

**Using a Systems Approach to Examine Net Greenhouse Gas Emissions
from Beef Production in Western Canada**

A Thesis Submitted to
The Faculty of Graduate Studies
The University of Manitoba

By
Ashley Ann Stewart

In partial fulfillment of requirements for the degree of
Master of Science
Department of Animal Science

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FACULTY OF GRADUATE STUDIES

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ABSTRACT

A model, based on Intergovernmental Panel on Climate Change (IPCC) equations, was used to estimate annual net farm greenhouse gas (GHG) emissions from various management strategies. The model included methane (CH_4) emissions from livestock and manure, direct and indirect nitrous oxide (N_2O) emissions from soil and manure, carbon dioxide (CO_2) emissions from energy use and soil carbon change. The model was used to examine the effects of 11 management practices (one baseline management scenario and ten variations of that baseline) on net whole-farm emissions from a beef production system, as estimated for hypothetical farms at four disparate locations in western Canada. Treatments from a systems-based research trial were also modeled, and treatment rankings obtained were examined along with those acquired from modeled predictions. The measured emissions were acquired from a field study which examined the mitigation potential of three fertility treatments applied to grazed forage. The treatments were no liquid hog manure (control); 242 kg total N/ha in a spring application of liquid hog manure (full); 121 kg total N/ha in each of a spring and fall application of liquid hog manure (split). Greenhouse gas emissions for a hypothetical treatment, where synthetic fertilizer was applied at 242 kg total N/ha in the spring, were estimated using the model and examined along with the other fertility treatments to determine mitigation potential. The discrepancies observed between the predicted estimates and measured values were used to identify those facets of whole-farm emissions most in need of further study and those components with the largest effect on net emissions. Emissions were reported as net farm emissions ($\text{Mg CO}_2\text{equivalents (CO}_2\text{e)}$), net farm emissions per hectare ($\text{Mg CO}_2\text{e/ha}$) and as net emissions per unit of protein exported off-farm (Mg

CO₂e/Mg protein). The latter strategy was utilized to ensure that farm productivity was accounted for. Of the ten management practices that were compared to the baseline management scenario, pasturing cattle on alfalfa-grass showed the largest decrease (0.39 to 0.70 Mg CO₂e/Mg protein) in emissions for all locations, while feeding lower quality forage over winter showed the greatest increase in emissions per unit protein on the southern Alberta (S.AB) (1.14 Mg CO₂e/Mg protein) and northern Alberta (N.AB) (1.09 Mg CO₂e/Mg protein) farms. Eliminating the fertilization of forages resulted in the largest increase (2.36 Mg CO₂e/Mg protein) in emissions per unit protein on the Saskatchewan (SK) farm, while reducing the fertilizer rate by half for all crops showed the largest increase (2.26 Mg CO₂e/Mg protein) on the Manitoba (MB) farm. The predictions and measured values for the fertility treatments showed the following ranking among treatments in net emissions per ha (Mg CO₂e/ha): full > split > synthetic fertilizer > control (measured emissions did not include the synthetic fertilizer treatment). The predicted estimates and measured values showed the following rankings when expressed per unit protein: split > full > synthetic fertilizer > control (measured emissions did not show as much difference between split and full). The analyses indicate that a systems-based approach must be used to quantify net farm GHG emissions, and expressing emissions on the basis of CO₂e per unit of protein exported off-farm provides a more accurate assessment of the impact of management changes. General recommendations on 'best' management practices cannot be made, as factors influencing GHG emissions differ with location. Uncertainty exists in both predicted and measured estimates of GHG emissions, and future research work should use both models and measurements to focus on the system components with the largest uncertainty and highest relative importance.

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ABBREVIATIONS

ADG	=	average daily gain
AG _{residue N}	=	nitrogen in above ground plant residue
ASH	=	manure ash content
BG _{residue N}	=	nitrogen in below ground plant residue
Bo	=	maximum methane producing capacity of manure
C	=	carbon
CH ₄	=	methane
CO ₂	=	carbon dioxide
CO ₂ e	=	carbon dioxide equivalents
CP	=	crude protein
DM	=	dry matter
DMI	=	dry matter intake
EF	=	emission factor
EF _{eco}	=	ecodistrict emission factor
E _x	=	energy requirement for fuel use or production of machinery and herbicides
g	=	grams
GEI	=	gross energy intake
GHG	=	greenhouse gas
Gt	=	gigatonnes; 10 ¹⁵ g; 1 petagram (Pg)
GWP	=	global warming potential
ha	=	hectare
kg	=	kilogram
MB	=	Manitoba
MCF	=	methane conversion factor
Mg	=	megagrams
Mt	=	megatonnes; 10 ¹² g; 1 teragram (Tg)
N	=	nitrogen
N.AB	=	northern Alberta
NE	=	net energy
NH ₃	=	ammonia
NH ₄ ⁺	=	ammonium ion
NI	=	nitrogen intake
NO ₂ ⁻	=	nitrite
NO ₃ ⁻	=	nitrate
N ₂ O	=	nitrous oxide
O ₂	=	oxygen
P	=	phosphorus
PI	=	protein intake
PR	=	protein retention
RFI	=	residual feed intake
S.AB	=	southern Alberta
SC	=	soil carbon change coefficient

SK	=	Saskatchewan
TDN	=	total digestible nutrients
TS	=	total solids
VFA	=	volatile fatty acid
VS	=	volatile solids
Ym	=	percent gross energy intake lost as methane

1.0 INTRODUCTION

In 2005, beef cattle in Canada released roughly 22 megatonnes carbon dioxide equivalents (Mt CO₂e) of methane (CH₄) through enteric fermentation and manure management, approximately 7 Mt CO₂e more than in 1990 (Collas and Liang 2007). It has been estimated that by 2030, global livestock-related CH₄ emissions could increase by 60% (Smith et al. 2007). Given that Canada has committed to a 6% reduction in emissions relative to 1990 through its commitment to the Kyoto Protocol, it is necessary to reduce emissions by roughly one third annually to reach Kyoto commitments (Environment Canada 2007).

A systems approach to calculating net greenhouse gas (GHG) emissions from beef production is needed because complex relationships are inherent to farming systems. Studying only one GHG ignores these interactions (Robertson and Grace 2004; Gregorich et al. 2005; Schils et al. 2005) and the effects on net emissions that a change in management may have. Although considerable GHG research has been conducted in western Canada to identify mitigation strategies associated with individual GHG's, very few field-scale studies have been conducted in which a systems-based approach examining multiple GHG's from multiple sources has been explored. Using a systems-based approach to examine agricultural ecosystems will allow us to learn how to better manage our agricultural land so that we not only optimize productivity and economic sustainability, but also manage the land for optimum ecosystem health and maintenance of other essential services, such as clean water, habitat for beneficial wildlife and soil carbon (C) sequestration (Robertson and Swinton 2005).

Models make it possible to link the flows and feedbacks within a system so that a meaningful estimate of net GHG emissions for a system can be produced. Large differences in GHG emissions exist between locations (even those in close proximity to one another) and seasons, making it difficult to obtain accurate regional estimates without the use of models (Gregorich et al. 2005).

The aim of the study was to estimate GHG emissions resulting from beef cattle production in a systems context. Our goal was to demonstrate that management changes in one area of the system have implications in other areas, and these have the potential to off-set any benefit derived from the management change. The study used a simple Intergovernmental Panel on Climate Change (IPCC)-based model to estimate net emissions from various different management scenarios on a beef cattle operation in four regions of the Canadian prairies. The same model was used to simulate a specific site in Manitoba, which examined three pasture management treatments. Model output of net emissions was compared and contrasted with actual measured emissions from the site.

2.0 LITERATURE REVIEW

Atmospheric gases, such as carbon dioxide (CO₂), methane (CH₄) and nitrous oxide (N₂O), are referred to as greenhouse gases (GHG's). They allow solar radiation to pass through the atmosphere to the earth, but absorb the infrared radiation from the earth, warming the atmosphere near the surface of the earth (Houghton 2004). Known as the “greenhouse effect”, this process is essential to life on earth as it keeps the planet at a habitable temperature. Individual GHG's differ in their ability to absorb radiation from the earth. According to Solomon et al. (2007), global warming potentials (GWP) “compare the integrated radiative forcing over a specified period (e.g. 100 years) from a unit mass pulse emission and are a way of comparing the potential climate change associated with emissions of different greenhouse gases”. Recently, the concentrations of the GHG's have been increasing, leading to higher average global temperature, a rise in global average sea level, a higher incidence and severity of extreme weather events and a change in precipitation patterns (Intergovernmental Panel on Climate Change (IPCC) 2007).

2.1. AGRICULTURE AS A SOURCE (AND SINK) OF GHG's

2.1.1. Global perspective

Increases in atmospheric concentrations of CO₂, CH₄ and N₂O have been observed since the industrial revolution (Table 1). The increase in global atmospheric CO₂ levels since the pre-industrial period is attributed to fossil fuel use and, to a lesser

extent, land use change (primarily deforestation). Rising atmospheric concentrations of CH₄ are due to agricultural activity and the use of fossil fuels. Anthropogenic increases in global N₂O concentrations also have been attributed to the agricultural sector (IPCC 2007).

Table 1. Change in atmospheric concentrations of CO₂ (ppm), CH₄ (ppb) and N₂O (ppb) from the pre-industrial period to 2005.

