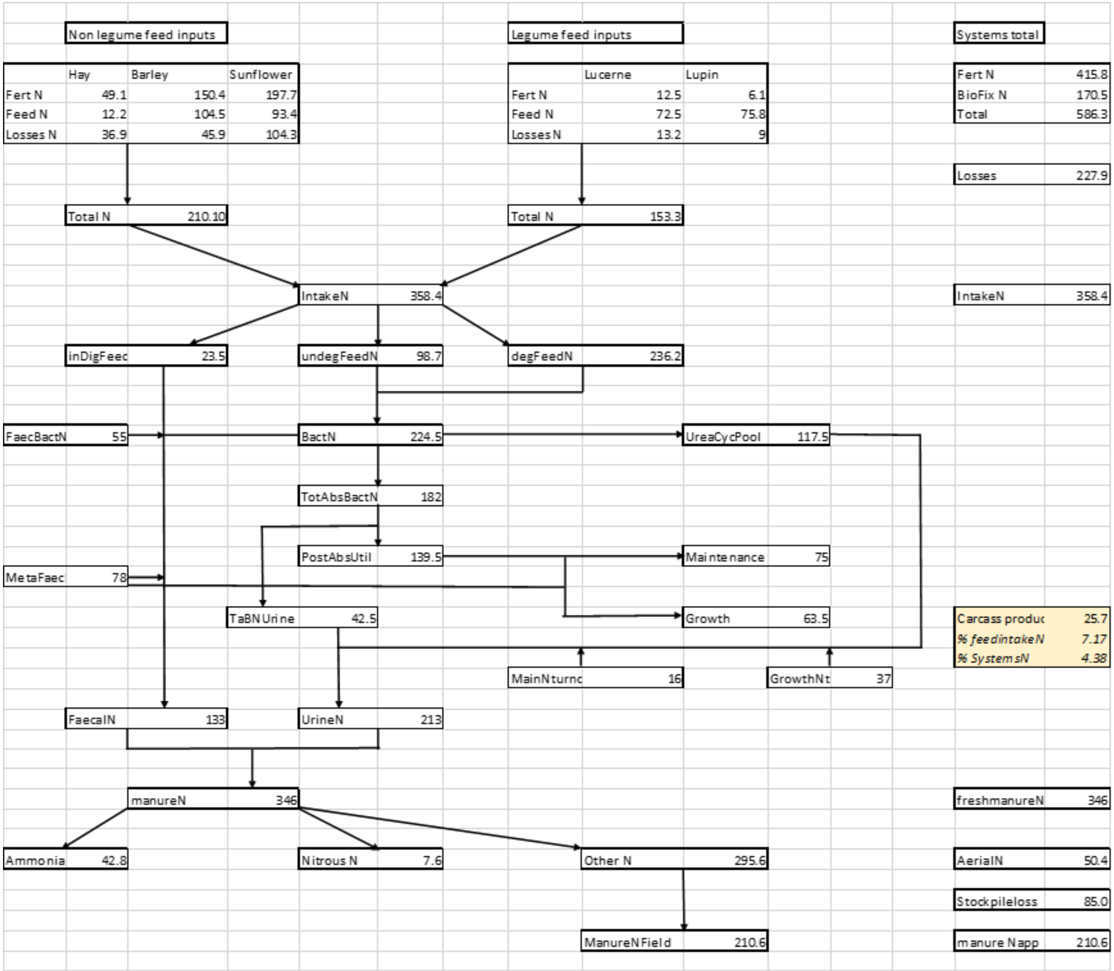


Figure 17 **N partition model of beef cattle consuming 11.7 kg DM/day of the high protein ration (358 g N/d) managed on a feed pad amended with lignite**

4.4.2 Emissions

The average NH_3 emission rates from control pen during the experimental period were 127.4 g/capita.day (Table 10). This rate is comparable with North American feedlot studies (Flesch et al., 2007; McGinn et al., 2007), and other Australian feedlot studies (Denmead et al., 2013). The addition of lignite to our treatment pen dramatically reduced NH_3 emissions over the control pen. We observed a 60% reduction, with an average emission rate of 42.8 g/capita.day.

Table 10 Previous studies on ammonia emissions from manure

	Additives to cattle manure	**N-NH_3 loss of total N (%)**
This study	Lignite	60
Other studies		
Van der Weerden et al (2014)	Sawdust	26
Chowdhury et al (2014a)	Straw	13.0
Chowdhury et al (2014b)	Barley straw	26.5
Pereira et al. (2013)	$^{+}AlCl_3$	22
	†NBPT	49
Carozzi et al (2013)	NBPT	NH_3 emission reduction by 22–47%
Taghizadeh-Toosi et al (2012)	Biochar	NH_3 emission reduction by 45%

†,N-(n-butyl) thiophosphoric triamide; ‡monensin, tylosin phosphate, growth implant, and zilpaterol hydrochloride; $^{+}$aluminum chloride.

Addition of lignite to cattle pen can retain approximately 30% of total N in cattle manure; this greatly increases the value of manure as a fertilizer. However, the potential increase of direct N_2O emissions (Bourdin et al., 2014; Pereira et al., 2013; Stackhouse-Lawson et al., 2013) or CH_4 emissions (Chowdhury et al., 2014b) could offset the reduction in NH_3 emissions. Similar to NH_3 emission calculations, we calculated N_2O and CH_4 emissions from cattle pen by given the background concentration of 311 ppb and 1716 ppb for N_2O and CH_4, respectively. The N_2O emissions were 38 g head^{-1} d^{-1} and 12 g head^{-1} d^{-1} for the control and lignite pens, respectively. Daily CH_4 emissions were affected by cattle body gain weights with 200.9 and 105.7 g CH_4/kg gain weights observed for the control and lignite treatments, respectively. The CH_4 emissions are from both the enteric and manure contributions, and in our study we cannot quantify the manure contribution. According to previous studies, cattle manure contributes 3–21% of total enteric CH_4 emissions, depending on the water content, diet and storage time (Chianese et al., 2009; Hindrichsen et al., 2005; Loyon et al., 2008; Miller and Berry, 2005).

Addition of lignite to feedlot cattle pen reduced NH_3 emissions by 60% (without greatly impacting N_2O and CH_4 emissions). The use of lignite could thus prove an effective mitigation tool for reducing the environmental impact of feedlots. The application of lignite to pens would also improve the value of the feedlot manure as a fertilizer, by both increasing the N-content of the manure and promoting crop growth via the addition of humic substances (Rose et al., 2014).

4.4.3 Modelling greenhouse gas emissions - Impact of addition of lignite to pen surfaces as a mitigation strategy

There are relatively few strategies to mitigate N emissions from feedlot systems. The current approaches focus on dietary manipulation, use of feed additives, ionophores or growth promotors to increase rate of N retained in the carcass or manure stockpile management (for example manure harvesting, dust control). There has been little focus on approaches that treat manure in the feed pad. Lignite possesses several promising characteristics that make it a suitable amendment for manure and potentially a good method of reduction ammonia and nitrous oxide emissions. Lignite is characterised as having low pH, high humic acid content (Dong et al. 2009), high CEC and adsorption capacity and high labile carbon content (Uzinger et al. 2014). Bai et al. (2015b) conducted a study that investigated the use of lignite as a pen amendment to reduce direct N emissions from feed pads thereby retaining more N in manure that could be recycled to agricultural land. Briefly, emissions of ammonia and nitrous oxide from were measured from feed pads where beef cattle were managed and consuming 318 g N/head/day from a ration containing 23.6 g N/kg DM (147 g crude protein/kg DM). Measured emissions of ammonia (NH_3-N) and nitrous oxide (N_2O-N) for the control (C) and lignite (L1) treatments were respectively, 127.6 and 42.8, and 24.2 and 7.6 g/head/day respectively or a reduction of 66.5% and 68.6% in ammonia and nitrous oxide emissions respectively.

The proportion of fertiliser N required to produce the feeds consumed in the animal's rations were typical of Australian feedlot rations (74 and 73% C vs. L1 respectively) reflecting the lower intakes of legumes in the mixes offered (Table 2) in this study compared to S1 and S2. The estimates of biological N fixation were approximately 26% of total systems N). The retention of feed N in the carcass was moderate with 7.6 and 8.1% of total feed N intake being accreted in animals in C and L1respectively reflecting moderate growth rates of the animals of 1.08 and 1.31 kg/day. Faecal and urine N outputs were calculated as mass balances of N intake and N requirements of the growing cattle. These outputs were used to calculate the direct emissions losses of N (as NH_3N + N_2O-N) from the feed pad. A substantial difference in direct emissions losses (44.6% and 14.6%; C vs. L1) of manure N was observed. These emissions represented 47.6 and 15.8% of feed N intake (Table 2). Estimates of stockpile N losses (using the nitrogen: phosphorus ratio in feed and manure) ranged from 25 to 27% of manure N across studies. These differences in N losses led to substantial differences in total systems N harvested. The total systems N harvested (carcass N offtake + manure N recycled) of C and L1 were 22.4% and 41.7% respectively.

Table 11 Modelling the impact of lignite addition to pens managed for growing beef cattle

Model	Input/output	Study (g N/head/day) C	L1	Model outputs Control: Proportion of N input to different compartments of model	Lignite: Proportion of N input to different compartments of model
Systems	Fertiliser N input	418.0	415.8	0.74	0.73
	Biological N fixation	150.5	150.5	0.26	0.27
	Total	568.5	566.3	1.00[a]	1.00[a]
Feeding	Feed intake	318.9	318.9	0.561[b]	0.563[b]
	Net carcass N	24.1	25.7	0.076[c]	0.077[c]
Manure	Faecal N	127.0	132.0	0.398[d]	0.414[d]
	Urine N	213.0	214.0	0.602[d]	0.586[d]
Losses	Aerial (NH_3N +N_2O-N)	151.8	50.4	0.446[e]	0.146[e]
	N retained in stockpile (incl. NO_3^-) after aerial loss	194.2	295.6	0.561[e]	0.854[e]
	Stockpile N losses	90.9	85	0.267[f]	0.246[f]
Manure resource	N applied to field	103.3	210.6	0.304[g]	0.609[g]

[a] fertiliser or biological N fixation as a proportion of total systems N estimated using (REF)
[b] feed intake N represent ed as proportion of systems N requirement
[c] net carcass N represented as the gross carcass N – growth N turnover as a proportion of measured feed N intake (NRC, 2007; Garrett, 1987)
[d] faecal and urine N outputs calculated using NRC (2007) as a proportion of measured manure N excretion
[e] measured emissions of ammonia (NH_3-N) and nitrous oxide (N_2O-N) from feed pad surface as a proportion of manure N

[f] estimate of losses of N from stockpiled manure using the N:P ratio method as a proportion of manure N

[g] manure N potentially available for soil amendment as proportion of manure N

4.4.4 Effect of lignite amended manure on crop growth

A cropping experiment was conducted at the University of Melbourne, Dookie Campus (36.37°S, 1450.7°E), 220 Km North of Melbourne, Victoria, Australia from 6th December 2013 to April 21st of 2014. The crop grown was a forage sorghum. The topography of the experimental site was a flat underlain with a loamy clay (Nalinga loam) and well-drained soil. The mean monthly maximum and minimum temperature varied in the range of 21.9-33.5 & 9.9-15.5ºC. The average annual rainfall of the area was 550 mm. During the experiment 148.6 mm rainfall fell (of which 84 mm was in April 2014). The crop was maintained through irrigation. The field site was previously under annual rye grass and grazed by sheep. The design of the experiment considered both Experiments 1 & 2. Experiment 2 treatments are reported:

1. lignite treated cattle manure:98t/ha: 186.5 kg N/ha),
5. urea fertiliser (228 kg/ha – 105 kg N/ha)
6. half rate of standard urea only (114 kg/ha- 52.5 kg N/ha)
7. lignite: urea treatment 34t/ha lignite + 114kg/ha urea (52.5 kg N/ha)
8. lignite: urea treatment: 34 t/ha + 228t/ha urea (105 kg N/ha)
9. lignite: urea treatment : 98 t/ha + 228kg/ha urea (105 kg N/ha)
10. lignite only (98t/ha)
11. control – no amendments

The soil characteristics for this experiment are reported in Experiment 1 (Section 4. 3.6.1).

Manure composition and rates of N application for the field trial

The chemical composition of the manures collected from the first experiment is shown in Table 12. The dry solids content for the freshly stockpiled lignite amended manure was 52% dry solids. The dry solids content of the lignite used was 76%. The nutrient composition of the harvested, lignite amended manure and lignite per se were 2.31 and 0.43% respectively. The manure N loadings for treatments 1, 7, 8, 9, & 10 were 117.7, 63.6, 116.1, 137.0 & 31.5 kg N/ha respectively. The total soil N reserves (top 10 cm) are estimated as 1300 kg N/ha.

Table 12 Manure composition (reported on a dry solids basis)

		Total Nitrogen	Boron	Calcium	Carbon	Copper	Iron	Magnesium	Manganese	Phosphorus	Potassium	Sodium	Sulfur	Zinc
		%	mg/Kg	%	%	mg/Kg	mg/Kg	%	mg/Kg	%	%	%	%	mg/Kg
		N	B	Ca	C	Cu	Fe	Mg	Mn	P	K	Na	S	Zn
Lignite amended	Mean	2.31	20.54	0.54	31.24	18.94	8263.33	0.49	191.28	0.72	2.57	0.46	0.33	62.47
	SD	0.08	2.19	0.06	4.32	1.49	1029.10	0.04	13.27	0.07	0.15	0.04	0.04	5.47
Lignite		0.43	ND	0.27	43.93	ND	ND	0.32	ND	0.04	0.03	0.39	1.35	ND

ND not determined

Table 13 Sorghum yield (t DM/ha), total N content (%) and N yield (kg N/ha)

	Harvest 1 yield t DM/ha	Harvest 2 yield t DM/ha	Harvest 1 fraction yield (t DM/ha)		Harvest 2 fraction yield (t DM/ha)		Total N content (%)		Total N yield (kg N/ha)		
			Leaf	Stem	Leaf	Stem	Harvest 1	Harvest 2	Harvest 1	Harvest 2	Total
Lignite amended	6.8	10.4	3.8	3.0	4.2	6.2	2.49	2.23	189.2	347.9	537.1
Urea 105 kg N	4.9	8.6	2.4	2.5	3.6	4.0	2.49	1.33	122.1	87.8	209.9
Urea 52.5 kg N	4.7	6.3	2.7	2.0	2.3	4.1	2.14	1.43	100.5	90.1	190.6
lignite: urea 34t/ha lignite + 114kg/ha urea (52.5 kg N/ha	5	6.8	2.3	2.7	5.0	4.5	7.3	11.8	2.05	2.14	102.5
lignite: urea 34 t/ha + 228t/ha urea (105 kg N/ha)	5.2	9.5	2.4	2.8	5.2	4.5	5.0	9.5	2.20	1.58	114.4
lignite: urea 98 t/ha + 228kg/ha urea (105 kg N/ha)	5.8	10.1	2.6	3.2	5.8	4.2	5.9	10.1	2.24	1.53	129.9
lignite only (98t/ha)	3.3	3.9	1.5	1.8	3.3	1.7	2.2	3.9	1.42	1.51	46.9
Control	2.4	5.6	1.5	0.9	1.6	2.1	1.38	1.44	33.0	51.8	84.9

The total nitrogen use efficiency ((N harvested in crop – control N yield) / (Total N reserve in soil + added fertiliser or manure N) was estimated for each treatment (Table 13). The NUE of sorghum grown under the various manure treatments was higher than the urea fertiliser treatments with the exception of the lignite only treatment. This reflects the more sustained release of N from the lignite-urea treatments potentially as a consequence of binding of ammonium to cation exchange sites. The lignite amened manure had the highest UE of all treatments (including those in Experiment 1) representing the highest yield of N per ha.

4.5 Effect of using lignite amendments to the feed pad surface to reduce direct and indirect greenhouse gas emissions (Experiment 3)

Experiment 2 demonstrated a marked abatement of ammonia through the use of 4 kg lignite/m^2 as a pen amendment. A substantial difference in direct emissions losses (44.6% and 14.6%; C vs. L1) of manure N was observed or 47.6 and 15.8% of feed N intake (Table 2). These differences in N losses led to a substantial difference in total systems N harvested. The total systems N harvested (carcass N offtake + manure N recycled) of C and L1 were 22.4% and 41.7% respectively. It is not known if 4 kg lignite/m^2 represents an optimal application rate. In Experiment 3, two rates of lignite were added to the pen surface (3 and 6 kg lignite/m^2) and the ration formulated to contain 13.5% crude protein and 12 MJ/kg DM. The objectives of the work were identical to Experiment 2 insofar that abatement potential of lignite would be determined by reduction in emissions of N based gases and N retention in manure.

4.5.1 Feeds, feed intake and live weight gain

Feed intake was constrained to 10.9 kg DM/day for all experimental periods. This was to ensure that N intake was kept as iso-nitrogenous between the two periods (Period 1 assessing 3 kg lignite/m^2 and Period 2 assessing 6 kg lignite/m^2). This led to rapid growth during Period 1 (2.0 kg LW/day – mean live weight 379 kg) and a lower growth rate during Period 2 (1.1 kg LW/day – mean live weight 444 kg). Estimated metabolic balances and N excretion models are in Figures 18 to 21.

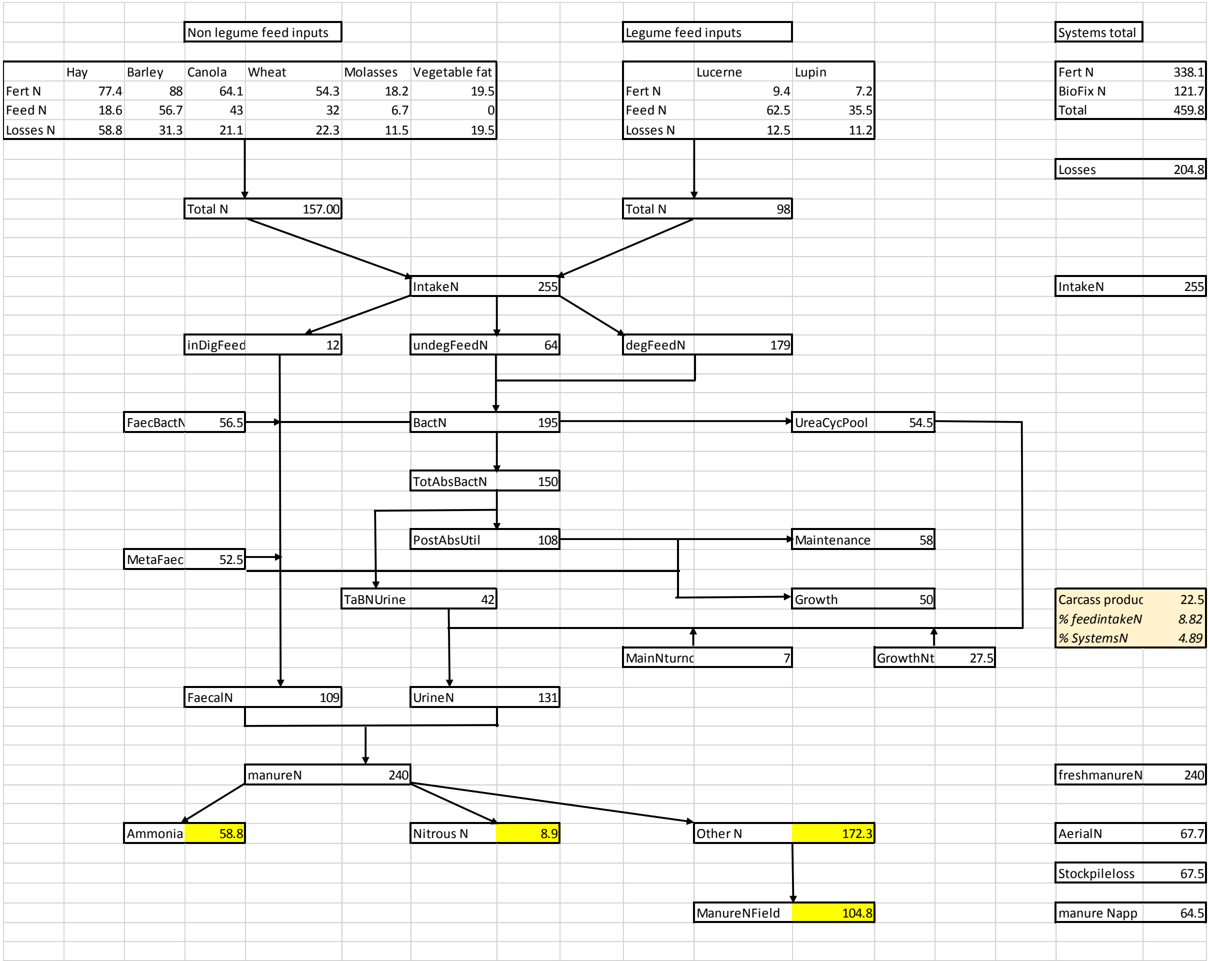


Figure 18 **N partition model of beef cattle consuming 10.9 kg DM/day of the low protein ration 255 g N/d) managed on a feed pad amended with lignite (3 kg/m^2)**

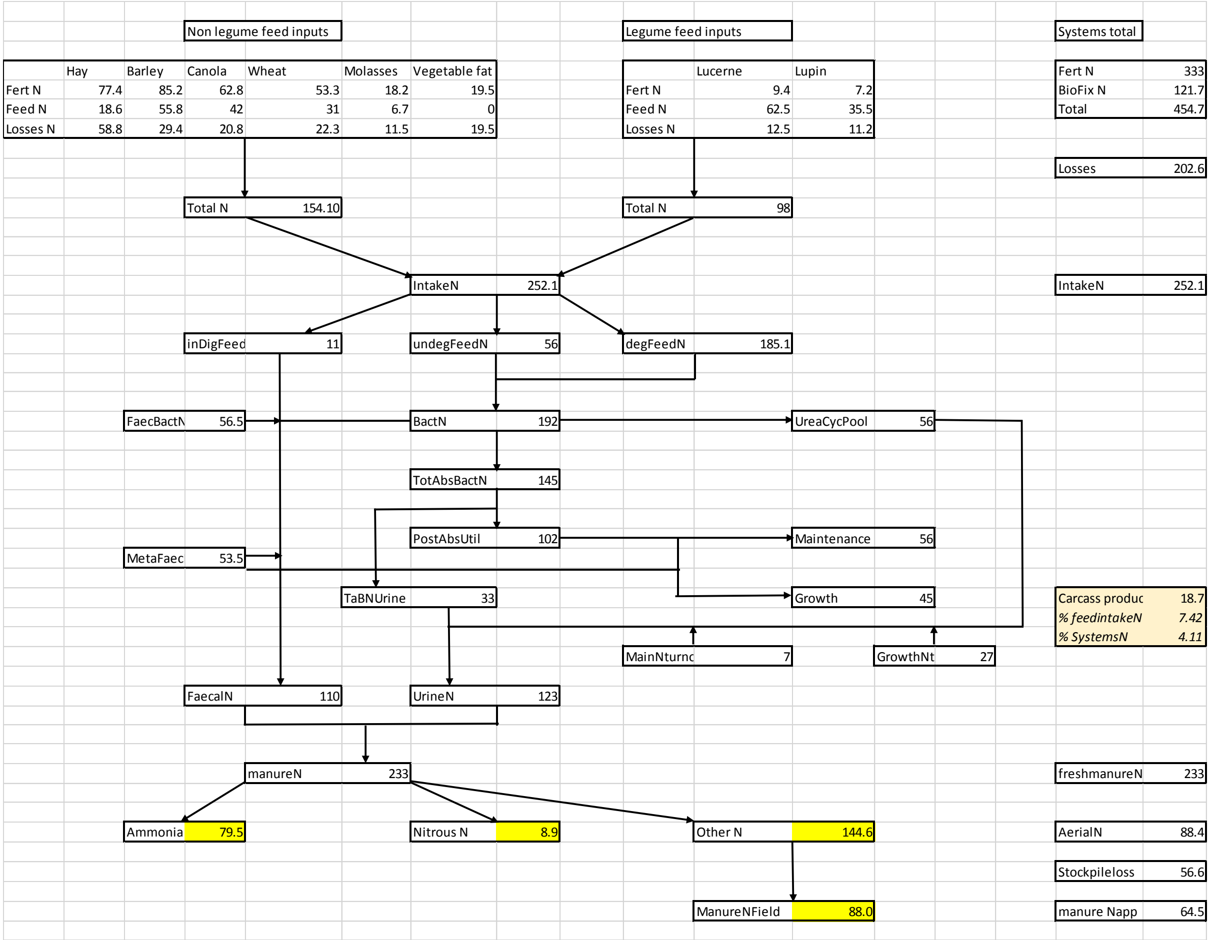


Figure 19 N partition model of beef cattle consuming 10.9 kg DM/day of the low protein ration 255 g N/d) managed on a feed pad not amended with lignite (3 kg/m^2)

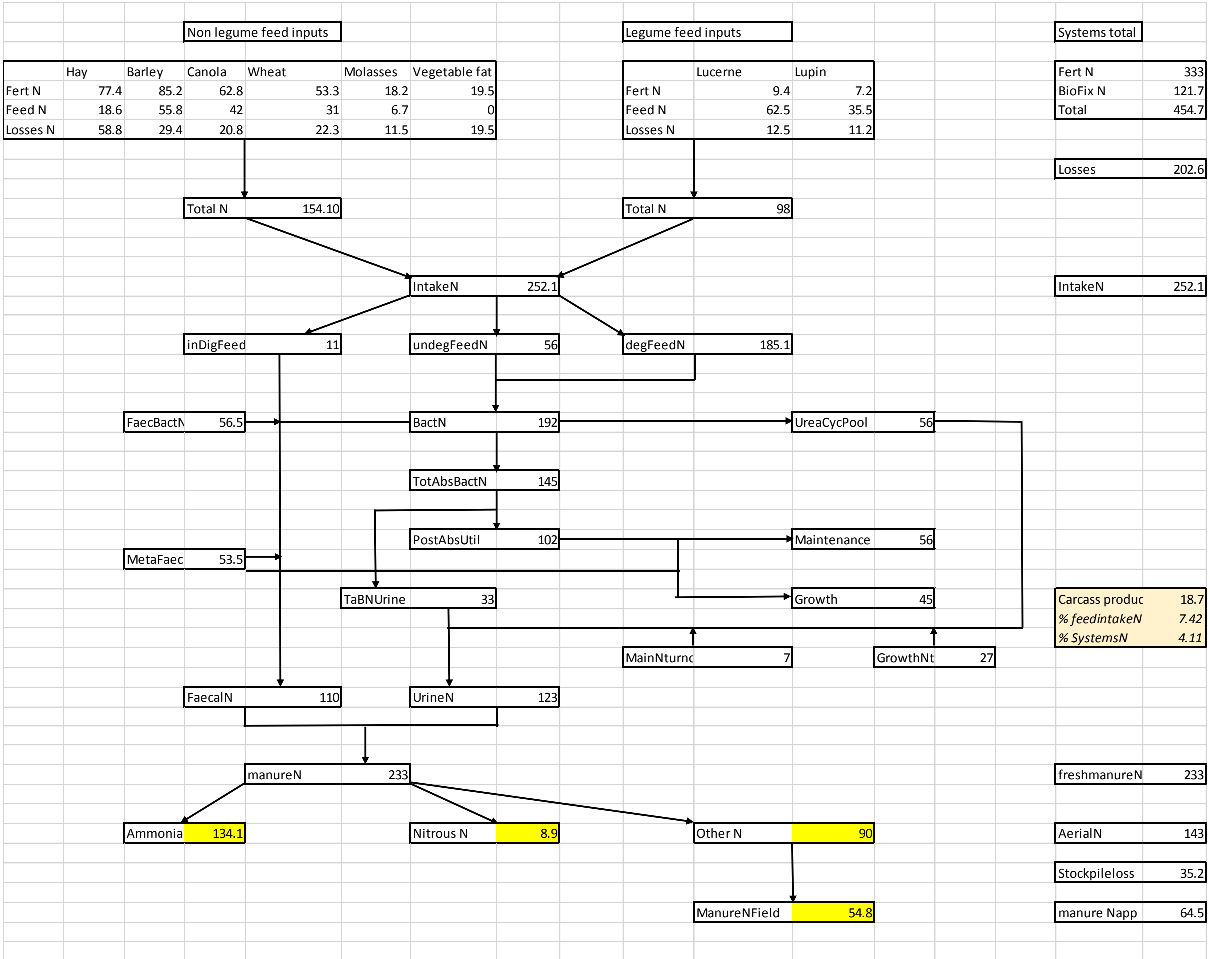


Figure 20 **N partition model of beef cattle consuming 10.9 kg DM/day of the low protein ration 255 g N/d) managed on a feed pad not amended with lignite (6 kg/m^2)**