Atmospheric Concentration	CO ₂ (ppm)	CH ₄ (ppb)	N ₂ O (ppb)
Pre-industrial period [*]	280	715	270
2005 [*]	379	1774	319
% increase	35	148	18

^{*} IPCC (2007)

According to Smith et al. (2007), agriculture, on a global basis, was responsible for 3.3 and 2.8 Gt CO₂equivalent (CO₂e) per year of CH₄ and N₂O, respectively, in 2005 (1 Gt = 1 Pg = 10¹⁵ g). Sixty percent of global anthropogenic N₂O and 50% of global anthropogenic CH₄ was produced by agriculture in 2005. The net flux of CO₂ in global agricultural systems is roughly balanced, with annual emissions of about 0.04 Gt CO₂/year. From 1990 – 2005, global emissions of agricultural CH₄ and N₂O increased 17%. Smith et al. (2007) estimated that 88% of this increase can be attributed to biomass burning (which releases both CH₄ and N₂O), enteric fermentation and soil N₂O emissions. They estimated that by 2030, livestock-related CH₄ emissions will increase by 60% if no changes in feeding practices or improvements in manure management are made. Even with these changes, emissions may still increase by about 21% by 2020. Agricultural N₂O emissions are expected to increase 50% from 1990 levels by 2020, and

35 – 60% by 2030 because of increased production of livestock manure and increased nitrogen (N) fertilizer use. These estimates indicate the importance of seeking ways to stem the rise in agricultural GHG emissions.

2.1.2. Canadian Perspective

Canada produces approximately 2% of global GHG (Neitzert et al. 2007). Due to its large size, climate-induced energy demands and a resource-based economy, Canada is one of the highest emitting countries per capita. In 2005, Canada emitted 747 Mt CO₂e – approximately 583, 5.2 and 0.14 Mt each of CO₂, CH₄ and N₂O (Neitzert et al. 2007).

Agriculture is responsible for approximately 8% of Canada's GHG emissions (or 57 Mt CO₂e), and 10% if CO₂ emissions from energy use are included (Janzen et al. 2008).

However, as the CO₂ emissions are attributed to the transportation and manufacturing sectors, they are not included in the values provided in Table 2. Agriculture produced 25 and 66% of Canada's CH₄ and N₂O emissions, respectively (Neitzert et al. 2007).

Livestock production is directly responsible (through enteric fermentation, manure management and manure deposited onto pasture) for approximately 59% of Canada's agricultural GHG emissions, while agricultural soils contribute the remainder.

Table 2. Sources of agricultural GHG's and their contribution to Canada's agricultural emissions in 2005.*

Emission Source	CH ₄ emissions (Mt CO ₂ e)	N ₂ O emissions (Mt CO ₂ e)	% of Canada's Agricultural Emissions	% of Canada's Total Emissions
Enteric Fermentation	25.0	0.0	43.8	3.3
Manure Management	3.2	5.4	15.4	1.2
Agricultural Soils	0.0	23.0	40.4	3.1

* Neitzert et al. (2007)

Methane and N₂O emissions from Canadian agriculture increased 19% from 1990 to 2005 (Table 3). This increase in emissions is due to increased use of N fertilizer and to larger numbers of beef cattle, poultry and swine on Canadian farms (Collas and Liang 2007; Janzen et al. 2008).

Table 3. The increase in CH₄ and N₂O emissions from each agricultural CH₄ and N₂O source from 1990 – 2005.

Emission Source of CH ₄ and N ₂ O	1990 GHG Emissions* (Mt CO ₂ e)	2005 GHG Emissions* (Mt CO ₂ e)	Emissions Increase (%)
Agriculture	46.0	57.0	19
CH ₄ - Enteric Fermentation	18.0	25.0	28
CH ₄ - Manure Management	6.7	8.6	22
N ₂ O – Manure Management	4.1	5.4	24
N ₂ O – Agricultural Soils	21.0	23.0	9

* Collas and Liang (2007)

2.2. GREENHOUSE GAS PRODUCTION AND REMOVAL IN A BEEF PRODUCTION SYSTEM

2.2.1. Methane

2.2.1.1. Enteric methane

The anatomy and physiology of ruminant animals cause them to generate more CH₄ than other types of animals (Monteny et al. 2001). Beef cattle produced 84% of Canadian enteric CH₄, and 19% of CH₄ in 2005 (Collas and Liang 2007; Neitzert et al. 2007). The emission of CH₄ from cattle is a concern from both an environmental and

production standpoint, as the release of CH_4 can represent a loss of anywhere from 2 – 12% of gross energy intake (GEI) (Johnson and Johnson 1995; Lassey 2008).

Enteric CH_4 is produced by microorganisms called methanogens. Most rumen methanogens belong to the *Methanobacteriaceae* family, and more specifically, the *Methanobrevibacter* genus (Skillman et al. 2006). Methanogens are anaerobic microorganisms that only exist in environments where there is a redox potential below - 290 mV (Stewart and Bryant 1988). Most species require a pH of between six and eight, although some are able to survive at a pH between one and 9.5 (Jones et al. 1987).

Several mechanisms and microorganisms are involved in the formation of CH_4 (Figure 1). As described by Miller (1991), simple sugars and amino acids are produced from the hydrolysis of starch, plant cell wall polymers and proteins by bacteria, fungi and protozoa. Next, the simple sugars and amino acids are fermented to volatile fatty acids (VFAs) (either acetate, propionate or butyrate), hydrogen and CO_2 . Finally, the rumen methanogens use mainly hydrogen, but also formate, as a substrate for CH_4 formation.

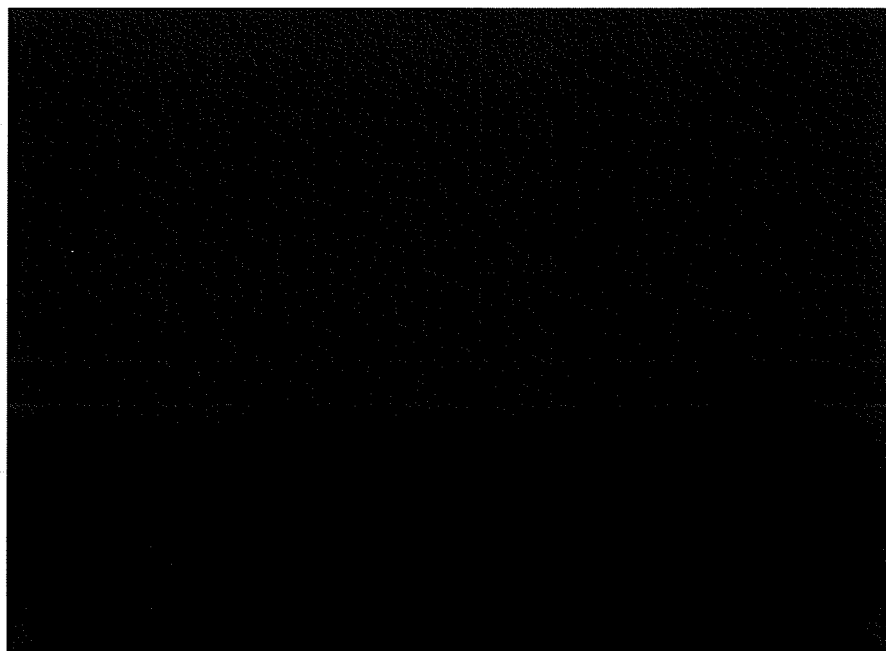


Figure 1. Microbial fermentation in the rumen (McAllister et al. 1996).

Research conducted using radioisotopes demonstrated that 87% of CH_4 is produced in the rumen, with the remainder produced in the lower digestive tract (Murray et al. 1976). Much of that produced in the lower digestive tract is absorbed into the portal blood and emitted via the lungs, leaving only 1% that is lost through the anus (Murray et al. 1976). More recently, McGinn et al. (2006) have demonstrated that differences between the chamber technique (respired and hind gut emissions) and SF_6 technique (respired emissions only) were approximately 4%, which were attributed to post-ruminal loss of CH_4 , as well as other losses associated with recovery of gas from the canister.

2.2.1.2. Methane produced from manure

Methane from manure is produced during the reduction of fatty acids (such as acetate) present in the manure (Clemens and Ahlgrimm 2001). However, CH_4 can only be formed when conditions are favourable. Methanogens require an anaerobic environment, a neutral pH, available electron acceptors, sufficient nutrient availability and a substrate with high organic matter content (Conrad 1989). As well, most methanogens require a temperature between 30 – 40°C (Conrad 1989), although some can survive at higher or lower temperatures (Jones 1991). The production of CH_4 in manure is similar to the processes that occur in the rumen (Figure 1) – microorganisms break down organic matter into simple organic compounds that can be used by the methanogens to produce CH_4 (Conrad 1989; Boone 1991).

Methane generated during liquid manure storage can only reach the atmosphere if its diffusion pathway is oxygen-free (or oxygen-limited). Liquid manure storages may release less CH_4 than they produce if oxygen (O_2) is present near the surface of the

manure storage and the CH_4 is prevented from moving quickly to the surface, allowing oxidation to occur (Hao et al. 2001). In situations where the CH_4 release is episodic, the CH_4 may form into bubbles (Whalen 2005). Upon release, the bubbles allow the CH_4 to reach the atmosphere without being subject to oxidation (Whalen 2005); however, a crust covering the manure storage system can slow the CH_4 bubbles down, and expose them to some oxidation (Petersen et al. 2005). Although manure kept in storage systems can be a significant source of CH_4 , raw manure applied directly to agricultural land produces almost zero CH_4 emissions because of adequate aeration after application (Hao et al. 2001).