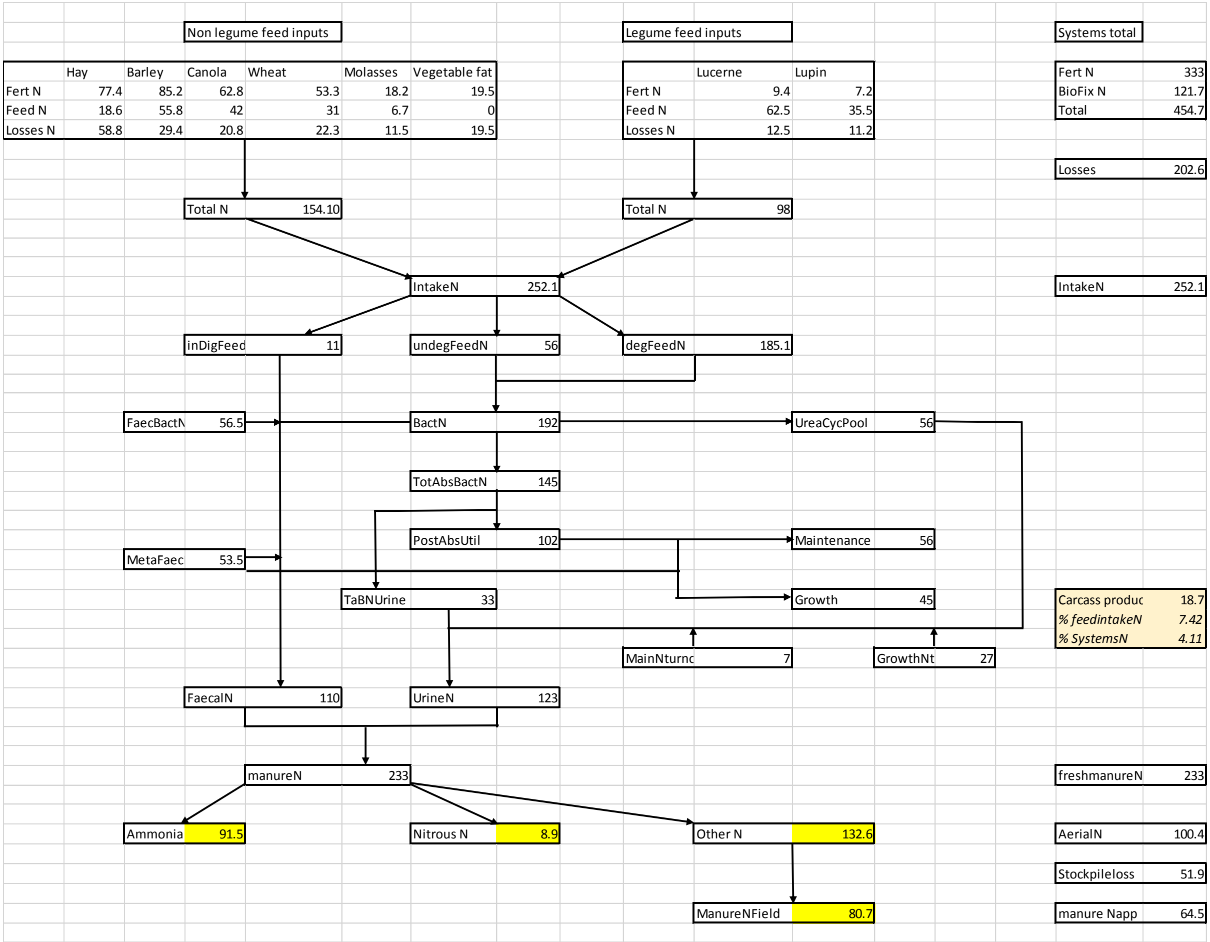


Figure 21 **N partition model of beef cattle consuming 10.9 kg DM/day of the low protein ration 255 g N/d) managed on a feed pad not amended with lignite (6 kg/m^2)**

4.5.2 Emissions

Ammonia emissions during Periods 1 & 2 were reduced by 25.0 and 31.7% respectively. These emissions measured by QCL-NH3 and estimated using IHF methods from accumulated emissions rather than mean emissions data. Lignite remains most effective in the first two to three weeks with efficiency (total accumulated NH3 reduction rate) above 50% for both application rates at Periods 1 & 2. The efficiency declined to less than 50% for both application rates however higher application rate (6kg/m2) showed a slower reduction of efficiency, compared to low application rate (3kg/m2). This indicated that the higher application rate of lignite may result in greater abatement of ammonia from the pen surface over time. Furthermore lignite applied acts as a permanent “reservoir” in terms of retaining ammonia as ammonium, an observation confirmed by the ammonia abatement curves (Figures 22 to 24) and the total N content of manure.

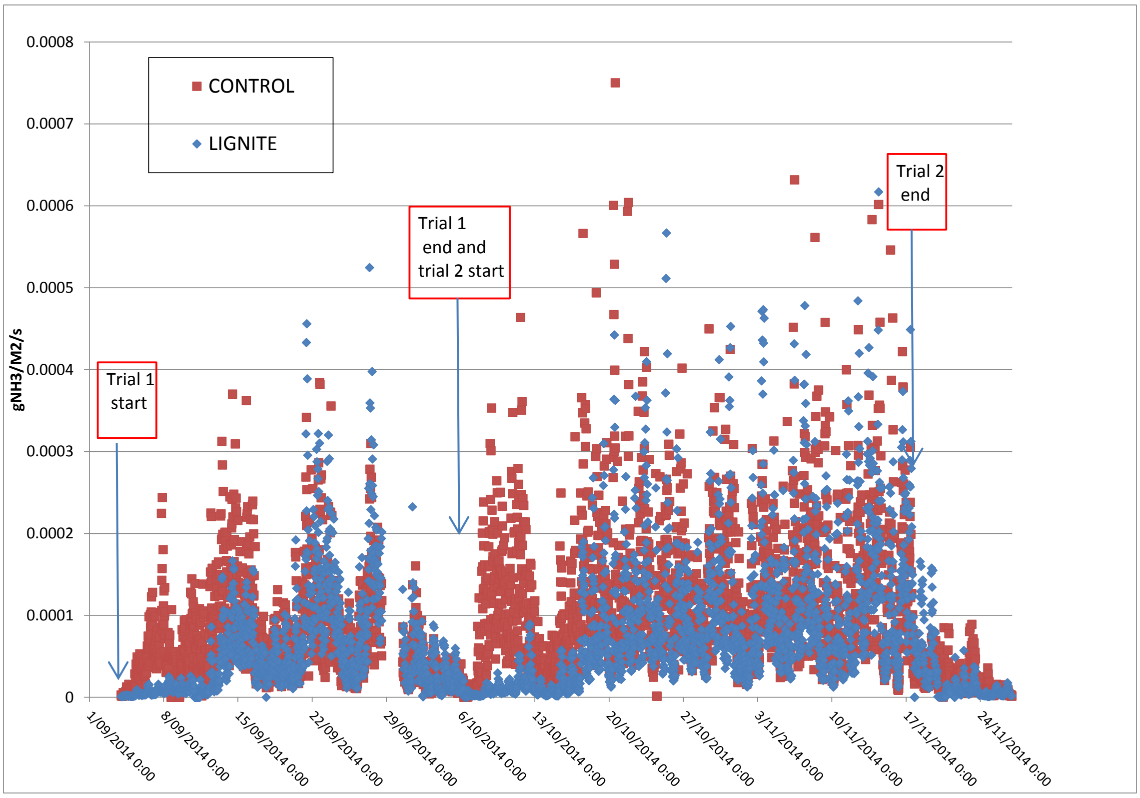


Figure 22 Instantaneous NH3 emissions measured half hourly using IHF method

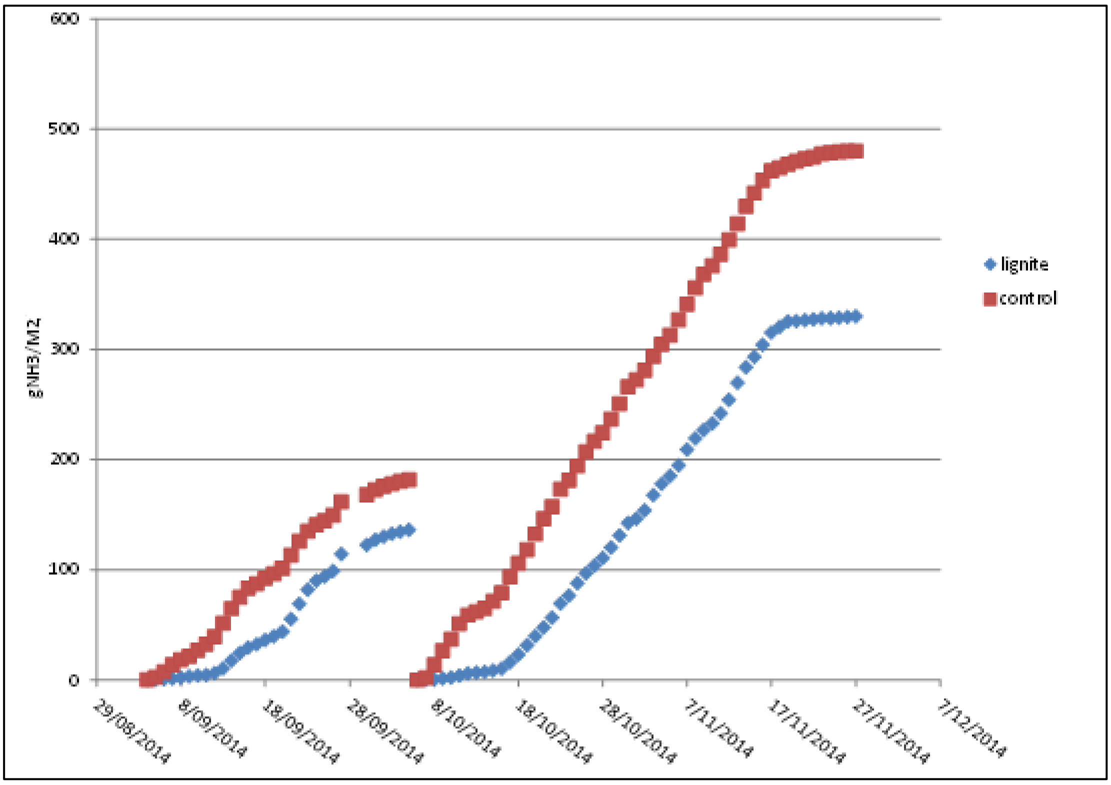


Figure 13 Accumulated NH3 emissions of each lignite trial

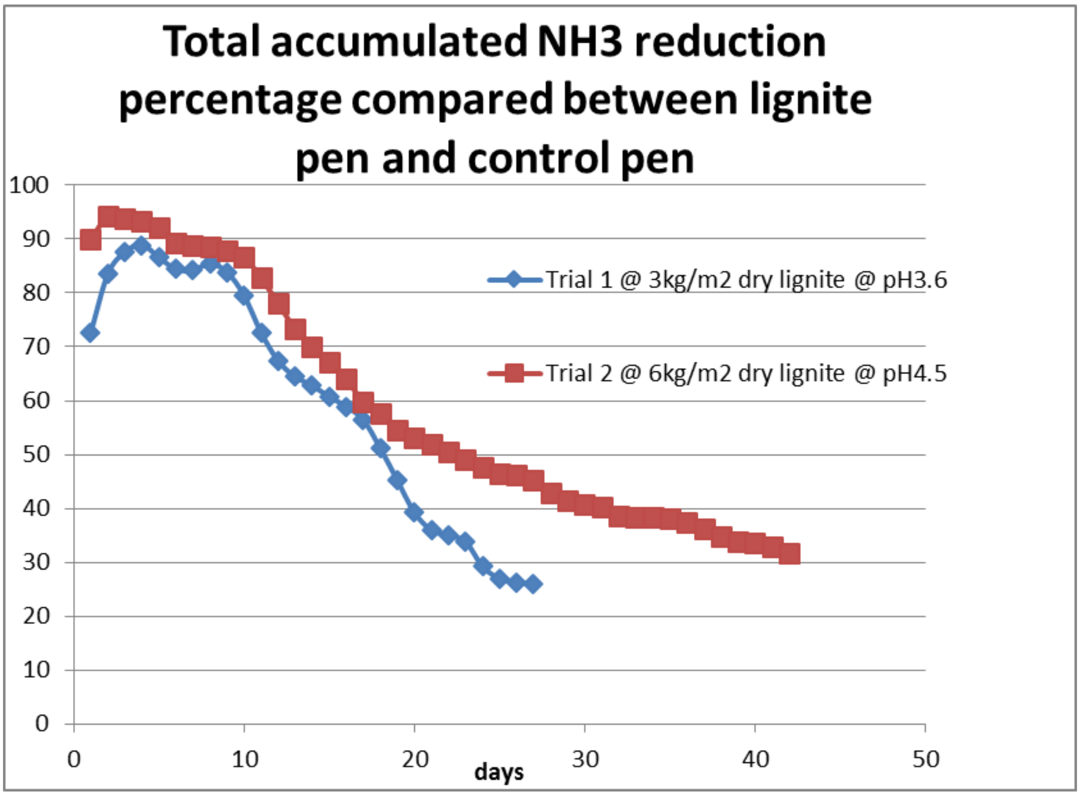


Figure 24 Lignite accumulated efficiencies of different rates at days after application

Nitrous oxide emissions from pen surfaces were monitored by a CP-FTIR and a QCL-N2O/CH4 analyser. The daily emission rates are similar to those measurements undertaken in the manure management study that used manual chambers (Section 4.5.3). The IHF calculated data for nitrous oxide were, for lignite, 20 - 40 mg N_2O-N/m^2/d, approximately twice of that from control pen which ranged from 5 - 20 mg N_2O-N/m^2/d (Figure 25)

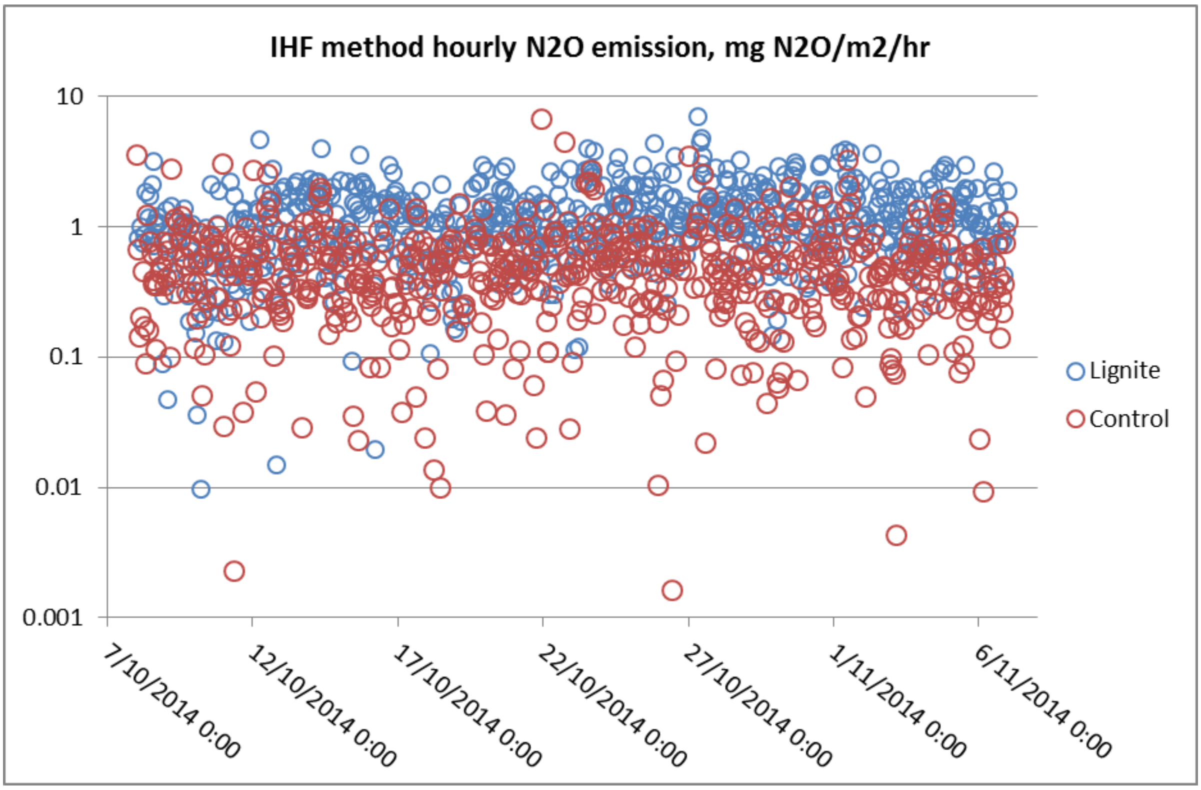


Figure 25 **Hourly N2O emissions from pen surface, calculated by IHF methods using QCL-N2O data**

4.5.3 Emissions from manure

The dry mass manure yielded at the end of Period 1 were 7577 kg and 4074 kg for lignite pen and control pen respectively. These weights include the old manure (one year stockpile) applied on both pen prior to the introduction of animals and lignite in the lignite treated pen. There was around 2300kg discrepancy between the two pens after subtracting 1200 kg lignite weight. This may reflect old manure, pen aggregate and errors in weighing and sampling for fresh and dry solids determination. The harvested lignite manure after Period 1 had 395 mg $NH4^+$ /kg dry solids (8th October 2014) and increased to 547mg $NH4^+$ /kg dry solids (13th November 2014 – mid Period 2) in comparison with control pen manure where the $NH4^+$ concentration had reduced from 261 to 148 mg/kg dry solids on the same dates respectively. Nitrate concentrations in manure stockpiles were relatively low in both the control and lignite manures with concentrations ranging from 11-42 mg/kg dry solids. Interestingly the concentration of nitrate in lignite amended manure was approximately twice that of the control, but no statistical difference could be drawn reflecting the high degree of variation. Total N content of each manure pile were approximately 1.40% dw/dw on 8th October and further reduced to 1.27% and 0.79% for lignite manure and control manure respectively (Figure 26).

The dry mass manure yielded at the end of Period 2 were 9973 kg and 5794 kg for lignite pen and control pen respectively. These weights include the residual manure (minimal) from trial 1 and lignite in the lignite treated pen. There was around 1779kg discrepancy between the two pens after subtracting 2400 kg lignite weight. The harvested manure piles were applied to a field crop trial, studying N efficiency of lignite manure and control manure.

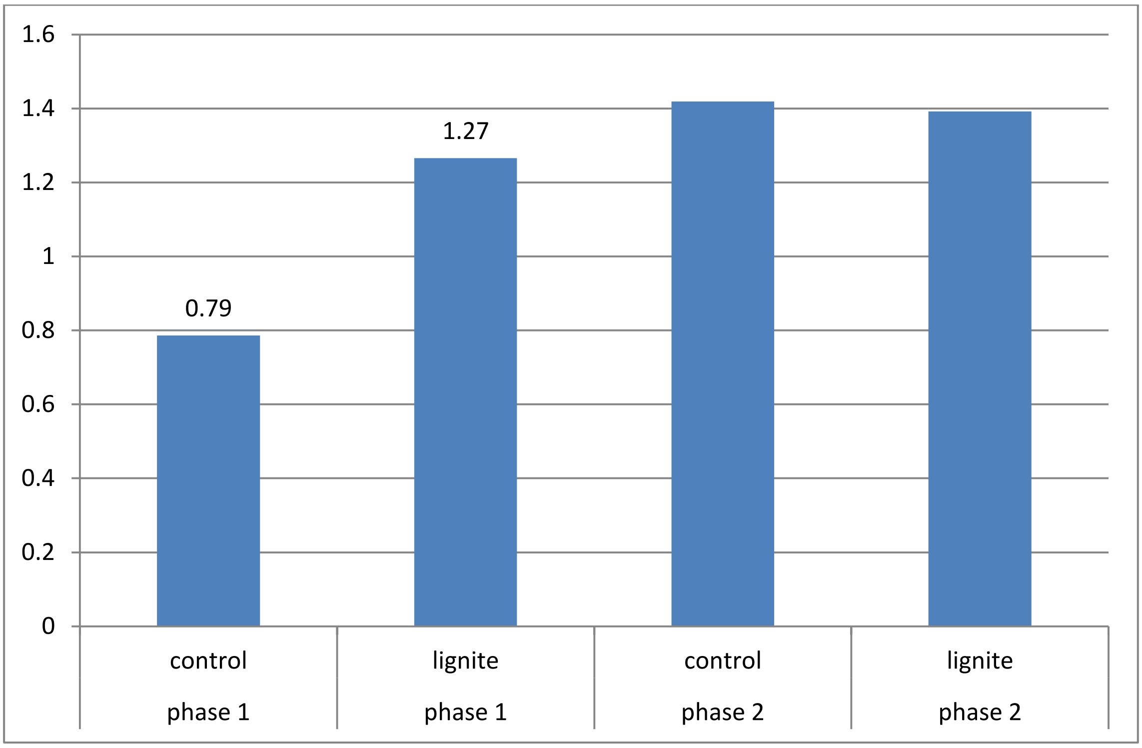


Figure 26 Total N content of manure

In contrast with Period 1 where the ammonium concentration increased during stockpiling, the ammonium concentration decreased during Period 2 in both treatments (Figure 27) reflecting losses of ammonia and a reduced abatement efficiency.

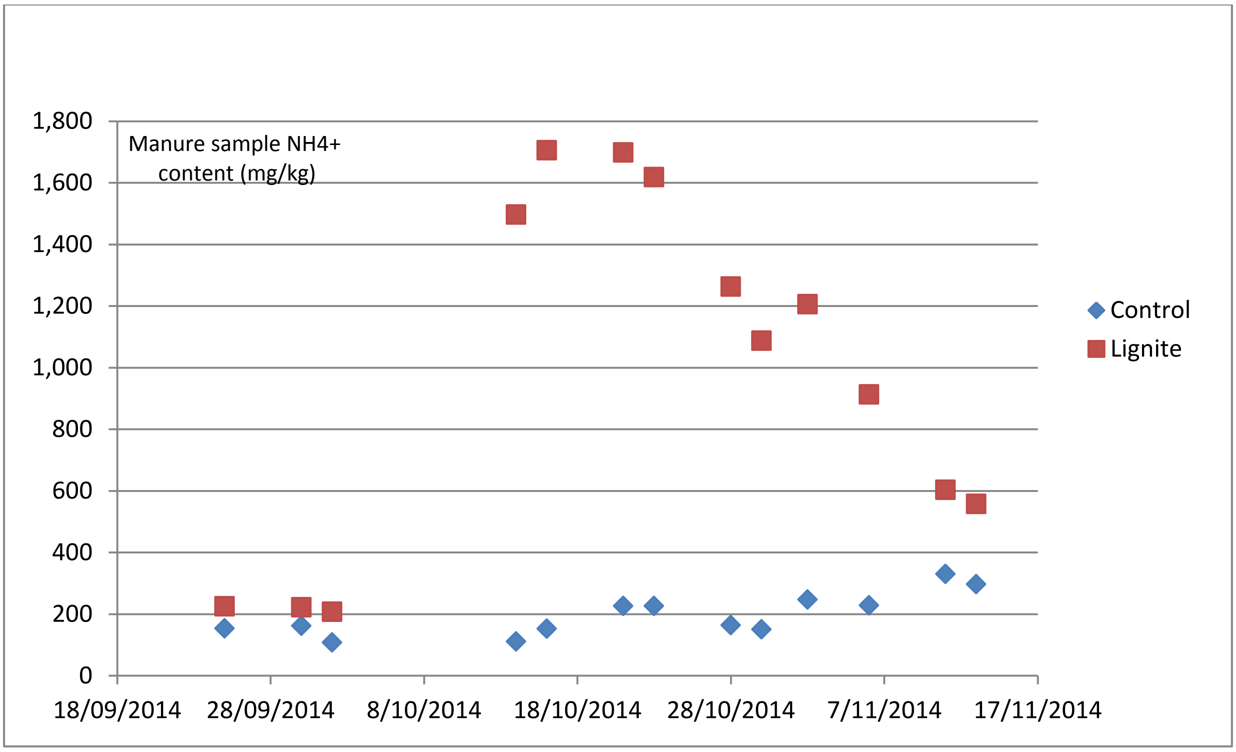


Figure 27 **Ammonium content of manure samples from pen surface**

The differences in concentrations of nitrous oxide emitted from control or lignite treated pen surfaces were insignificant during Period 2 (Figure 28).

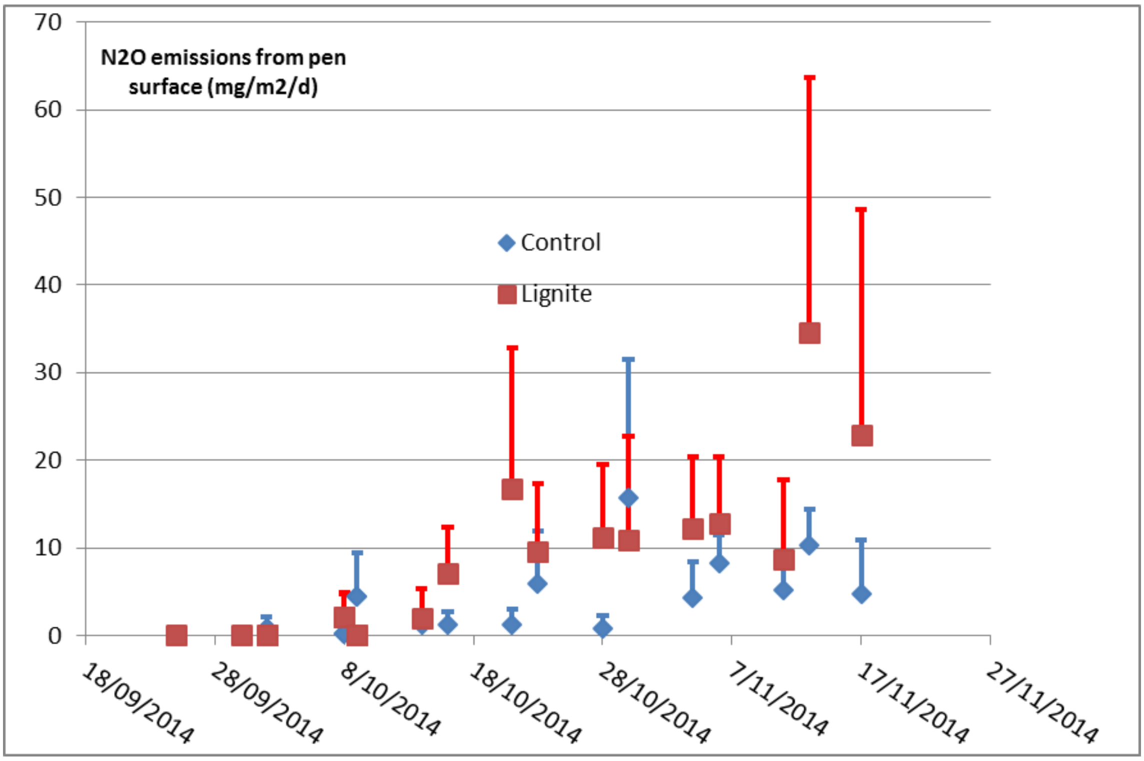


Figure 28 **N2O emissions from pens surface during Period 2**

4.5.4 Modelling of emissions

The proportion of fertiliser N required to produce the feeds consumed in the animal's rations was typical of Australian feedlot rations (approximately 73% across all treatments) again reflecting the lower intakes of legumes in the mixes offered (Table 15). The retention of feed N in the carcass was moderate with approximately 8.0% of total feed N intake being retained in the carcase. These outputs were used to calculate the direct emissions losses of N (as NH_3N + N_2O-N) from the feed pad. The N retained in manure from feed N intake were estimated as 34.5% for control in Period 1, 41.1% in lignite treated in Period 1 (or 19.1% increased retention), and in Period 2 control 21.4% with 31.5% retained in lignite (or 47.3% increase in retained N).

Table 15 Modelling the impact of lignite addition to pens managed for growing beef cattle

Model	**Input/output**	**Study (g N/head/day)**		**Study (g N/head/day)**	
		C	**L3 kg**	**C**	**L6 kg**
Systems	Fertiliser N input	333.0	333.0	333.0	333.0
	Biological N fixation	121.7	121.7	121.7	121.7
	Total	454.7	454.7	454.7	454.7
Feeding	Feed intake	255.0	255.0	255.0	255.0
	Net carcass N	27.1	27.0	25.8	25.7
Manure	Faecal N	109	109	109	1099
	Urine N	131	131	131	131
Losses	Aerial (NH_3N +N_2O-N)	81.2	61.1	136.7	94.6
	N retained in stockpile (incl. NO_3^-) after aerial loss	144.6	172.3	90.0	132.6
	Stockpile N losses	56.6	67.5	35.2	51.9
Manure resource	N applied to field	88.0	104.8	54.8	80.7

[a] fertiliser or biological N fixation as a proportion of total systems N estimated using (REF)
[b] feed intake N represent ed as proportion of systems N requirement

[c] net carcass N represented as the gross carcass N – growth N turnover as a proportion of measured feed N intake (NRC, 2007; Garrett, 1987)
[d] faecal and urine N outputs calculated using NRC (2007) as a proportion of measured manure N excretion
[e] measured emissions of ammonia (NH_3-N) and nitrous oxide (N_2O-N) from feed pad surface as a proportion of manure N

4.5.5 Effect of lignite amended manure on crop growth

A cropping experiment was conducted at the University of Melbourne, Dookie Campus (36.37°S, 1450.7°E), 220 Km North of Melbourne, Victoria, Australia from 24th December 2014 to 26th February 2015. The crop grown was forage sorghum. The topography of the experimental site was a flat underlain with a loamy clay (Nalinga loam) and well-drained soil. Manure was collected from the pen surfaces of both treatments at the end of the measurement periods (7 October 2014 and 23 December 2014). A replicated plot design (randomised block design with 6 treatments and three replicates per treatment) was installed with the aim of determining soil N transactions from a range of treatments measured under growing conditions. A forage sorghum crop was established in late December 2013 and harvested after 3, 6 and 12 weeks of growth. The crop yield was assessed only. The treatments were:

1. Control (no manure of N fertiliser)
2. Urea (98 kg N/ha)
3. Experiment 3 – Period 1 control manure (harvested 7 October 2014: 132.2kg N/ha)
4. Experiment 3 – Period 1 Lignite manure (harvested 7 October 2014:120.2 kg N/ha)
5. Experiment 3 – Period 2 control manure (harvested 23 December 2014:117.6 kg N/ha)
6. Experiment 3 – Period 2 Lignite manure (harvested 23 December 2014: 112.7 kg N/ha)

The yields of sorghum from the various treatments are in Table 16. In this trial, lignite amended manure tended to reduce the total biomass yield compared to the urea control but increased yields in relation to the control (no urea addition).