2.2.1.3. Methane production/consumption in soils

Methane is produced in anaerobic soils when organic matter is oxidized by reducing CO_2 (Mosier et al. 1998a). Rice paddies and wetlands are an example of this process. If these soils lose moisture, large releases of accumulated CH_4 are possible (Garcia 1990). Plants can slow the production of soil CH_4 by releasing O_2 into the root zone. However, plants also transport CH_4 to the atmosphere, allowing it to reach the soil surface without being exposed to methanotrophs (Topp and Pattey 1997). Methane is only formed in the soil if other electron acceptors, such as nitrate (NO_3^-), sulphate or ferric iron, are not present (Garcia 1990; Jones 1991). However, if O_2 is present, soil methanotrophic bacteria consume CH_4 as a source of carbon (C) and energy, releasing CO_2 (Mosier et al. 1998a). Although measurable, the rates of CH_4 consumption by soils are often negligible compared to those of other GHG's, when all are compared as CO_2e (e.g., Ellert and Janzen 2008).

2.2.2. Carbon Dioxide

The sources and sinks of CO₂ are tied together by the carbon cycle (Figure 2). Carbon produced by one component of the cycle is taken up by another. When more CO₂ is produced than is taken up, the balance of the cycle is disrupted and atmospheric levels of CO₂ rise (Janzen 2004).

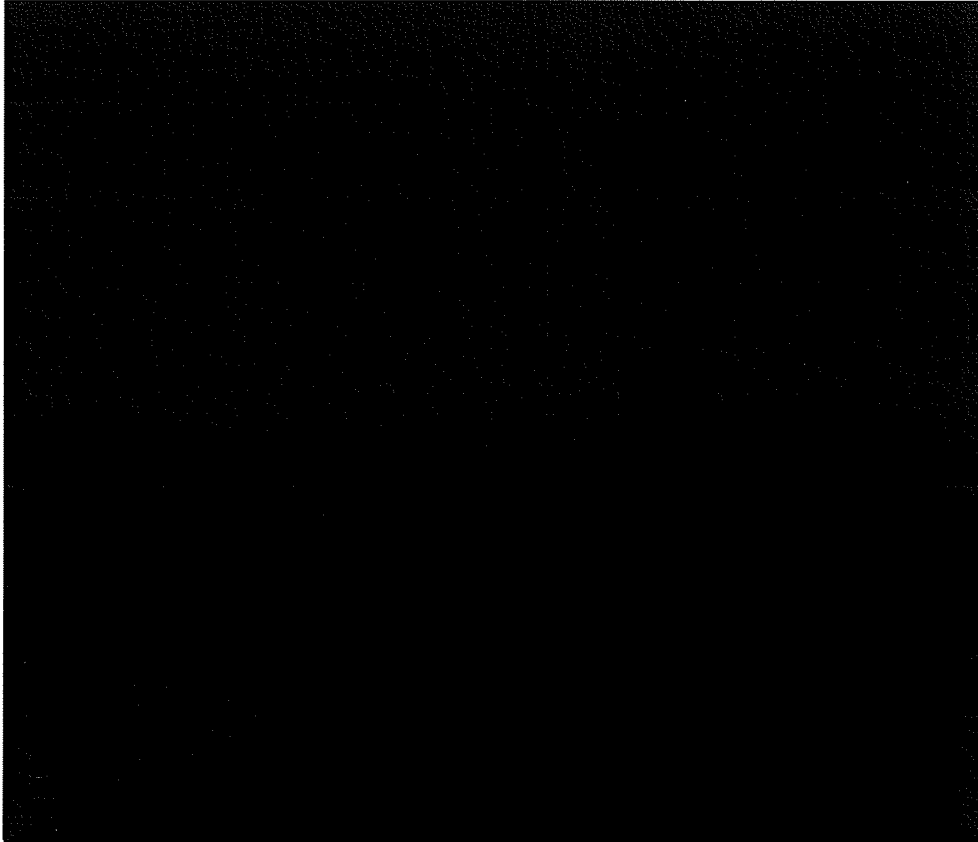
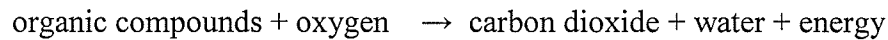


Figure 2. Global C cycle showing fossil C stock, CO₂ emissions, and the fate of CO₂ in the 1990's. Carbon stocks are in units of Pg C; annual flows and changes in atmospheric CO₂ are in Pg C/yr (Janzen 2004).

2.2.2.1. Carbon dioxide production through respiration and uptake via photosynthesis

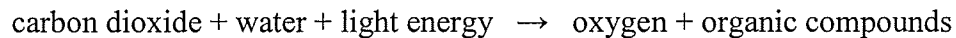
Carbon dioxide is released to the atmosphere mostly by respiration, which occurs in the mitochondria of microorganism, plant and animal cells. Carbon dioxide is a by-

product of this process by which food is broken down to create energy. Respiration can be summarized as:



The organic compounds consumed during respiration can be fats, proteins or carbohydrates. Carbon dioxide is formed during the first two stages of respiration – glycolysis and the Krebs cycle. Glycolysis, which occurs in the cytosol of cells, converts glucose to pyruvate. In the mitochondrial matrix, CO₂ is formed as a by-product as pyruvate is broken down to other compounds in a series of Krebs cycle reactions (Campbell et al. 1999).

The reverse process of respiration is photosynthesis. Whereas respiration is a source of CO₂, photosynthesis removes CO₂ from the atmosphere. In the chloroplasts of plant cells, CO₂ is converted to O₂ and organic compounds. Photosynthesis can be summarized as:



There are two stages in photosynthesis – the light reactions and the Calvin cycle. Oxygen is formed during the light reactions in the chloroplasts. Chlorophyll in the chloroplasts absorb light energy, which causes water to be split into hydrogen and O₂ when the chlorophyll transfers hydrogen and electrons to nicotinamide adenine dinucleotide phosphate (NADP⁺). The O₂ from the split water molecule is released from the plant as a by-product (Campbell et al. 1999).

Respiration by soil microbes while decomposing above-ground dead plant material and plant roots (collectively referred to as soil respiration), is one of the major C fluxes in most ecosystems (Davidson et al. 2002). While soil microbes produce CO₂

through respiration, a roughly equivalent amount of C enters soil via photosynthesis (Paustian et al. 1998) (Figure 2). The net differences in C inputs and C losses determine the rate of soil C change. Soil C decreases if C losses are greater than C inputs, whereas if C inputs are higher than C losses, soil C accumulates.

2.2.2.2. Soil carbon change resulting from land use change

Since 1700, the global area of land used for pastures and crops has increased by approximately 1200 and 2930 million hectares (ha), respectively, at the expense of forests and grasslands (Goldewijk 2001). Upon conversion to cultivated cropland, as much as 30% or more of the soil C may be lost (e.g., Janzen et al. 1998). Use of land for agriculture often reduces the amount of biomass (and therefore C) returned to the soil (Lal 2004a), as a large proportion of the C captured in plants is removed from the land (Janzen et al. 1997). When land use change occurs, C is lost through the decomposition of vegetation and the mineralization of soil organic C (Lal 2004b).

Conversely, C loss can be reversed if agricultural management practices intended to increase C sequestration are implemented (Conant et al. 2001). Carbon sequestration is described by Lal (2004a) as “the net removal of CO₂ from the atmosphere into long-lived pools of C, such as terrestrial and geologic”. In theory, as much as 50 – 66% of the soil C lost can be recovered with good management (Lal 2004c). In agricultural soils that have been returned to grasslands, it has been found that soil C accumulates at an average global rate of 33.2 g C/m²/year (Post and Kwon 2000). Beef production can maintain or enhance soil C reserves because grasslands and perennial forages tend to promote soil C conservation (Janzen et al. 1998).

2.2.2.3. Carbon dioxide release from the use of energy

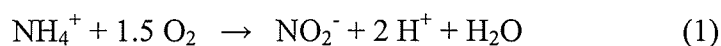
In a beef production system, CO₂ results from fossil fuel use for daily operations (such as delivering feed to cattle or harvesting feed crops) and energy use during the production of machinery and agricultural inputs (fertilizers and pesticides) (West and Marland 2002). The CO₂ generated from each of these activities depends on the efficiency of the machine or process and the type of fossil fuel being used. For example, the combustion of diesel fuel produces approximately 74,000 kg CO₂/TJ while motor gasoline produces about 69,000 kg CO₂/TJ (IPCC 2006).

2.2.3. Nitrous Oxide

Nitrous oxide is a component in several transformations of the nitrogen cycle (Figure 3). The areas of N₂O production which are discussed below are manure storage, crop and pasture land, as well as indirect sources of N₂O losses.