Table 16 Sorghum yield (t DM/ha) from lignite amended trial plots

Treatment	Yield (t DM/ha)
Control	5.70
Urea (98 kg N/ha)	8.43
Period 1 control manure (harvested 7 October 2014: 132.2kg N/ha)	8.27
Period 1 Lignite manure (harvested 7 October 2014:120.2 kg N/ha)	7.28
Period 2 control manure (harvested 23 December 2014:117.6 kg N/ha)	7.08
Period 2 Lignite manure (harvested 23 December 2014: 112.7 kg N/ha)	7.25

4.6 Cost effectiveness of lignite amendment of feedlot pens to reduce N losses

The Iowa State University Extension Service Facility Economic Analysis software was used to assess the impact of using lignite as a possible abatement method to reduce both direct and indirect greenhouse gas emissions. The assessment used inputs for a 13750 head finishing feedlot (approximately 120 days on feed and 2.5 cycles of production). Feed costs were estimated from the rations (discounted by 15% reflecting small batches of feed) offered in the work conducted at Dookie, Victoria in 2014 and 2015. Animal performance (live weight and gain) used was measured data with animals entering the feedlot at an average weight of 400 kg. Labour, facility finance and operational costs were estimated using average labour costs, 65% ownership with loans extending over a 15 year period at 5%. Nutrients in the manure were valued according to the pricing structures of N, P and K fertilisers (wholesale price rather than retail: $500/tonne urea; $475/tonne MAP and $335/tonne potash).

When lignite was added as a pen amendment, pen loadings per capita were calculated on the basis that the manure harvesting cycle would be 56 days (8 weeks). Lignite costings were based on Sustainable Business Australia estimates of $9.2/tonne wet with transportation costs of $39.41/tonne (500 km radius freight costs: Freight Metrics). Total cost including transport was estimated as $48.61/tonne dry delivered. Changes in nutrient density, estimated changes in nutrient availability and savings on fertiliser, impact of extra crop production feeding (20%, 25 and 30% replacement with 3 to 6 kg lignite/m^2) into the feed supply (valued as a replacement to the hay component of the ration in this simulation), and increased costs associated with labour, manure harvesting and management were considered. The outcomes of the analysis are reported as a valuation of the net income with and without manure valuations to the production system and reported as a total $ value and $ per capita value (Table 17).

Table 17 Cost effectiveness of lignite amendment of feedlot pens to reduce N losses

Lignite rate	No manure valuation		Manure valuation included	
	Actual ($)	$/capita	Actual ($)	$/capita
No lignite	4,268	0.31	200,165	14.56
3 kg/m^2	37,409	2.72	265,119	19.28
4 kg/m^2	56,658	4.12	277,478	21.08
6 kg/m^2	59,137	4.30	295,887	21.52

The analysis is sensitive to transport costs (especially diesel and freight distance) of lignite to the feedlot and fluctuations in the price of synthetic fertilisers and other inputs to the farming system. It is estimated (using average abatement of ammonia and nitrous oxide emissions from these studies) that the abatement potential of lignite is 2.04 kg CO_2e/kg lignite added to the feedlot systems (without liabilities for transport or fugitive emissions). If a CFI method was to be developed for the abatement technology the net abatement return for a feedlot system would be equivalent to approximately $104,150 in credits or $7.57 per capita (exclusive of transaction and compliance costs).

5 Discussion

Five objectives were identified for the work conducted in this project. These were:

1. Determine the chemical and biochemical processes that drive the conversion of urea and other nitrogen sources to ammonia in manure derived from beef feedlots.
2. Evaluate a range of co-amendments to manure that could mitigate emissions of ammonia or nitrous oxide thereby reducing total direct and indirect greenhouse gas emissions.
3. Evaluate the impact of recycling manures with elevated concentrations of retained N under cropping systems.
4. Determine the impact of dietary manipulation to mitigate methane emissions from cattle on subsequent manure composition and N emissions.
5. Based on the above, develop abatement methodologies that can be included on the DCCEE 'positive' list for abatement of N-based feedlot greenhouse gas emissions.

Objectives 1 to 4 were successfully completed in this project and Objective 5 is contingent on further discussions with the feedlot industry. Currently there are no CFI methods that are statutory instruments that can be used easily in feedlot systems. The only methods available are based on the supply of nitrate through feeding blocks or licks as a direct mitigant of enteric methane production or the accelerated finishing of grass-fed cattle using intensive feeding areas (specifically excludes feedlots).

5.1 Chemical and biochemical processes that drive ammonia emissions from feedlots

The sources of nitrogen loss from feedlots and manure management systems are well characterised and understood. The main losses of N are through the yield of ammonia and nitrous oxide from microbial degradation of principally urea inputs from urine and faecal deposition to the feed pad. Ammonia volatilization is highly variable and is influenced by the amount of total nitrogen (TAN), environmental temperature, wind speed, pH, chemical and microbiological processes, diffusive and convective transport in the manure, and gas phase resistance in the boundary layer above the source (Arogo et al., 2006). Little is known about the total annual emissions of ammonia from Australian beef feedlots, and few robust long-term measurements have been made. The concentration of ammonia in the atmosphere is derived overwhelmingly from agriculture (Galloway 1998; Aneja 2008; Galloway et al. 2008; Behera et al. 2013) and accounts for as much as 95% of the total emission sources of this pollutant (Isermann 1994; Webb et al. 2005). Cattle feedlots represent emission hotspots for ammonia in Australian agriculture with the consequence that high atmospheric concentrations of the gas around feedlots can lead to rapid N-exchange adjacent and downwind to the feedlot. There are approximately 965,000 cattle on feed currently (ALFA, 2015), each excreting approximately 78 kg N annually. Much of the N is excreted as urinary urea, which is rapidly converted to ammonia by urease (Voorburg and Kroodsma 1992), and subsequently lost to the atmosphere via volatilization. In previous MLA funded research (B.FLT.0331), and international studies, about 60% of the feed N is lost to the atmosphere as ammonia (Flesch et al. 2007; Denmead et al. 2013). Hence, a 15,000 to 20,000 head feedlot may emit 3 tonnes of ammonia N per day or about 2300 tonnes urea equivalent per annum. On a local scale, it is thought that a significant portion of the ammonia emissions are deposited near the site of original emission (Sutton et al. 1993; Fowler et al. 1998; Loubet et al. 2009; Denmead et al. 2013). Loubet et al. (2009) estimated that between 2 and 60% of emitted ammonia is deposited within 1 km of the source, mainly as dry deposition. Wet deposition, caused by the removal of ammonia from the atmosphere via precipitation, is also a pathway for N deposition. Asman and van Jaarsveld (1992) calculated that wet deposition starts about 500 m from the source, but is relatively insignificant until a few km downwind. This localized deposition may affect crop productivity (Yan et al. 2014) adjacent to the feedlot as well as potentially result in excessive N accumulation in soils (van Breemen and van Dijk, 1988; Sanderson et al. 2006; Sutton et al. 1993).

Another important consequence of ammonia deposition is an inevitable increase in N2O emissions from the soil. Deposition provides available N to the soil microbes that produce N2O. These “secondary, off site” emissions, generally referred to as indirect emissions, are potentially an important component of the greenhouse gas budget for sources like feedlots. Nitrous oxide (N2O) is a potent greenhouse gas and a chemical that has a major role in ozone depletion (Crutzen 1970). N2O emissions tend to be low in the feedlot environment (B.FLT.0331 & B.CCH.1020) as manure management does not generally promote anaerobic conditions. There have been many reviews on the emissions of nitrous oxides from organic manures. Nitrous oxide is largely emitted from manure and soils as a result of microbial nitrification and denitrification. Nitrification is an aerobic process that oxidises NH_4^+ to NO_3^- with N_2O as a by-product, while denitrification is an anaerobic process that reduces NO_3^- into N_2, with N_2O as an obligatory intermediate. Nitrification is often an essential prerequisite for denitrification and plays a dominant role in the conversion of N inputs from urine and urea or ammonium-based fertilisers into NO_3^-. For instance in animal production systems high loadings of nitrate in the soil (for instance under poorly consolidated feed pads) and low soil aeration (wet or compacted soils) can result in high N2O emissions (Klein et al. 2006). Nitrous oxide has received considerable attention recently and has become a major measured component of the total N budget of agricultural systems reflecting its potent global warming potential. Chen et al. has measured and reported nitrous oxide emissions from feedlot manure managed systems in a number of projects managed through MLA (for instance B.FLT.0331).

Abatement of N based gaseous emissions has focussed on a range of management techniques as well as attempting to reduce urease activity (B.CCH.1020). The majority of these studies have clearly demonstrated how difficult it is to reduce ammonia and nitrous oxide emissions in practice and the recent focus to develop new feed pad amendment strategies have been successful (this project). Mineral and chemical adsorbants have been identified as a suitable strategy to reduce ammonia emissions from feedlots. The ideal adosrbant would have a moderate to high cation exchange capacity and charge density, be acidic in nature (e.g. below pH 4.5) and contain very low levels of heavy metals or other potential toxicants. Further, the critical issue for the success of these strategies is not only the total abatement potential but the costs associated with the supply and management of the feed pad surface that has received an adsorbent. Restricted supply will lead to increased costs associated with abatement. A range of products have been used to abate ammonia from manure under various climatic and management environments including zeolite, clays, biochar and lignite. Zeolite and clays can be discounted as they are expensive and supply is reasonably restricted in Australia (limited number of companies that can provide the volumes required at a low cost).

Biochar is a carbon rich by product that is produced from pyrolysis of organic matter under oxygen limited conditions (Kookana, Sarmah et al. 2011). There have been a number of studies that have demonstrated no, minor and substantial benefits of addition of biochar to soils (for instance Clough et al. 2010; Jeffery et al. 2011; Afeng et al. 2012; Wang et al. 2012). There is however. The review of literature on biochar and other organic amendments clearly demonstrate considerable conjecture over the mechanisms involved with abatement of N based gases from manures as a result of biochar addition. It is not known how much ammonia or nitrous oxide can be abated using the product, or whether the limited supply of a consistent product (of known provenance) would have an impact on the usage of the product under commercial conditions. An incubation study using biochar and lignite was conducted in this project to further understand the abatement potential of this product. Lignite (or brown coal) is a low rank, high moisture coal that is mainly used to produce electricity (Ciuk and Piwocki, 1990: www.dpi.vic.gov.au/earth-resources; Geoscience Australia, 2012). Lignite has been used as a soil amendment for many years with the aim of building soil organic matter and humic acid content (Kwiatkowska et al, 2005; Katzur et al. 2002 & 2003). Lignite possesses several promising characteristics that make it a suitable amendment for manure and potentially a good method of reduction ammonia and nitrous oxide emissions. Lignite is acidic (Vorres, 2000; Bernal et al. 2009), high humic acid content (Kalaichelvi (2006), high CEC and adsorption capacity and high labile carbon content thereby making it a potential manure amendment (Rhoades et al. 2010; Yuliana et al., 2010; Oussa, 1978 and Qi et al. 2011). There has been limited research on the use of lignite in manure management systems. Berg et al. (2009) demonstrated that ammonia emissions from pig manure could be reduced by 67% with the addition of 3% (mass/mass) of powdered brown coal. Kagimbo et al. (2013) suggested substantial retention of ammonia in the lignite was possible and provided a good science base to conduct large scale trials in managed feedlot pens to understand the abatement potential of the product.

The initial incubation studies undertaken in the project supported the opinions of Kagimbo et al (2013) insofar as they demonstrated that ammonia emitted from artificial urine was bound chemically to the cation exchange site of lignite as ammonium (reflecting the acidic conditions resulting from lignite addition). There was little or no evidence of immobilisation of ammonia N into the organic matter phase. Biochar is alkaline in pH and led to an increase in ammonia emissions compared to the control. These studies supported the view that lignite may be a suitable pen management technique to reduce N based gaseous emissions, retain ammonium in manure in the stockpile and release that N when incorporated into a crop production system.

5.2 Reducing the crude protein content of the ration

The global emissions of the two important nitrogen-based gases (ammonia and nitrous oxide) from agriculture amount to 27 to 38 Tg NH3-N.yr-1 and 4.2 to 7.1 Tg N2O-N.yr-1 respectively (Beusen et al.; Reay et al. 2012). Agricultural emissions of ammonia are predominately from two sources: synthetic N fertiliser and livestock manures. Beusen et al. estimated that 10 to 12 Tg NH3-N.yr-1 was derived from fertiliser N sources and the balance (16 – 27 Tg NH3-N.yr-1) was from livestock systems. If the ammonia emissions from livestock systems are further subdivided, it is estimated that 31 to 55% are derived from housing and storage of manure, 23 to 38% from spreading of manures to agricultural land and a further 17 to 37% from grazing animals through direct voiding of faeces and urine to pasture and rangeland. The estimates of emissions of nitrous oxide from agricultural systems are predominately (>90%) from agricultural soils receiving synthetic N fertilisers or livestock manures with the balance being derived from burning of biomass, for instance stubbles and crop wastes, and savannah. One criticism of the current information on global emissions is the lack of segmentation into ruminants, monogastric and avians, leading to considerable uncertainties in estimates of the impacts of individual sectors on global or regional emissions.

The important drivers of emissions of nitrogen-based gases from agricultural systems are the quantity of reactive N entering the agricultural system, losses, including excretion, of reactive N from the agricultural system, the inter-conversion of one species of reactive N to another – for instance the conversion of ammonia to nitrous oxide after atmospheric redistribution of the former, and the type and extent of agricultural systems. By far the most important drivers of emissions are the inputs and efficiencies of use of those inputs by the farming system. It is general acknowledged that ruminant (beef cattle) production systems are inefficient converters of N from feed to N in edible product (estimates range from 7 to 10% of N fed is retained in product consumed; NRC, 2007) and therefore gains in efficiency are essential to reduce the overall losses as well as the potential impacts of those feed and rearing systems on the environment. However, attempts to manipulate N use efficiency in growing beef cattle have demonstrated little scope to improve N capture in products and therefore management of N inputs and waste are the key focus of improving the overall systems efficiency.

Arguably, the approach of managing beef cattle through feedlot systems, an example of a concentrated animal feeding operation, has a medium to long-term positive impacts on the environmental management of input (diet) energy and nitrogen. Feedlot systems are designed to increase growth rates of animals through the delivery of diets that contain the optimal levels of energy, protein and minerals to maximize the growth rate of the animal (Gleghorn et al. 2004; NRC 2007). In turn, this approach leads to an improved life cycle efficiency of energy and nitrogen utilization per product produced reflecting an effective reduction in maintenance requirements of the animal during production (Capper and Cady, 2012; Capper, 2011). For example, the recent studies by Capper and Cady comparing feedlot and forage-based finishing systems demonstrated 2.5, 2.8 and 12 fold increased energy, methane and land area required to produce 1 kg of beef in the forage-based system compared to feedlot produced beef.

On of the major issues facing the Australian feedlot industry into the future is the issue of rations supplying N contents greater than those required by the animal to maintain high levels of productivity. This scenario is driven by the use of high energy supplements that also contain high concentrations of crude protein (for example cottonseed meal). An analysis of 43 rations feed to beef cattle managed under lot feeding systems in Australia identified a 'baseline' CP content of a ration to be 13.92% or 22.3 g N/kg DM (range min-max = 11.5 to 15.09%: B.CCH.1086). The 95% CI for CP content for this sample of rations was 13.49 - 14.35%. Theoretically the minimum CP content of a ration to support the lower 95% CI of growth is 10.2% (NRC, 2007), To maintain an acceptable growth rate (1.85 kg/d) the minimum CP content for a finishing ration that a producer is likely to offer is 11.6% (NRC, 2007). If the mass of ammonia emitted from feedlots is to be reduced, the producer has one of two options – a reduction in crude protein content and re-formulation of rations or use of technologies that abate ammonia emissions from co-mingled faeces and urine on the feedpad surface.

In Experiment 1 of this project, the impact of feeding elevated levels of crude protein in ration on ammonia and nitrous oxide emissions was examined. The direct emissions losses of N (as NH_3N + N_2O-N) from the feed pad were consistent across the two studies (61.2 and 62.7% for low and high protein feeding systems respectively). These emissions were 54.4 and 58.9% of feed N intake respectively. These losses are consistent with those reported by Todd et al. 2004). However, there was an increase in estimated manure stockpile losses (calculated from mass balances) from the high protein manure management system of more than 50%, thereby posing a significant N loss from the whole farm system as opposed to losses directly from the pen.

Todd et al. (2004) identified that daily per capita NH_3-N losses increased by 10-64% after the daily per capita N in feed rations increased by 15-26%. Further, a number of studies have suggested that more than 35 to 60% of N fed to animals could be lost through ammonia emissions from the whole farming system (mixed emissions sources from feed pads per se, lagoons, manure management and handling areas: Flesch et al. (2004) Harper et al. (2004) Cole et al. (2006) Erickson and Klopfenstein (2001) Erickson et al. (2000), Bierman et al. (1999). In our studies (low vs. high protein), direct emissions of NH_3-N from the feed pad increased by 35.6 g (21.3%) through an increase of 42.9 g N in feed intake (14.3%), a comparable estimate to North American studies. We also measured direct emissions of nitrous oxide from the feed pads and demonstrated an increase of 85.4% (8.9 to 16.5 g N_2O-N/head/day) with increasing feed N intake (42.9 g N/day). To our knowledge this is the first direct measures of nitrous oxide from the feed pad as a single source (i.e. not a mixed source including lagoons and manure stockpiles). The source and extent of nitrous oxide production from manure consolidated on to the feed pad surface is poorly understood. Undoubtedly there would be a difference in urine output between animals managed on low and high protein (estimated urine volume: low - 14.3 to 25 litres/day vs. high - 17.2 – 30.2 litres/day: Dijkstra et al. 2013), but the fate of urea from urine and its subsequent transformations of NH_3 and NO_3^- via nitrification and denitrification to N_2 and N_2O is not known. The net retention of N in the carcass ranged from 7.8% (high protein) to 9.1% (low protein) of feed N consumed or 4.8 to 5.8 % (S2 and S1 respectively) of total N input. These data are similar to those of Rotz et al. (2013) and Chatzimpiros and Barles (2013) in terms of N retained and exported in the carcass as a proportion of feed N consumed, but are poor in terms of overall systems N use. One of the main reasons for the low N efficiencies of low and high protein system compared to US or European datasets is the consistent use of diets that contain greater than 14.5% crude protein (or 23.2 g N/kg DM) – a ration concentration greater than NRC recommendations (Gleghorn et al. 2004) of ca. 20.8 g N/kg DM (13% crude protein/kg DM).

5.3 Lignite as an direct and indirect greenhouse gas abatment technology

In Experiment 1, a 20% reduction in crude protein content (a comparison of rations containing 17.5% or 14.5% CP) of the ration, led to an estimated 35% reduction in ammonia emissions from the feed pad surface. However, the losses from stockpile manure were modelled as being greater for manure mass harvested from high protein ration feed pad compared to the low crude protein feeding system. These data clearly demonstrated that any method to reduce the overall losses of N (in the form of ammonia, N based GHG or manure) would improve the nitrogen use efficiency in intensive beef (feedlot) production systems as well as reducing costs associated with purchased-in N to the farm. Kagimbo et al. (2013) recently demonstrated that N losses from manure co-amended with lignite reduced ammonia losses by 72% compared to non-amended controls. The application of these technologies is not evident in the intensive animal industries reflecting the current practice of stockpiling of manure in the pen and outside prior to field application of manure with little or no pen amendment/mitigation strategy.

The three studies conducted on the potential mitigation of ammonia and nitrous oxide emissions from feed pads clearly demonstrated that direct emissions of ammonia could be reduced by up to 67%. The average over the three studies was ca. 40%. Nitrous oxide abatement was more variable and no clear pattern of abatement was noted. This level of abatement is promising as it demonstrates than ammonia can be sequestered in the lignite in the form of ammonium and hence a potential retention of N in manure. The N retained in manure from feed N intake in Experiment 3 were estimated as 34.5% for control in Period 1, 41.1% in lignite treated in Period 1 (or 19.1% increased retention), and in Period 2 control 21.4% with 31.5% retained in lignite (or 47.3% increase in retained N). On average, the physiological N responses (kg DM/kg N fertiliser) of sorghum to urea fertiliser and lignite amended manure were 26.8 kg DM/kg N and 16.8 kg DM/kg N. These physiological N responses were typical for forage sorghum (range from 12 to 40 kg DM/kg N; Lupatini et al. 1996).

An economic analysis of the use of lignite in feedlot systems was undertaken. The analysis is sensitive to transport costs (especially diesel and freight distance) of lignite to the feedlot and fluctuations in the price of synthetic fertilisers and other inputs to the farming system. It was estimated that when manure valuations (on the basis of nutrient retained in the farming systems), a net income per capita of $4.72 to $6.67 was possible reflecting the rate of lignite application. Further it was estimated (using average abatement of ammonia and nitrous oxide emissions from these studies) that the abatement potential of lignite is 2.04 kg CO2e/kg lignite added to the feedlot systems (without liabilities for transport or fugitive emissions). If a CFI method was to be developed for the abatement technology the net abatement return for the 13750 head feedlot system modelled would be equivalent to approximately $104,150 in credits or $7.57 per capita (exclusive of transaction and compliance costs).

The work undertaken in this project has clearly demonstrated that lignite can be considered as a cost effective mitigation technology that can be applied to feedlot systems. The emissions of ammonia can be reduced substantially from the feed pad by using a rate of between 3 to 4 kg/m^2 (based on economic analysis of a range of addition of lignite from 3 to 6 kg/m^2 over a period of 56 day manure harvesting periods). In contrast to the manure harvested from pens managed under differing crude protein contents, lignite retained N in the stockpile and its subsequent use in cropping systems achieved good yield increases (approximately 63% physiological N response in relation to urea controls). This retention of N inevitably reduces overall ration costs as increased yields of forages grown on land receiving lignite amended manure can be recycled into the feed formulation at a lower cost than purchased-in feeds.

6 Conclusions/recommendations

The project has clearly demonstrated that there is a potential abatement strategy (lignite) that is cost effective. The research conducted has provided good evidence that the addition of lignite to the feed pad is an effective strategy for mitigating ammonia emission from cattle feedlot and recycling nitrogen in agricultural systems. The findings have major economic and environmental implications for effective nitrogen management in agriculture, especially in intensive feedlot systems.

Key recommendations from the study are:

1. N use efficiency in the feedlot system is low in Australia, often less than 10%. The main pathways of N losses are in gaseous N forms (NH_3, NOx, N_2O and N_2). However, there is no long-term, complete measurements of these losses at the realistic scale not only in Australia but worldwide. Proposed work would include:

    a. quantification of gaseous N losses, as ammonia (and other gaseous N sources) from all sources over the whole production cycle;
    b. critical point analysis of emissions to evaluation of novel mitigation or sequestration technologies that can be applied at key management points of the N lifecycle and
    c. accurately model the N footprint of a range of feedlot systems and develop decision support tools to increase producers ability to use and manage surplus N in the production system (leading to progress towards a scientific benchmark for a sustainability index).

2. A scaled up version of the lignite pen studies is required to test proof of concept on a whole production system. This work could be achieved in conjunction with 1. above.

3. Better understanding of management of ration protein supply is required with future work focussing on energy supplementation and the possibility of using alternative feed supplies that do not contain as high a protein content.

7 Key messages

1. Feeding elevated levels of crude protein in the ration did not alter the N efficiency of the animal but did lead to manures with higher labile N content. Direct ammonia emissions from feed pads may not differ between manures from animals offered contrasting N contents, but losses from the stockpile are greater from manures of animals offered elevated levels of crude protein in the feed.

2. Lignite can be considered as a cost effective mitigation technology that can be applied to feedlot systems. The emissions of ammonia can be reduced substantially from the feed pad by using a rate of between 3 to 4 kg/m2 (based on economic analysis of a range of addition of lignite from 3 to 6 kg/m2 over a period of 56 day manure harvesting periods). In contrast to the manure harvested from pens managed under differing crude protein contents, lignite retained N in the stockpile and its subsequent use in cropping systems achieved good yield increases (approximately 63% physiological N response in relation to urea controls). This retention of N inevitably reduces overall ration costs as increased yields of forages grown on land receiving lignite amended manure can be recycled into the feed formulation at a lower cost than purchased-in feeds

8 Bibliography

Adams, J. R., Farran, T. B., Erickson, G. E., Klopfenstein, T. J., Macken, C. N., & Wilson, C. B. (2004). Effect of organic matter addition to the pen surface and pen cleaning frequency on nitrogen mass balance in open feedlots. JOURNAL OF ANIMAL SCIENCE, 82(7), 2153-2163.

Aguerre, M. J., Wattiaux, M. A., & Powell, J. M. (2012). Emissions of ammonia, nitrous oxide, methane, and carbon dioxide during storage of dairy cow manure as affected by dietary forage-to-concentrate ratio and crust formation. [Article]. Journal of Dairy Science, 95(12), 7409-7416.

ALFA. 2015. Australian Lot Feeders' Association. Media Release, 16 February 20

Ambus, P. & Robertson, G. P. 2006. The effect of increased N deposition on nitrous xide, methane and carbon dioxide fluxes from unmanaged forest and grassland communities in Michigan. Biogeochemistry, 79: 315-337.

Aneja, V. P., Blunden, J., James, K., Schlesinger, W. H., Knighton, R., Gilliam, W., Jennings, G., Niyogi, D. & Cole, S. 2008. Ammonia assessment from agriculture: US status and needs. Journal of Environmental Quality, 37: 515-520.

Arogo, J., P. W. Westerman, et al. (2001). "Ammonia emissions from animal feeding operations." In White Papers on Animal Agriculture and the Environment. National Centre for Manure and Animal Waste Management, North Carolina State Univ., Raleigh, NC. [Online]. Available: http://www.cals.ncsu.edu/waste_mgt/natlcenter/whitepapersummaries/ammoniaemissions.pdf [2012 Sep. 04].

Arogo, J., Westerman, P., Heber, A., Robarge, W. & Classen, J. 2002. Ammonia emissions from animal feeding operations. In National Center for Manure and Animal Waste Management White Papers, Animal Agriculture and the Environment. ASABE Publication 913C0306.

Afeng, Z., B. Rongjun, et al. (2012). "Effects of biochar amendment on soil quality, crop yield and greenhouse gas emission in a Chinese rice paddy: A field study of 2 consecutive rice growing cycles." Field Crops Research 127: 153-160.

Alho, C., A. D. Cardosoo, et al. (2012). "Biochar and soil nitrous oxide emissions." PESQUISA AGROPECUARIA BRASILEIRA 47(5): 722-725.

Ali, M. I. c. (2005). Struvite crystallization from nutrient rich wastewater. PhD Electronic resource, James Cook University.

Allen, S. J., L. J. Whitten, et al. (1997). "The Adsorption of Pollutants by Peat, Lignite and Activated Chars." Journal of Chemical Technology & Biotechnology 68(4): 442-452.

APEX (2010) The agricultural policy/environmental extender (APEX) model: The emerging tool for landscape and watershed environmental analyses. Transactions of the ASABE Vol. 53: 711-740

Asman, W. A. H. & van Jaarsveld, H. A. 1992. A variable-resolution transport model applied for NH_χ in Europe. Atmospheric Environment. Part A. General Topics, 26: 445-464.

Bai, M., Jianlei Sun, Kithsiri B. Dassanayake, Marcelo A. Benvenutti, Julian Hill, Owen T. Denmead, Thomas Flesch, Deli Chen (2015a) Non-interference measurement of CH4, N2O and NH3 emissions from cattle. Animal Production Science

Bai, M., Jianlei Sun, K.B.. Dassanayake, Marcelo A. Benvenutti, Julian Hill, Owen T. Denmead, Thomas Flesch, Deli Chen (2015b) Lignite reduces NH3 volatilization from the feedlot cattle pen. Animal Production Science submitted

Banath, C. L., & Holland, J. F. (1976). Iron pyrites as a sulphur fertilizer in an alkaline soil. Australian Journal of Experimental Agriculture, 16(80), 376.