2.2.3.1 Nitrification and denitrification

Nitrification is an aerobic process. Carbon dioxide serves as a C source for soil bacteria that oxidize ammonium ions (NH₄⁺) to obtain energy (Granli and Bøckman 1994). The two reactions that occur during nitrification are (Haynes 1986):



The bacteria that convert NH₄⁺ to nitrite (NO₂⁻) are able to produce N₂O by using NO₂⁻ as an electron acceptor if O₂ is not present (Poth and Focht 1985; Firestone and Davidson

1989) (Figure 4). Nitrous oxide is also formed when NO_2^- and intermediates between NH_4^+ and NO_2^- chemically decompose (Granli and Bøckman 1994).

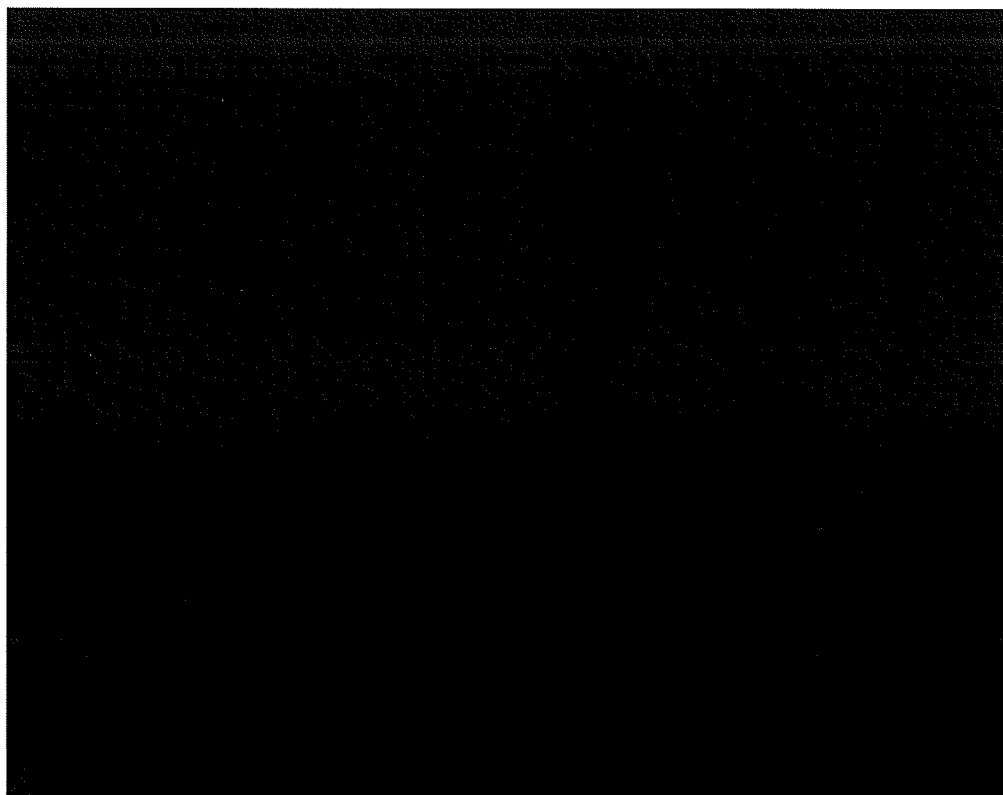
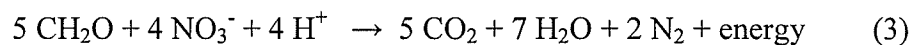


Figure 3. Depiction of the nitrogen cycle of agricultural soils and its relationship to N_2O production (Mosier et al. 1998b).

Denitrification is the anaerobic process which converts fixed N into atmospheric N_2 and produces energy. Microorganisms use NO_3^- in a series of reactions to extract energy out of organic compounds. The denitrification process can be summarized in the following equation:



Nitrous oxide is formed as one of the intermediates in the conversion of NO_3^- to N_2 (Granli and Bøckman 1994) (Figure 4).

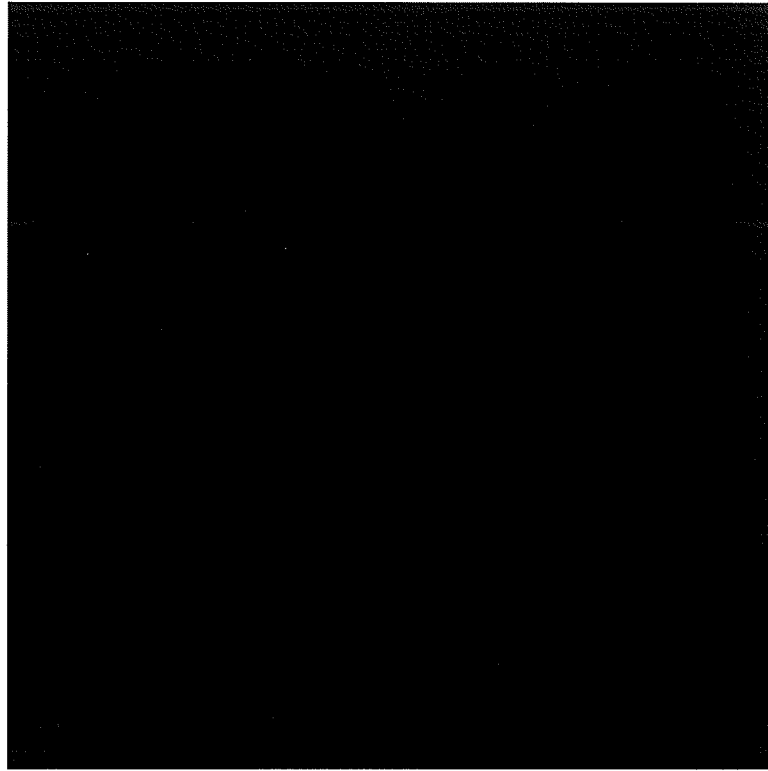


Figure 4. A simplified view of the nitrogen cycle in a cropping system (Lemke and Janzen 2007).

For denitrification to occur, the denitrifying bacteria must have access to an anaerobic environment and sufficient amounts of organic C as well as NO_3^- , NO_2^- or NO. The amount of N_2O produced relative to N_2 is affected by pH, moisture content, temperature, amount of organic material present, oxidant availability (NO_3^- or NO_2^-), O_2 availability and the activity of the N_2O reductase (Firestone and Davidson 1989). Maximum rates of denitrification occur at temperatures around 60 - 75°C, and can occur at temperatures as low as 0 – 5°C (Galbally 1989), although more recent work suggests that denitrifying microbes prefer temperatures between 5°C and 40°C (Lemke and Janzen 2007), with the ability to acclimatize to local conditions (Malhi et al. 1990). A doubling

in the rate of denitrification occurs for every 10°C increase in substrate temperature (Galbally 1989).

2.2.3.2 Nitrous oxide released from manure

Solid manure storage, the most common type of manure storage on beef farms (Statistics Canada 2003), is a large source of N₂O emissions (Janzen et al. 2008) (Figure 3). Solid manure is characterized by a solids content of more than 20%, with storage of manure in bedding packs or in piles (Statistics Canada 2003). Liquid hog manure is sometimes used to fertilize land used for pasturing beef cattle. In Canada, 86% of swine farms use liquid manure systems, which have a solids content of less than 5% (Statistics Canada 2003). Liquid manure can be stored in steel or concrete tanks or earthen basins (referred to as lagoons), with or without a cover. In storage, N₂O is produced from nitrification and denitrification of the manure ammonium N (Mosier et al. 1998b). Manure that contains large amounts of bedding material (such as straw), which decreases the bulk density of the stored manure, produces higher levels of N₂O than manure stored with lower levels of aeration. Materials such as straw make diffusion of O₂ into and N₂O out of the stored manure easier (Brown et al. 2000).

2.2.3.3 Soil release of nitrous oxide from crop and pasture land

The N₂O released from soil depends on climate, soil characteristics, cropping practices and the interactions between them. Although we cannot control climate, and have only limited control of soil characteristics (for example, we can control which land receives fertilizer or manure and moisture level to a limited extent, through irrigation or

drainage), we can control which cropping practices we employ (Beauchamp 1997). Different crop management strategies influence N_2O emissions to varying degrees. As well, some management practices have the potential to influence soil characteristics, and affect N_2O emissions. For example, applying fertilizer with the seed reduces the number of passes farm machinery makes over the field. This reduces soil compaction, thereby lowering N_2O emissions (Beauchamp 1997) by increasing the O_2 content of the soil.

The processes of nitrification and denitrification release N_2O from both fertilized and unfertilized soils (Figure 3). Soils that receive N inputs (either as synthetic fertilizer or as manure, which can be deposited directly onto the land by grazing animals or applied as a fertilizer) have increased rates of N_2O release because the N inputs provide greater quantities of substrate for the soil microbes which carry out the N_2O -producing processes. Although these N inputs represent a potential source of GHG's, they are necessary to maintain soil fertility and crop yields to meet the growing demand for food (Mosier et al. 1998b). Annually-cropped soils tend to release more N_2O than pasture or perennially cropped soils (Gregorich et al. 2005). The above-ground residue and roots of perennial crops have a slower rate of decay than annual crops. Perennial crops take up more nutrients from the soil because they have a longer period of active growth than annual crops. The presence of growing plants can decrease N_2O because the plants compete with the microbes for the available NO_3^- (Granli and Bøckman 1994). Permanent grassland reduces the rates of denitrification and nitrification by lowering the amount of available soil water (Grant et al. 2004). As well, less fertilizer is usually applied to lands under perennial cover than to lands in annual cropping systems.