Berg, W. E., & Kraatz, S. (2009). Emission Reduction by Brown Coal Powder. Paper presented at the 2009 ASABE Annual International Meeting Sponsored by ASABE, Paper Number: 097187, Grand Sierra Resort and Casino Reno, Nevada June 21 - June 24, 2009.

Beusen, A.H.W., Bouwman, A.F., Heuberger, P.S.C., van Drecht, G. and H.W. van der Hoek (2008) Bottom up uncertainty estimates of global ammonia emissions from global agricultural production systems. Atmosphere and Environment 42: 6067-6077.

Bierman, S.; Erickson, G. E.; Klopfenstein, T. J.; Stock, R. A.: Shain, D. H. “Evaluation of nitrogen and organic matter balance in the feedlot as affected by level and source of dietary fiber”, J. Anim. Sci. 1999, 77, 1645-1653.

Case, S. D. C., Whitaker, J., McNamara, N. P., & Reay, D. S. (2012). The effect of biochar addition on N2O and CO2 emissions from a sandy loam soil – The role of soil aeration. [doi: 10.1016/j.soilbio.2012.03.017]. Soil Biology and Biochemistry(0).

Capper JL. 2011 The environmental impact of beef production in the United States: 1977 compared with 2007. J Anim Sci. 12: 4249-61

Capper JL, Cady RA. 2012 A comparison of the environmental impact of Jersey compared with Holstein milk for cheese production. J Dairy Sci. 95:165-76.

Chatzimpiros, P., Barles, S., 2013. Nitrogen food-print: N use related to meat and dairy consumption in France. Biogeosciences 10 (1), 471–481

Chen, D, Li ,Y, Kelly, K. & Eckard, R. 2010. Simulation of N2O emissions from an irrigated dairy pasture treated with urea and urine in Southeastern Australia. Agriculture, Ecosystems and Environment, 136: 333–342.

Cole, N.A.; Defoor, P.J.; Galyean, M.L.; Duff, G.C.: Gleghorn, J.F. 2006 Effects of phase-feeding of crude protein on performance, carcass characteristics, serum urea nitrogen concentrations, and manure nitrogen of finishing beef steers, J. Anim. Sci. 2006, 84, 3421-3432.

Clough, T. J., J. E. Bertram, et al. (2010). "Unweathered Wood Biochar Impact on Nitrous Oxide Emissions from a Bovine-Urine-Amended Pasture Soil." SOIL SCIENCE SOCIETY OF AMERICA JOURNAL 74(3): 852-860.

Clough, T. J. and L. M. Condron (2010). "Biochar and the Nitrogen Cycle: Introduction." Journal Of Environmental Quality 39(4): 1218-1223.

Cole, N. A., R. W. Todd, et al. (2007). Use of Fat and Zeolite to Reduce Ammonia Emissions from Beef Cattle Feedyards, St Joseph, ASABE.

Crutzen, P.J. 1970. The influence of nitrogen oxides on the atmospheric ozone content. Quarterly Journal of the Royal Meteorological Society, 96: 320.

Denmead, O.T., Freney, J.R. & Dunin, F.X. 2008. Gas exchange between plant canopies and the atmosphere: Case-studies for ammonia. Atmospheric Environment 42: 3394-3406.

Denmend, O., Chen, D., Rowell, D., Loh, Z., Hill, J., Muir, S., Griffith, D., Naylor, T., Bai, M., Phillips, F. & McGinn, S. 2013. Gaseous nitrogen emissions from Australian cattle feedlots. In: Sutton, M., Mason, K., Sheppard, L., Sverdrup, H., Haeuber, R. & Hicks, W. (eds.) Nitrogen Deposition, Critical Loads and Biodiversity. Proceedings of the International Nitrogen Initiative Workshop, Linking Experts of the Convention on Long-range Transboundary Air Pollution and the Convention on Biological Diversity. Springer Publishing.

Dewes, T. (1999). Ammonia emissions during the initial phase of microbial degradation of solid and liquid cattle manure. Bioresource Technology, 70(3), 245-248. doi: http://dx.doi.org/10.1016/S0960-8524(99)00046-2

Dijkstra, J., Oenema, O., van Groenigen, J.W., Spek, J.W., van Vuuren, A.M. and Bannink, A. (2013) Diet effects on urine composition of cattle and N2O emissions. Animal 7:292–30

Dong, L., Córdova-Kreylos, A.L., Yang, J., Yuan, H. and Scow, K.M., 2009. Humic acids buffer the effects of urea on soil ammonia oxidizers and potential nitrification. Soil Biol. Biochem., 41(8): 1612-1621

Eckard, R. J., D. Chen, et al. (2003). "Gaseous nitrogen loss from temperate perennial grass and clover dairy pastures in south-eastern Australia." Australian Journal of Agricultural Research 54(6): 561-570.

Eckard, R. J., C. Grainger, et al. (2010). "Options for the abatement of methane and nitrous oxide from ruminant production: A review." Livestock Science 130: 47-56.

Erickson, G. E.; Milton C. T.: Klopfenstein T. J. 2000. Dietary protein effects on nitrogen excretion and volatilization in open-dirt feedlots, In Proceedings 8th International Symposium on Animal, Agricultural & Food Processing Waste. (ISAAF): Amer. Soc. Agric. Eng., St Joseph, MI, 2000.

Erikson, G.E.: Klopfenstein, T.J. 2001 Managing N inputs and the effect on N losses following excretion in open-dirt feedlots in Nebraska", The Scientific World Journal1, 830-835.

Flesch, T.K., Wilson, J.D., Harper, L.A., Crenna, B.P. and Sharpe, R.R., 2004. Deducing ground-to-air emissions from observed trace gas concentrations: A field trial. J. Appl. Meteorol., 43: 487-502.

Flesch, T.K., Wilson, J.D., Harper, L.A., Todd, R.W. and Cole, N.A., 2007. Determining ammonia emissions from a cattle feedlot with an inverse dispersion technique. Agr. Forest Meteorol., 144(1-2): 139-155.

Flesch, T.K., McGinn, S.M., Chen, D.L., Wilsona, J.D. and Desjardins, R.L., 2014. Data filtering for inverse dispersion emission calculations. Agri. Forest Meteorol., 198-199: 1-6.

Fowler, D., Pitcairn, C., Sutton, M., Flechard, C., Loubet, B., Coyle, M. & Munro, R. 1998. The mass budget of atmospheric ammonia in woodland within 1 km of livestock buildings. Environmental Pollution, 102: 343-348

Fox, D.G. Tedeschi, L.O., Tylutki, T.P., Russell, J.B., van Amburgh, M.E., Chase, L.E., Pell, A.N. and Overton, T.R. 2004. The Cornell Net Carbohydrate and Protein System model for evaluating herd nutrition and nutrient excretion. Anim. Feed Sci. Technol. 112(1-4):29-78.

Freer, M., Moore, A.D. and Donnelly, J.R. (1997) GRAZPLAN: Decision support systems for Australian gazing enterprises II. The animal biology model for feed intake, production and reproduction and the GrazFeed DSS. Agricultural Systems 54: 77–126.

Galloway, J.N. 1998. The global nitrogen cycle: changes and consequences. Environmental Pollution, 102: 15-24.

Galloway, J.N., Townsend, A.R., Erisman, J.W., Bekunda, M., Cai, Z., Freney, J.R., Martinelli, L.A., Seitzinger, S.P. & Sutton, M.A. 2008. Transformation of the nitrogen cycle: recent trends, questions, and potential solutions. Science, 320: 889-892.

Gleghorn, J.F.; Elam, N.A.; Galyean, M.L.; Duff, G.C.; Cole, N.A.: Rivera J.D. “Effects of crude protein concentration and degradability on performance, carcass characteristics, and serum urea nitrogen concentrations in finishing beef steers”, J. Anim. Sci. 2004, 82, 2705-2717.

Glendining, M. J. and J. U. Smith (1999). "Using a Rotational Modelling System to Explore the Effect of Straw Incorporation on the Efficiency of Nitrogen Use." Managing Risks of Nitrates to Humans & the Environment: 58-64.

Gonzalez, E. and Suirez, L.G. (2006) Methane conversion factors from cattle manure in Mexico. Atmosfera 20: 83-92

Harper, L.A. Flesch, T.K.; Todd, R.W.: Cole, N.A. “Air-quality and global-change gas emissions in a beef feeding operation”, Presented at the 2004 Annual Meetings, American Society of Agronomy, Seattle, WA. 2004

Harper, L.A., Sharpe, R.R., Langdale, G.W. & Giddens, J.E. 1987. Nitrogen cycling in a wheat crop: soil, plant, and aerial nitrogen transport. Agronomy Journal, 79:965-973

Hill, J and Leaver, J.D. 1999. Energy and protein supplementation by concentrate source on the intake and productivity of the lactating dairy cow given urea treated whole-crop wheat. Animal Feed Science and Technology 82: 177-193.

Hillel, D. and J. L. Hatfield (2005). Encyclopedia of soils in the environment / editor-in-chief, Daniel Hillel ; editors Jerry L. Hatfield ... [et al.], London : Elsevier, 2005.

Hwang, H. S., Oh, I. H., & Jang, Y. S. (2004). A study on ammonia reduction from swine manure slurry with additives of brown coal and oak charcoal. Paper presented at the Korean Society for Agricultural Machinery Proceedings

IPCC (1997) Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories: Volume 1, Greenhouse Gas Inventory Reporting Instructions; Volume 2, Greenhouse Gas Inventory Workbook; Volume 3, Greenhouse Gas Inventory Reference Manual IPCC/OECD/IEA, Paris, France.

IPCC (2000) Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories, Japan.

ISO 14040:2006 (Environmental Management – Life Cycle Assessment – Principles and Framework). International Standards Organisation, Geneva, Switzerland

ISO 14044:2006 (Environmental Management – Life Cycle Assessment – Requirements and guidelines) International Standards Organisation, Geneva, Switzerland

Isermann, K. 1994. Agriculture's share in the emission of trace gases affecting the climate and some cause-oriented proposals for sufficiently reducing this share. Environmental Pollution, 83: 95-111.

Jeffery, S., F. G. A. Verheijen, et al. (2011). "A quantitative review of the effects of biochar application to soils on crop productivity using meta-analysis." AGRICULTURE ECOSYSTEMS & ENVIRONMENT 144(1): 175-187.

Jones, FM, Phillips, FA, Naylor, T, Mercer, NB (2011) Methane emissions from grazing Angus beef cows selected for divergent residual feed intake. Anim. Feed Sci.Tech. 166, 302-307.

Kagimbo, F. M., Weatherley A., and Suter H. 2013 The effectiveness of lignite coal and biochar in reducing nitrogen (N) losses from cattle feedlot manure, Third RUFORUM Biennial Meeting 24 - 28 September 2012, Entebbe, Uganda.

Katzur, J., Fischer, K., Bocker, L., Tonder, K., & Liebner, C. (2002). Effect of N-modified brown coal on the soil biology. ARCHIVES OF AGRONOMY AND SOIL SCIENCE, 48(6), 647-662.

Kithome, M., Paul, J. W., & Bomke, A. A. (1999). Reducing Nitrogen Losses during Simulated Composting of Poultry Manure using Adsorbents or Chemical Amendments. J. Environ. Qual., 28(1), 194-201. doi: 10.2134/jeq1999.00472425002800010023x

Kephart, K. B. (2004). "Production and Storage Strategies for Controlling Ammonia Emissions " Retrieved April 04, 2012, from http://www.mawaterquality.org/public_education/wye_mills_materials/Kephart_NH3_3-041.pdf.

Klein, C. A. M. d., L. Barton, et al. (2003). "Estimating a nitrous oxide emission factor for animal urine from some New Zealand pastoral soils." Soil Research 41(3): 381-399.

Knowles, O. A., Robinson, B. H., Contangelo, A., & Clucas, L. (2011). Biochar for the mitigation of nitrate leaching from soil amended with biosolids. [doi: 10.1016/j.scitotenv.2011.05.011]. Science of The Total Environment, 409(17), 3206-3210.

Knowles, O. A., Robinson, B. H., Contangelo, A., & Clucas, L. (2011). Biochar for the mitigation of nitrate leaching from soil amended with biosolids. Science of the Total Environment, 409(17), 3206-3210.

Kookana, R. S., A. K. Sarmah, et al. (2011). Biochar application to soil. agronomic and environmental benefits and unintended consequences. 112: 103-143.

Laubach, J, Bai, M, Pinares-Patiño, CS, Phillips, FA, Naylor, TA, Molano, G, Cárdenas Rocha, EA, Griffith, DWT (2013) Accuracy of micrometeorological techniques for detecting a change in methane emissions from a herd of cattle. Agr. Forest Meteorol. 176, 50-63.

Leach, A.M., Galloway, J.N., Bleeker, A., Erisman, J.W., Kohn, R., Kitzes, J., 2012. A nitrogen footprint model to help consumers understand their role in nitrogen losses to the environment. Environ. Develop. 1 (1), 40–66

Li, Y., Chen, D., Zhang, Y., Edis, R. & Ding, H. 2005. Comparison of three modeling approaches for simulating denitrification and nitrous oxide emissions from loam-textured arable soils. Global Biogeochemical Cycles, 19: 1-15.27

Li, Y., White, R., Chen, D., Zhang, J., Li, B., Zhang, Y., Huang, Y. & Edis, R. 2007. A spatially referenced water and nitrogen management model (WNMM) for (irrigated) intensive cropping systems in the North China Plain. Ecological Modelling, 203: 395-423.

Li, C., Salas, W., Zhang, R. Krauter, C. Rotz, A. and Mitloehner, F. (2012)Manure-DNDC: a biogeochemical process model for quantifying greenhouse gas and ammonia emissions from livestock manure systems. Nutr Cycl Agroecosyst DOI 10.1007/s10705-012-9507-z

Li, Y., Chen, D., Barker-Reid, F. & Eckard, R. 2008. Simulation of N2O emissions from rain-fed wheat and the impact of climate change in south eastern Australia. Plant and Soil, 309: 239–251.

Li, Y., Barton, L. & Chen, D. 2012. Simulating response of N2O emissions to fertiliser N application and climatic variability from a rain-fed and wheat-cropped soil in Western Australia. The Journal of Science of Food and Agriculture. 92:1130-1143.

Li, Y., Wang, W., Reeves, S. & Dalal. C. 2013. Simulation of N2O emissions and mitigation options for rainfed wheat cropping on a Vertosol in the subtropics. Soil Research, 51: 152–166.

Liang, Y., J. J. Leonard, et al. (2006). "Influence of carbon and buffer amendment on ammonia volatilization in composting." Bioresource Technology 97(5): 748-761.

Leip, A. Franz Weiss, Tom Wassenaar, Ignacio Perez, Thomas Fellmann, Philippe Loudjani, Francesco Tubiello, David Grandgirard, Suvi Monni, Katarzyna Biala (2010): Evaluation of the livestock sector's contribution to the EU greenhouse gas emissions (GGELS) – final report. European Commission, Joint Research Centre.

Loh, Z., Chen, D., Bai, M., Naylor, T., Griffith, D., Hill, J., Denmead, T., McGinn, S. & Edis, R. 2008. Measurement of greenhouse gas emissions from Australian feedlot beef production using open-path spectroscopy and atmospheric dispersion modelling. Animal Production Science, 48: 244-247.

Loubet, B., Asman, W. H., Theobald, M., Hertel, O., Tang, Y. S., Robin, P., Hassouna, M., Dämmgen, U., Genermont, S., Cellier, P. & Sutton, M. 2009. Ammonia deposition near hot spots: processes, models and monitoring methods. In: Sutton, M., Reis, S. & Baker, S.H. (eds.) Atmospheric Ammonia. Springer Netherlands.

LRNS (2012) Large Ruminant Nutrition System. http://nutritionmodels.tamu.edu/lrns.html (accessed 28/01/2015)

Lupatini, G.C., E. L. Moojen, J. Restle, and J. H. S. Silva, “Resposta do milheto (Pennisetum americanum (L) Leeke) sob pastejo à adubação nitrogenada,” Pesquisa Agropecuária Brasileira, vol. 31, pp. 715–720, 1996

McGinn, S. M., Koenig, K. M., & Coates, T. (2002). Effect of diet on odorant emissions from cattle manure. CANADIAN JOURNAL OF ANIMAL SCIENCE, 82, 435-444.

McGinn, SM, Flesch, TK, Coates, TW, Charmley, E, Chen, D, Bai, M, Bishop-Hurley, G (2014) Evaluating dispersion modelling options to estimate methane emissions from grazing beef cattle. J. Environ. Qual. (in press)

McGinn, S.M., Chen, D., Loh, Z., Hill, J. Beauchemin, K.A. & Denmead, O.T. 2008. Methane emissions from feedlot cattle in Australia and Canada. Australian Journal of Experimental Agriculture, 48:183–185.

McGinn, S., Flesch, T., Chen, D., Crenna, B., Denmead, O., Naylor, T. & Rowell, D. 2010. Coarse particulate matter emissions from cattle feedlots in Australia. Journal of Environmental Quality, 39: 791-798.

Meat & Livestock Australia (2012) National Guidelines for Beef Cattle Feedlots in Australia. North Sydney, Australia.

Miller, D. N. and E. D. Berry (2005). "Cattle feedlot soil moisture and manure content: 1. Impacts on greenhouse gases, odor compounds, nitrogen losses, and dust." J. Env Qual. 34: 644-655.

Minson, D.J. and McDonald, C.K. (1987) Estimating forage intake from the growth of beef cattle. Tropical Grasslands 21:116–122.

Mosier, A., Kroeze, C., Nevison, C., Oenema, O., Seitzinger, S. & Van Cleemput, O. 1998. Closing the global N2O budget: nitrous oxide emissions through the agricultural nitrogen cycle. Nutrient Cycling in Agroecosystems, 52: 225-248.

Moore, P. A., Miles, D., Burns, R., Pote, D., Berg, K., & Choi, I. H. (2011). Ammonia Emission Factors from Broiler Litter in Barns, in Storage, and after Land Application. Journal Of Environmental Quality, 40(5), 1395-1404.

Muir (2012) Grenhouse gas emissions from Australian beef feedlots. PhD Thesis, University of Melbourne, Australia

NIR (2010) Australian National Greenhouse Accounts. Volume 1. Canberra, Australia

NRC (2007) Nutrient requirements of beef cattle. Seventh revised edition. National Academies Press.

Oussa, A. R. (1978). THE SORPTION OF AMMONIA ON BROWN COAL. Doctor of Philosophy, UNIVERSITY OF MELBOURNE, UNIVERSITY OF MELBOURNE.

Ozkan, S, Hill, J & Cullen, B.R. (2015) Effect of climate variability on pasture-based dairy feeding systems in south-east Australia. Animal Production Science (in press).

Parker, D. B., Pandrangi, S., Greene, L. W., Almas, L. K., Cole, N. A., Rhoades, A. B., & Koziel, J. A. (2005). Rate and frequency of urease inhibitor application for minimizing ammonia emissions from beef cattle feedyards. [Article]. TRANSACTIONS- AMERICAN SOCIETY OF AGRICULTURAL ENGINEERS, 48(2), 787-793.

Parker, D. B., Rhoades, M. B., Cole, N. A., & Sambana, V. P. (2012). Effect of urease inhibitor application rate and rainfall on ammonia emissions from beef manure. TRANSACTIONS OF THE ASABE, 55(1), 211-218.

Pierera M, Winiwartera,W., Leach, A.M. and Galloway, J.N. (2014) The nitrogen footprint of food products and general consumption 4 patterns in Austria. Food Policy (in press)

Powers, W. (2004). "Practices to Reduce Ammonia Emissions from Livestock Operations. PM 1971a " Retrieved August 12, 2012 from http://www.extension.iastate.edu/Publications/PM1971a.pdf.

Powlson, D. S. (1999). "3. Fate of Nitrogen from Manufactured Fertilizers in Agriculture." Managing Risks of Nitrates to Humans & the Environment: 42-57.

Qi, Y., Hoadley, A. F. A., Chaffee, A. L., & Garnier, G. (2011). Characterisation of lignite as an industrial adsorbent. Fuel, 90(4), 1567-1574. doi: 10.1016/j.fuel.2011.01.015

Reay,D.S., Davidson, E.A., Smith, K.A., Smith, P., Melillo, J.M., Dentener, F. and Crutzen, P.J. (2012) Global agriculture and nitrous oxide emissions Nature climate change 2:410-416

Ren, L., Schuchardt, F., Shen, Y., Li, G., & Li, C. (2010). Impact of struvite crystallization on nitrogen losses during composting of pig manure and cornstalk. [Article]. Waste Management, 30(5), 885-892. doi: 10.1016/j.wasman.2009.08.006

Ro, K. S., Reddy, G., Lima, I., Mahajan, D., & Cyrus, J. (2011). Ammonia adsorption capacity of biomass and animal-manure derived biochars. ABSTRACTS OF PAPERS OF THE AMERICAN CHEMICAL SOCIETY, 242.

Rogovska, N., Laird, D., Cruse, R., Fleming, P., Parkin, T., & Meek, D. (2011). Impact of biochar on manure carbon stabilization and greenhouse gas emissions. Soil Science Society of America Journal, 75(3), 871-879.

Rothman, LS, Jacquemarta, D, Barbeb, A, Chris Bennerc, D, Birkd, M, Browne, LR, Carleerf, MR, Chackerian, C, Chancea, JgK, Couderth, LH, Danai, V, Devic, VM, Flaudh, J-M, Gamachej, RR, Goldmank, A, Hartmannh, J-M, Jucksl, KW, A.G. Makim, KW, Mandini, J-Y, Massien, ST, Orphalh, J, Perrinh, A, Rinslando, CP, Smitho, MAH, Tennysonp, J, Tolchenovp, RN, Tothe, RA, Vander Auweraf, J, Varanasiq, P, Wagnerd, G (2005) The HITRAN 2004 molecular spectroscopic database. J. Quant. Spectrosc. Ra. 96, 139-204.

Rotz, C.A., F. Montes, S.D. Hafner, A.J. Heber, and R.H. Grant. 2013. Ammonia emission model for whole farm evaluation of dairy production systems. J. Environ. Qual. doi:10.2134/jeq2013.04.0121

Sanderson, M. G., Collins, W. J., Johnson, C. E. & Derwent, R. G. 2006. Present and future acid deposition to ecosystems: The effect of climate change. Atmospheric Environment, 40: 1275-1283.

SCA (1990) SCA (Standing Committee on Agriculture) (1990) Feeding standards for Australian livestock, Ruminants. SCA Ruminant Sub-Committee, CSIRO Australia.

Shcherbak,, L., Millara, N. & Robertson, G.P. 2014. Global metaanalysis of the nonlinear response of soil nitrous oxide (N2O) emissions to fertilizer nitrogen. PNAS, 111: 9199–9204.

Shand, C. A., Williams, B. L., Smith, S., & Young, M. E. (2000). Temporal changes in C, P and N concentrations in soil solution following application of synthetic sheep urine to a soil under grass. Plant and Soil, 222(1-2), 1-13.

Shi, Y., Parker, D. B., Cole, N. A., Auvermann, B. W., & Mehlhorn, J. E. (2001). Surface amendments to minimize ammonia emissions from beef cattle feedlots. Transactions of the ASAE, 44(3), 677-682.

Singh, B. P., B. J. Hatton, et al. (2010). "Influence of biochars on nitrous oxide emission and nitrogen leaching from two contrasting soils." Journal of Environmental Quality 39(4): 1224-1235.

Staebler, RM, McGinn, SM, Crenna, BP, Flesch, TK, Hayden, KL, Li, S-M (2009) Three-dimensional characterization of the ammonia plume from a beef cattle feedlot. Atmos. Environ. 43, 6091-6099.

Steiner, C., Das, K. C., Melear, N., & Lakly, D. (2010). Reducing Nitrogen Loss during Poultry Litter Composting Using Biochar. [Article]. Journal Of Environmental Quality, 39(4), 1236-1242. doi: 10.2134/jeq2009.0337

Stevens, R. J., R. J. Laughlin, et al. (1998). Soil Biol. Biochem. 30: 1119.

Sutton, M., Pitcairn, C. E. & Fowler, D. 1993. The exchange of ammonia between the atmosphere and plant communities, Academic Press.

Taghizadeh-Toosi, A., T. Clough, et al. (2012). "Biochar adsorbed ammonia is bioavailable." Plant and Soil 350(1): 57-69.

Tiwari, K. N., Dwivedi, B. S., & Pathak, A. N. (1986). Relative effectiveness of iron pyrites and gypsum as sulfur fertilizer for different cultivars of mustard (<i>Brassica juncea</i>). Nutrient Cycling in Agroecosystems, 8(3), 279.

Todd, R. W., N. A. Cole, L. A. Harper, T. K. Flesch, and B. H. Baek. 2005. Ammonia and gaseous nitrogen emissions from a commercial beef cattle feedyard estimated using the flux-gradient method and N: P ratio analysis. Proc. Symposium State of the Science: Animal Manure and Waste Management, Jan. 5–7, 2005, San Antonio, TX.

Todd, R.W., Cole, N.A., Rhoades, M.B., Parker, D.B. & Casey, K.D. 2011. Daily, monthly, seasonal and annual ammonia emissions from southern high plains cattle feedyards. Journal of Environmental Quality, 40:1-6

Uludag-Demirer, S., Demirer, G. N., & Chen, S. (2005). Ammonia removal from anaerobically digested dairy manure by struvite precipitation. [Article]. Process Biochemistry, 40, 3667-3674. doi: 10.1016/j.procbio.2005.02.028

Uzinger, N., Rékási , M., Draskovits, E. and Anton, A., 2014. Stabilization of Cr, Pb, and Zn in soil using lignite. Soil Sediment Contam., 23(3): 270-286

Varel, V. H., Nienaber, J. A., & Freetly, H. C. (1999). Conservation of nitrogen in cattle feedlot waste with urease inhibitors. Journal of Animal Science, 77(5), 1162-1168.

Varel, V. H., Wells, J. E., & Miller, D. N. (2007). Combination of a urease inhibitor and a plant essential oil to control coliform bacteria, odour production and ammonia loss from cattle waste. Journal of Applied Microbiology, 102(2), 472-477. doi: 10.1111/j.1365-2672.2006.03120.x

Van Breemen, N. & Van Dijk, H. 1988. Ecosystem effects of atmospheric deposition of nitrogen in the Netherlands. Environmental Pollution, 54: 249-274.

Van Haarlem, R., Desjardins, R., Gao, Z., Flesch, T. & Li, X. 2008. Methane and ammonia emissions from a beef feedlot in western Canada for a twelve-day period in the fall. Canadian Journal of Animal Science, 88: 641-649.

VanderZaag, A. C., S. Jayasundara, et al. (2011). "Strategies to mitigate nitrous oxide emissions from land applied manure." Animal Feed Science and Technology 166-167: 464-479.

Voorburg, J. & Kroodsma, W. 1992. Volatile emissions of housing systems for cattle. Livestock Production Science, 31: 57-70.

Wang, J., M. Zhang, et al. (2011). "Effects of biochar addition on N 2O and CO 2 emissions from two paddy soils." Biology and Fertility of Soils 47(8): 887-896.

Webb, P. A. (2003). "Introduction to Chemical Adsorption Analytical Techniques and their Applications to Catalysis." MIC Technical Publications Micromeritics Instrument Corp., Norcross, Georgia 30093(Jan-2003).

Webb, J., Menzi, H., Pain, B., Misselbrook, T., Dämmgen, U., Hendriks, H. & Döhler, H. 2005. Managing ammonia emissions from livestock production in Europe. Environmental Pollution, 135: 399-406.

Wiegand, A.N., Menzel,, S. King, R., & Tindele, N. 2011. Modeling the aeolian transport of ammonia emitted from poultry farms and deposited to a coastal waterbody. Atmospheric Environment, 45:5732-5741.

Wilson, S. M. & Serre, M. L. 2007. Examination of atmospheric ammonia levels near hog CAFOs, homes, and schools in Eastern North Carolina. Atmospheric Environment, 41: 4977-4987

Wortmann, C. S. and C. A. Shapiro. (2012, 2012). "Composting manure and other organic materials." NebGuide http://elkhorn.unl.edu/epublic/live/g1315/build/#target3 Retrieved 30 August, 2012, from http://ianrpubs.unl.edu/epublic/live/g1315/build/g1315.pdf.

Yao, Y., Gao, B., Zhang, M., Inyang, M., & Zimmerman, A. R. (2012). Effect of biochar amendment on sorption and leaching of nitrate, ammonium, and phosphate in a sandy soil. Chemosphere(0). doi: 10.1016/j.chemosphere.2012.06.002

Yan (2014) Nitrogen recovery efficiencies in crop production systems of China with and without consideration of the residual effect of nitrogen. Environmental Research Letters, 9: 95002 (9 pp).

Zaman, M., & Nguyen, M. L. (2010). Effect of lime or zeolite on N2O and N2 emissions from a pastoral soil treated with urine or nitrate-N fertilizer under field conditions. Agriculture, Ecosystems & Environment, 136(3–4), 254-261. doi: http://dx.doi.org/10.1016/j.agee.2009.12.002