Land cropped to legumes (either annual or perennials) is a significant source of N_2O (Figure 3). The atmospheric N_2 that is fixed by legumes is available to soil microbes, increasing the available substrate for N_2O production from nitrification and denitrification (Gregorich et al. 2005). Earlier studies suggested that the *Rhizobia* in legume root nodules are a source of N_2O as they are capable of the denitrification process (O'Hara and Daniel 1985), but more recent studies have shown that the denitrification potential of *Rhizobia* is minimal compared to other microorganisms (Garcia-Plazaola et al. 1993) and that *Rhizobia* populations are too small to affect the rate of denitrification in soils (Breitenbeck and Bremner 1989). A review by Rochette and Janzen (2005) concluded that "the significance of N_2O emission from legumes during N fixation remains uncertain and unproven", but that decay of N-rich residues from legumes could lead to appreciable N_2O emissions.

The breakdown of all crop residues not removed from the field also contributes available N to the soil. Because the N in crop residue represents an addition of N to the soil, nitrification and denitrification are stimulated, increasing N_2O production (Mosier et al. 1998b).

Nitrous oxide can be released from soils during spring thaw and freeze/thaw events in winter and spring. During these events, N can be released (Wang and Bettany 1993) and the saturation of the soil provides optimum conditions for denitrification and N_2O production (Gregorich et al. 2005).

2.2.3.4 Indirect sources of nitrous oxide

Inorganic N in the soil from fertilizer, manure or decomposition of organic matter can be lost via volatilization of NH_3 or leaching of NO_3^- . Volatile NH_3 lost from synthetic fertilizer, crop residues or manure can be later re-deposited as NH_4^+ in soil. When nitrified, this N can then lead to N_2O emissions. Leaching occurs when NO_3^- moves downwards in the soil via soil macropores from the N source (synthetic fertilizer, crop residues, manure or decomposed organic matter) to nearby ground and surface waters. Runoff occurs when NO_3^- is carried away in runoff water. Nitrification and denitrification then occur in the groundwater below the N source or in riparian areas of surface waters, resulting in N_2O emissions from the original N source (IPCC 2006). The N_2O losses from volatilized or leached N are sometimes called ‘indirect emissions’, because they occur away from the farmlands, but originate from N on those lands.

2.3. FACTORS INFLUENCING THE AMOUNT OF GREENHOUSE GASES PRODUCED OR REMOVED IN A BEEF PRODUCTION SYSTEM

2.3.1. Factors related to feed/diet characteristics

Feeding strategies or diets that influence carbohydrate fermentation and, therefore, the ratio of VFA produced (most specifically the ratio of acetate to propionate) will impact the GHG's produced by cattle. More CH_4 is generated when higher proportions of acetate are produced. The ratio of VFA production is affected by diet type (high forage or high grain content) and additives, such as fats or ionophores, that may be included in the diet (Johnson and Johnson 1995).

Cattle receiving diets with a high forage content tend to produce more CH_4 than animals on low forage diets (Blaxter and Wainman 1964; Moss et al. 2000). Depending

on forage quality, as much as 12% GEI can be lost as CH₄ on forage diets (Johnson and Johnson 1995). This increase in CH₄ production is associated with fermentation of the fibre in plant cell walls, resulting in a higher ratio of acetate to propionate. The hydrolyzed breakdown products of cell walls are easier for rumen methanogens to use than the breakdown products from the fermentation of starch found in grains, contributing to the higher rates of CH₄ released from cattle on high forage diets (Johnson and Johnson 1995). The amount of GEI lost as CH₄ on *ad libitum* intake diets that contain high levels of grain (more than 90%) can be as low as 2 – 3% (Johnson and Johnson 1995). However, including grain in maintenance level diets can increase CH₄ production (McAllister et al. 1996). Cows fed hay and barley have higher rumen ciliate populations than cows fed only hay (Bonhomme et al. 1990). As a symbiotic relationship exists between the ciliates and methanogens (Finlay et al. 1994), CH₄ production increases (McAllister et al. 1996).

Ionophores make rumen conditions unfavourable for the growth of microorganisms that generate substrate for the methanogens to use during CH₄ formation (McAllister et al. 1996). The amount of propionic acid produced increases (decreasing the acetate to propionate ratio) (Johnson and Johnson 1995; McGinn et al. 2004) and feed intake declines when ionophores are incorporated into high grain cattle diets (Guan et al. 2006). Methane production can be reduced by as much as 25% by feeding ionophores (Johnson and Johnson 1995), but several researchers have found that rumen microbes are able to adapt to ionophores, making them a useful tool for lowering CH₄ production for only a short time (Johnson and Johnson 1995; McAllister et al. 1996). A study by Guan et al. (2006) found that ionophores suppressed CH₄ production by 27% for only the first

two weeks in animals fed a high concentrate diet. These same authors observed a 30% reduction in CH₄ production for the initial four weeks when ionophores were included in low concentrate diets.

Methanogenesis can be reduced by including long-chain polyunsaturated fatty acids in the diet. These fatty acids are toxic to methanogens, protozoa and gram-positive cellulolytic bacteria, thereby impeding fibre digestion and lowering the production of butyrate and acetate. The gram-negative bacteria that produce propionate are not inhibited by fatty acids, so levels of propionate are increased, thereby reducing CH₄ production (McAllister et al. 1996). Further, hydrogen preferentially uses unsaturated fatty acids over CO₂ as an electron acceptor, and in their presence, less CH₄ is produced (Czerkawski et al. 1966). The unsaturated fatty acids that do not receive hydrogen accumulate in the rumen, reducing the capacity of rumen microbes to break down fibre (National Research Council (NRC) 2001), resulting in lower fibre digestibility and less CH₄ produced (McGinn et al. 2004). However, systems analysis is still needed to determine if there is a net decrease in CH₄ production, as incorporating fats into cattle diets can reduce average daily gain (ADG) (Boadi et al. 2004a), increase age to market and therefore, potentially lead to a net increase in CH₄ production.

The rate of passage of forages through the rumen also influences CH₄ production. McAllister et al. (1996) have speculated that poor quality forages are more fibrous, take longer to be digested and produce more CH₄ than good quality forages. Rumen fermentation of legumes generally produces less CH₄ than fermentation of grasses (Varga et al. 1985) because legumes have a faster rate of passage through the digestive system than grasses (Donefer et al. 1960; Demeyer and Van Nevel 1975), lower fibre levels

compared to grasses, and tend to shift VFA production towards increased propionate concentrations (Demeyer and Van Nevel 1975). Cows grazing alfalfa-grass pasture lost approximately 7.1% GEI as CH₄, compared to losses of 9.5% GEI from cows grazing straight grass pasture (McCaughey et al. 1999).

Increasing the level of feed intake increases total CH₄ production, but decreases CH₄ produced per unit of feed consumed (Blaxter 1967). Higher levels of intake increase the passage rate through the rumen, providing less time for rumen microbes to access the feed, reducing the level of fermentation and the amount of CH₄ produced (Mathison et al. 1998). Restricted feed intake has been shown to increase the amount of CH₄ lost as a % of GEI (Whitelaw et al. 1984). More recently, however, no difference between restricted and *ad libitum* diets has been observed in the % of GEI lost as CH₄ (Boadi and Wittenberg 2002).

2.3.2. Animal factors

The physical size of a ruminant animal affects the size and capacity of its rumen. The larger the rumen, the longer feed stays in the rumen, increasing CH₄ production (Boadi and Wittenberg 2002; Galbraith et al. 1998). The larger gut capacity of Holstein-Friesian x Harian cross cattle than Holstein-Friesian cattle resulted in the former having higher energy losses as CH₄ (Boadi and Wittenberg 2002). Methane production of bison (*Bison bison*), wapiti (*Cervus elaphus*), and white-tail deer (*Odocoileus virginianus*) when fed lucerne pellets was 6.6, 5.2 and 3.3% GEI, respectively (Galbraith et al. 1998). Methane production was lower in animals with a smaller rumen due to a decreased retention time of feed in the rumen.

Several studies have demonstrated that there is considerable animal-to-animal variation in CH₄ production. A study by Boadi and Wittenberg (2002) has demonstrated that animal-to-animal variation has more effect on CH₄ production than breed does. Their study found no difference in CH₄ production at either of two feeding levels (*ad-libitum* or a restricted diet) between beef and dairy heifers. Similarly, Pinares-Patiño et al. (2003) found that in cows grazing different maturities of timothy, 54-70% of the variation in daily CH₄ production was due to variation between individual animals.

It is well documented that differences exist among cattle in the level of feed intake required for the same production (Koch et al. 1963). This trait, more recently described as residual or net feed intake (RFI), is the difference between actual feed intake and the expected requirements for maintenance and production (Hegarty et al. 2007). An Australian study demonstrated that Angus steers with low RFI consumed less feed, had the same ADG, but produced 24% less CH₄ per unit of ADG compared to steers with high RFI (Hegarty et al. 2007). These authors conclude that selection of cattle based on the RFI offers an avenue through which CH₄ emissions may be reduced without compromising productivity, particularly when consuming low digestibility diets.

2.3.3. Factors based on soil, climate and management

Soil can be a source or sink of GHG's, depending on soil characteristics, management and climate. Changes in any of these parameters can lead to either an increase or decrease in emissions of CO₂, CH₄ and N₂O (Gregorich et al. 2005).

Soil uptake of atmospheric C is affected by soil N content. Because both C and N are constituents of soil organic matter, soil C content cannot increase unless there is a

simultaneous increase in soil N levels (Janzen et al. 2003). Management practices such as intensive tillage and conversion of native land to agricultural ecosystems can increase atmospheric C levels; practices such as zero-tillage, improved grazing management and integrated nutrient management (involving prudent use of fertilizers) increase soil C content (Lal 2004a). For example, a US study using full carbon cycle accounting, estimated that an average monoculture corn system released 253 kg C/ha/year, while the same system, after conversion to zero-tillage, removed 215 kg C/ha/year (West and Marland 2002).

The water content of a soil determines how much O₂ is in the soil pores. High soil water content is conducive to N₂O formation, as the microbial processes require both water and the anaerobic conditions created by high soil water content (Granli and Bøckman 1994). As such, management practices that increase soil water content, such as irrigation and, to a lesser extent, summer fallow, can increase N₂O emissions. Alternatively, the aerobic conditions associated with low soil water content promote CH₄ uptake by the soil, as soil methanotrophs use CH₄ as a source of both C and energy (Gregorich et al. 2005).

Increasing levels of available N (for example, through practices such as fertilization, manure application and biological fixation) increases the N₂O forming potential of the soil (Granli and Bøckman 1994). Production of N₂O can be increased by an average of 13 times in soil with N additions as compared to soil without any N additions (Ambus 1998). As well, large N inputs into the soil increase net CH₄ emissions by reducing the ability of the soil to uptake atmospheric CH₄ (Gregorich et al. 2005).

Soil texture and structure also influence the rate of production and consumption of GHG's. Clay soils, with their high affinity for water and small pore space, can produce large amounts of N_2O . However, because of the high soil density, the N_2O moves slowly toward the soil surface, and is prone to being reduced to N_2 . Conversely, coarse-textured soils have a low rate of N_2O production, but the quick rate of diffusion to the surface means that nearly all of the N_2O produced will reach the soil surface (Granli and Bøckman 1994). In moist areas, the higher soil density and water levels resulting from zero-tillage can increase N_2O emissions (Lemke and Janzen 2007). However, in drier regions, such as the Canadian prairies, zero-tillage can reduce N_2O emissions. Compaction and degradation of soil structure can lead to increased N_2O production and reduced soil consumption of CH_4 by reducing the O_2 content of the soil (Hansen et al. 1993).

2.3.4. Manure factors

Greenhouse gas emissions from livestock manure are affected by the composition of the manure. The total solids and ammonia (NH_3) in the manure affect CH_4 and N_2O release during manure storage. High levels of NH_3 (found in manure with high total solids content) inhibit methanogens, reducing the amount of CH_4 released (Massé et al. 2003). High NH_4^+ content also reduces the amount of N_2O formed (Brown et al. 2000), because it inhibits nitrification, preventing a build-up of NO_3^- , a substrate for N_2O formation. A review by Gregorich et al. (2005) showed that varying amounts of GHG emissions were released from corn fields fertilized with manure. Most of the variation among the studies was due to differences in soil type, climate and manure composition.

Manure consistency also influences GHG emissions, as liquid manure produces more GHG than solid manure. For example, in a recent review, Gregorich et al. (2005) found that liquid manure emitted approximately 2.8 kg N₂O-N/ha/year whereas solid manure released only about 0.99 kg N₂O-N/ha/year. The high soil moisture, lower O₂ availability and large amount of available C that occur when liquid manure is land applied are conducive to denitrification. In contrast, when solid manure is land applied, most of the N is tied up in organic compounds, and is not available for N₂O forming processes. The slow rate of mineralization of N in solid manure allows plants to take up the N as it becomes available, with little mineral N available for denitrification (Gregorich et al. 2005).

Manure storage conditions also affect how much CH₄ is produced. Higher air temperatures have been shown to slightly increase the rate of CH₄ loss from manure storage (Massé et al. 2003). Manure temperatures have also been found to be strongly related to the rate of CH₄ release from manure storage (Park et al. 2006). Total CH₄ emissions increase as storage time lengthens (Massé et al. 2003) and the volume of manure in storage increases (Park et al. 2006).

2.4. POSSIBLE METHODS OF REDUCING GHG EMISSIONS FROM A BEEF PRODUCTION SYSTEM

Table 4 gives a brief description of potential mitigation strategies. The table is by no means a complete compilation of possible mitigation practices, but it illustrates the range of options available to producers.

Table 4. Mitigation strategies for reducing GHG emissions from beef production systems.

Mitigation Strategy	Explanation/Concerns
<i>Alter consumption patterns</i>	Most people in the industrial world have excessive consumption patterns; reducing food consumption would reduce animal numbers and, hence, GHG emissions (Clemens and Ahlgrimm 2001)
<i>Reduction of Enteric CH₄</i>	
Increase animal size	Larger animals would yield more meat. This is possible from a production standpoint, but processing methods would have to change so consumers would not have to buy large cuts of meat.
Grazing management	
i) use legumes in pasture	Methane losses from cattle are lower (% of GEI) when alfalfa is included in the pasture composition (McCaughey et al. 1999)
ii) rotational grazing	Rotational grazing maximizes forage quality and quantity. Increased production efficiency = fewer emissions per unit of product (Boadi et al. 2004b)
iii) tannin-containing legumes	Tannins reduce CH ₄ production (McMahon et al. 2000; Rochfort et al. 2008), although they can depress animal performance if fed at levels exceeding 40-50 g/kg DM (McMahon et al. 2000).
Increase animal performance	
i) breed high producing animals and cull low-producers	Absolute CH ₄ emissions per animal increase, but emissions per unit product decrease. More product/animal = fewer animals needed (Clemens and Ahlgrimm 2001). In an Irish study, Casey and Holden (2005) showed decreases in GHG emissions of 14-18% by using more efficient cows, 14 – 26% by eliminating non-milking cows and 28-33% by utilizing both management strategies.
ii) breed for animals with a lower RFI	As RFI is moderately heritable (Robinson and Oddy 2004), selecting for animals with a low RFI will produce animals that require less feed to reach the same level of production (such as ADG) than less efficient cattle (Hegarty et al. 2007).
iii) artificial insemination	Use of artificial insemination could more rapidly increase genetic potential, increasing production efficiency and could potentially reduce the size of pedigree herds (Clemens and Ahlgrimm 2001)
iv) use of hormone	Hormone treatments can increase animal productivity,

treatments and anabolic steroids to increase productivity

reducing the number of cattle needed (Clemens and Ahlgrimm 2001).

Bovine somatotropin is a hormone used to increase milk production and lengthen lactation, which could lead to reductions in animal herd sizes and decreased CH₄. But hormone residues might remain in the milk (Clemens and Ahlgrimm 2001).

Anabolic steroids lead to weight increase, better feed utilization and lean tissue accretion. Increases in production performance are in the range of 5 – 10% (Clemens and Ahlgrimm 2001).

v) improved balance of dietary CP and energy

A correct balance of CP and energy in the diet can optimize performance and growth. (NRC 1996).

Alter microbial populations in the rumen to change fermentation pathways

Rumen microbial populations can be altered by improving diet digestibility (Clemens and Ahlgrimm 2001) or using ionophores or chemicals to inhibit methanogenesis (Boadi et al. 2004b). However, the inhibition is short-lived (Guan et al. 2006).

Plant extracts, such as garlic oil, are currently being examined for potential use as methanogen inhibitors (McAllister and Newbold 2008).

Altering fermentation patterns such that lower proportions of acetate are produced reduces CH₄ production (Johnson and Johnson 1995).

Reduction of CH₄ and N₂O from Animal Excreta

Decrease the amount of N excreted

Fewer animals = less N excreted (Clemens and Ahlgrimm 2001)

Feed proteins with low rumen degradability to achieve better protein utilization and less N excretion (Clemens and Ahlgrimm 2001)

Use nitrification inhibitors when land applying manure

Prevent the applied NH₄⁺ from being converted to N₂O (Clemens and Ahlgrimm 2001).

Currently, research is being conducted to examine the effect on N₂O emissions of supplementing grazing animals with a nitrification inhibitor. The inhibitor is excreted in unaltered form, and has the potential to reduce N₂O emissions from manure deposited on pasture (de Klein and Eckard 2008).

Incorporate manure into soil after land application

Manure incorporation soon after land application reduces losses of NH₃ to the atmosphere (Granli and Bøckman 1994).

Reduction of N₂O from Soils and Crop Production

Match N demands of crop to N application	<p>If the ideal amount of N for the crop type is applied, then plant N use will be most efficient.</p> <p>Less excess N applied to soil = lower N₂O emissions (Cole et al. 1997)</p> <p>Timing of fertilizer/manure application should be synchronized with crop N needs, reducing emissions (Granli and Bøckman 1994).</p>
Irrigate only as much as necessary	<p>Providing sufficient, but not excessive, water for crop growth, and ensuring even water coverage over the field prevents over-wetting the soil (Granli and Bøckman 1994).</p>

Reduction of CO₂

Increase soil C sequestration	<p>Soil C can be increased by (Lal 2004b):</p> <ul style="list-style-type: none"> - returning more organic matter/crop residue to the soil - using diversified crop rotations - applying fertilizers and chemicals judiciously - reducing tillage intensity
Minimizing fuel use	<p>Reducing tillage intensity and the amount of synthetic fertilizer can minimize fuel use on-farm (Janzen et al. 2006).</p>

The global potential for GHG mitigation by 2030 is estimated at approximately 5500 – 6000 Mt CO₂e/year (Smith et al. 2007). Of this, about 89% is via C sequestration, 9% from mitigation of CH₄, and 2% from mitigation of soil N₂O emissions. The potential for GHG mitigation by 2030 in North America is approximated at 200 – 600 Mt CO₂e/year.

2.5. BENEFITS OF REDUCING GHG EMISSIONS

2.5.1. International commitments

The Kyoto Protocol is an agreement formed in 1997 between members of the United Nations Framework Convention on Climate Change. In the Kyoto Protocol, 38

member countries agreed to reduce average annual emissions to a certain level below 1990 emissions between 2008 and 2012. The size of the reduction varied for each country – Canada committed to reducing emissions to 6% below 1990 levels (Environment Canada 2007).

Since 1997, Canada's GHG emissions have increased to 35% above our original Kyoto Protocol targets (Figure 5). To reach the Kyoto target, Canada would now have to reduce annual emissions by roughly one third (Environment Canada 2007). As a consequence of Canada not being able to achieve their Kyoto Protocol commitments, the Canadian federal government has set a national goal of reducing absolute emissions by 20% below 2006 levels by 2020 (Environment Canada 2008).

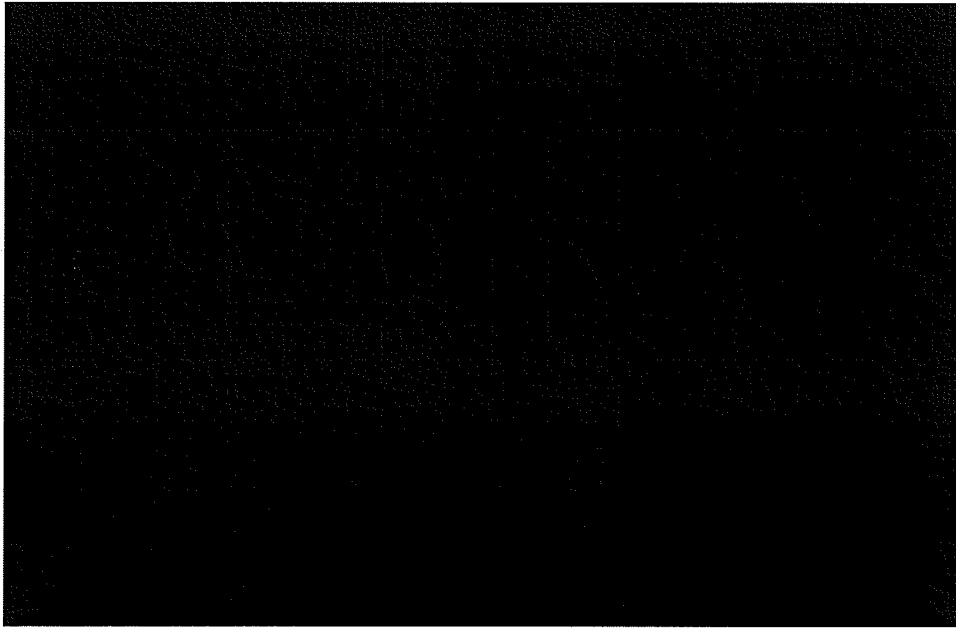


Figure 5. The current level of emissions compared to Canada's Kyoto Protocol targets (Environment Canada 2007).

Most countries are presently using methodologies developed by the IPCC to estimate their GHG emissions. Countries can choose to follow either of two Tiers for inventorying their GHG emissions. Tier 1 is used where detailed information is not

available. Tier 1 GHG estimates are based on a default emission factor and animal populations, fuel use, N inputs to the land, area of managed/drained soils and the amount of manure deposited onto pasture by grazing animals (IPCC 2006). Tier 2 estimates require much more detailed information about agricultural activity and animal characteristics, and emissions can be calculated for more specific categories. These methodologies provide algorithms and suggest emission factors and coefficients applicable to a wide range of environmental and management conditions. The IPCC encourages individual countries to use country-specific data where possible to modify the algorithms, emission factors and coefficients to more accurately depict the environmental and management conditions in that country and give a more accurate estimate of GHG emissions.

2.5.2. Increased productivity and efficiency

Greenhouse gas mitigation practices frequently have side effects that increase productivity. These mitigation strategies benefit the environment and the farmer. Productivity improvements can occur from decreases in emissions of CH₄ and N₂O and from increased C sequestration.

Most mitigation practices that reduce enteric CH₄ emissions will also increase animal performance (Boadi et al. 2004b). Methane represents a loss of dietary energy (Johnson and Johnson 1995), and as such, any strategy that reduces CH₄ production increases the energy available for meat or milk production, thereby improving production efficiency. However, some practices that reduce CH₄ emissions (such as increasing the proportion of grain in the diets) need to be systematically evaluated to make sure that the

increase in emissions arising from increased crop production does not counteract the reduction in CH₄ emissions achieved by feeding grain to the animal (Boadi et al. 2004b).

The efficiency of N fertilizer use can be improved by practices aimed at reducing N₂O emissions. Increased N fertilizer efficiency (achieved by soil testing and applying fertilizer to meet crop needs) can reduce the amount of N fertilizer demanded by crop growers, thereby lowering the emissions associated with use of fertilizers (Cole et al. 1997; Smith et al. 2007).

Management practices that promote accumulation of soil organic matter will improve crop production through enhanced soil fertility, improved soil structure and reduced erosion. Practices that sequester soil C have the potential to improve long term crop production and global food security. Crop production and water use can be improved by using zero tillage (Smith et al. 2007). In regions where water may be a limiting factor, crop yields, and, therefore, soil organic matter, can be increased due to the increased moisture content of soils under zero tillage.

2.6. SYSTEMS APPROACH TO ESTIMATING GHG EMISSIONS FROM A BEEF PRODUCTION SYSTEM

2.6.1. Ecological importance of looking at the entire beef production system – a ‘systems’ analysis of GHG emissions

Land under agricultural production carries out services beyond the production of marketable commodities. These ‘ecosystem services’, often unseen, include clean air and water, a source of habitat and food for songbirds, beneficial insects and wildlife, as well as C storage, pollination and disease suppression (Robertson and Swinton 2005). These services are essential to ecosystem health and long-term agricultural sustainability. To

maintain these non-commodity ecosystem services, it is imperative that we understand the system; that is, how each component of the ecosystem affects the other components. A barrier to agricultural land management that promotes development of these additional services is our lack of a “systems understanding” (Robertson and Swinton 2005). Research into this area is warranted because development and preservation of these additional services have the potential to decrease GHG emissions while ensuring that agriculture is sustainable long into the future. Many of the ecosystem services are linked. Lal (2004c), for example, demonstrates the synergy that exists in the soils as soil C storage removes CO₂ from the atmosphere while at the same time increasing soil fertility and conservation, soil invertebrate diversity and plant water-use efficiency. These benefits, in addition to reducing GHG's, demonstrate the importance of finding ways of managing our finite agricultural land resources, responsibly and sustainably, through systems research.

2.6.2. Examples of studies where the ‘systems’ approach has been taken

Scientists are beginning to look at GHG's from a systems approach. For example, Allard et al. (2007) looked at CO₂, N₂O and CH₄ emissions from a semi-natural grassland site continuously grazed by cattle in two large, separate paddocks. One paddock received N fertilizer, while the other did not and was, therefore, grazed at half the stocking density as the fertilized paddock. Net ecosystem exchange was determined, and included the C lost from the system in each of the following: CH₄, live-weight gain of the cattle, soil C storage, and respired CO₂. Soil N₂O and CH₄ emissions were measured, as were CH₄ emissions from enteric fermentation.

Researchers in Denmark and Germany (Olesen et al. 2006) have modelled GHG emissions from conventional and organic dairy farms in Europe. This second example of a systems approach to GHG accounting modelled CH₄ and N₂O emissions from on-farm production of saleable commodities, as well as the CO₂, CH₄ and N₂O emissions associated with the imported energy, feedstuffs, fertilizers and other farm inputs. Emissions from farm products were not included after they left the farm.

Soussana et al. (2007) studied the full GHG budget of nine European grassland sites. The study included net ecosystem exchange of CO₂, soil N₂O emissions and CH₄ release from enteric fermentation. The researchers also included C in biomass removed from the site and C added to the site through manure/slurry application in their systems analysis.

These studies demonstrated the value of a systems approach, but also indicated that more research is needed, particularly in the area of modeling and further, that even in a systems analysis, large amounts of uncertainty still exist.

2.6.3. The systems studied in this thesis

The first manuscript of this thesis compares modeled emissions estimates of different management practices in a typical western Canadian beef production system. The beef production system includes a cow/calf operation that raises the calves to market weight. The same system was implemented in four different Prairie regions, with site specific details being used where available. The system includes the cattle, the feed the animals consume, the land needed to produce that feed, the manure the animals produce, the fertilizer applied to the land, and the protein contained in the animals and

unconsumed plant material. The model, described in detail in Manuscript I, calculates CH₄ emissions from enteric fermentation and manure, soil oxidation of CH₄, N₂O release from soils and manure, CO₂ emissions from energy use and soil C change. The system described above was entered into the model and emissions (both net farm emissions and emissions per unit of protein exported off farm) for various management scenarios were estimated and compared.

The second manuscript aims to incorporate measured values into a systems analysis. A grassland pasture system which has been divided into three fertility treatments was the basis for the system in Manuscript II of this thesis. The three fertility treatments were Control (targeted zero liquid hog manure), Full (targeted 242 kg/ha of total N via liquid hog manure in a single spring application) and Split (targeted 121 kg/ha of total N via liquid hog manure in each of two separate applications, one in the spring and the second in the fall). The pasture system used in Manuscript II includes the liquid hog manure applied to the pasture, the cattle grazing the pasture, the pasture soil, the manure the cattle produce, and the protein that leaves the pasture as weight gain in the steers. The model utilized is the same as that described in Manuscript I. Differences in the modeled and actual measured emissions are discussed.

2.7. THE POTENTIAL OF MODELS TO ESTIMATE GHG EMISSIONS FROM BEEF PRODUCTION SYSTEMS

2.7.1. Roles of models

Models can help us to estimate emissions from a whole farming ecosystem. They can combine knowledge of each area (soil, crops and animals) and link the flows of nutrients from one area to another to provide us with an accurate estimate of the

emissions from the system. The use of models is the only way to quantify net emissions through a detailed examination of the interactions and feedbacks that are the essence of systems (Janzen et al. 2006). Because of the nature of GHG emissions, models may also be the only efficient way to estimate regional GHG fluxes. Determining regional GHG emissions is inaccurate due to the large spatial and temporal variation in sources and sinks of GHG's (Gregorich et al. 2005). A large number of measurements would be needed to accurately estimate the GHG emissions from a region because subtle changes in local factors such as topography and soil type can have large impacts on the size of the emission. Inaccurate estimates of regional GHG emissions could potentially lead to incorrect evaluation of the mitigation potential of management strategies.

Models also provide us with the ability to include time in our systems analyses. The amount of time that has elapsed since a mitigation practice was put in place affects the amount of current benefit derived from that practice (Lemke and Janzen 2007). For example, zero tillage is used to sequester C, but soil has a limited capacity to hold C, and once the soil C reaches a new equilibrium, little or no further C is sequestered by that soil. The amount of C that soil is currently gaining influences the effectiveness of zero tillage as a mitigation strategy. Time must be considered in a systems analysis because the current condition of the system is affected by what has happened earlier (Janzen et al. 2008).

Once a systems-based model has been developed, it forces the user to focus on the entire system as effects of management on individual components cannot be analyzed singly. Output from a systems-based model is a more representative estimate of real levels of emissions (Janzen et al. 2006). Models can help to identify gaps in our

understanding of how systems work and where future research efforts and resources should be allocated. This makes models necessary, because, as Robertson and Swinton (2005) said, “in very few agricultural systems do we have a systems-level understanding of important properties and processes”.

A model that accounts for all flows of nutrients and GHG emissions in a farm production system can be used to assess which mitigation options may or may not reduce net emissions from that particular system (Janzen et al. 2006; Olesen et al. 2006). It is imperative that we know the effects that a mitigation option has on net GHG emissions before it is recommended to and implemented by farmers.

The fundamental role of models is not to generate a perfectly correct answer, but to force our GHG analysis to encompass the whole ecosystem (Janzen et al. 2008). They also show us where we need to invest our resources to gain a more complete understanding of the complex interactions within farming systems. Algorithms can be added to existing models as more knowledge becomes available, so that the models are always evolving and improving our estimates of GHG emissions from whole systems.

2.7.2. Limitations of models

But models are not without weaknesses. In some cases, detailed model inputs are needed to get meaningful and accurate model estimates. These detailed inputs can be difficult to get without extensive and costly research prior to model use, limiting the audience that is able to use the model. In most instances where data for model inputs is not readily available, existing data from other regions is used. Although this is a common and unavoidable practice, it may produce inaccurate estimates of GHG

emissions. In Olesen et al. (2006), for example, an NH_3 emission factor was needed, and as no unified European emission factor with enough detail existed, a value that was specific for conditions in Denmark was used. The authors state that “the NH_3 emission factors from the housing and manure storages may have been underestimated for Southern European conditions, where temperatures are higher”.

Models are often designed for one production system, with a particular set of parameters in mind. Consequently, the model may not accurately predict emissions from the system if important variables, such as diet type or dry matter intake, are altered. For example, Kebreab et al. (2006) explains that some models give unrealistically high emissions estimates when the dry matter intake of cattle increases above that for which the model was created.

There are several models available that calculate emissions of one or two gases from a portion of an agricultural system. For example, the Denitrification-Decomposition (DNDC) model predicts CO_2 , CH_4 , NH_3 and N_2O emissions from agricultural soil (Li 2000). It is capable of calculating N_2O emissions from land applied or injected animal manures, but cannot calculate N_2O emissions from manure deposited by grazing animals or manure storage (Brown et al. 2001). Although the CENTURY model is more detailed, it still does not evaluate a whole beef production system. The CENTURY model was designed to describe C and N dynamics in grasslands, crops and forests. With the CENTURY model several farming practices, such as seeding, tillage, grazing, fertilizer application and organic matter addition, can be evaluated (Smith et al. 2001). There are several examples of models that have been designed in order to predict CH_4 emissions from enteric fermentation of cattle. Most of the original CH_4 prediction

models are based on feed characteristics and intake level, while the more recently developed models depend on a wide range of animal and dietary variables to produce the emissions estimates (Kebreab et al. 2006). Nonetheless, they examine only enteric emissions and cannot be used to estimate net GHG emissions associated with a given production system. Clearly, there is a need for the development of more holistic models that may be used to quantify net GHG emissions.

2.8. SUMMARY

Global atmospheric concentrations of CO₂, CH₄ and N₂O are increasing. Sixty percent of global anthropogenic N₂O emissions and 50% of global anthropogenic CH₄ emissions are attributed to agriculture. Canada's agriculture industry is a significant contributor to the country's total GHG emissions.

Greenhouse gas emissions from beef production systems are influenced by several factors, some of which include animal size, soil characteristics and manure storage type. Each of these factors is in turn affected by multiple processes and management decisions. Because of the inherent complexity in all farming systems, and the feedbacks and interactions between the carbon and nitrogen cycles, calculating net GHG emissions is not easy.

There are management options available that producers can adopt to decrease the GHG emissions from their farms. Before any management practices can be advocated for reducing GHG's, a thorough systems evaluation needs to be done to ensure that a reduction in one facet of the farm is not actually increasing net emissions. Some available mitigation practices involve increasing C sequestration and improving animal

production efficiency. Canada needs to find methods of reducing its GHG output in order to meet its emission-reduction commitment to the Kyoto Protocol.

Beef production systems must be looked at with a holistic approach. Models are needed to accomplish this. Many of the currently available models do not focus on an entire ecosystem; they estimate emissions of one or two GHG's, or they only provide accurate estimates for the specific system with which they were designed. There is a need for a model that incorporates the entire farming system, and that can be used to calculate emissions from different management strategies in different regions.

3.0 RESEARCH HYPOTHESES AND OBJECTIVES

3.1. HYPOTHESES

The identification of management strategies to reduce net whole-farm greenhouse gas (GHG) emissions can be achieved using systems-based approach. This strategy of examining multiple GHG's in a production system is superior to an examination of single emissions, either methane (CH_4) or nitrous oxide (N_2O), in that it gives a more accurate assessment of the effectiveness of management practices at reducing net farm GHG emissions. It is anticipated that those changes in management that affect the nitrogen applied to crop/forage land will cause the largest decrease in net emissions and in emissions per unit of protein exported. Further, the increased enteric CH_4 production that results from the practice of feeding a reduced quality forage to cattle will increase net emissions and emissions per unit protein. Including feed additives (such as ionophores) will have little effect on net whole-farm emissions or on net emissions per unit of protein exported. Some practices will exhibit a decrease in emissions of one GHG, while simultaneously resulting in a rise of one or more other GHG's. As there is more confidence in our knowledge of enteric CH_4 formation than soil nitrous oxide (N_2O) generation, the model's predicted estimates of CH_4 will more closely reflect the measured emissions values than the predicted N_2O estimates. Predicted estimates of N_2O will be greater than the measured values as a consequence of shortcomings associated with methodologies used to collect this data. Nonetheless, the observed trends in ranking the various treatments will be the same for the predicted emissions estimates as those from the measured estimates.

3.2. OBJECTIVES

The overall objective is to demonstrate the importance of quantifying GHG emissions from agricultural systems as entire ecosystems and to emphasize the importance of using a systems-based approach to reducing GHG emissions. Specific objectives include: (i) to demonstrate the importance of looking at the whole-farm GHG emissions (rather than only from individual components); (ii) to compare the effects of changes in management practices on the net whole-farm GHG emissions and productivity from a beef production system, (iii) to compare how net responses to selected practices vary among ecozones in western Canada, (iv) to evaluate the effectiveness of two proposed mitigation strategies using two systems-based approaches: a) predicted emissions derived from an IPCC-based model and (b) measured emissions derived from a multidisciplinary field study; and (v) to demonstrate, based on an evaluation of measured and predicted values, the key areas where knowledge is lacking and improvements are most urgent.

4.0 MANUSCRIPT I

Evaluating greenhouse gas mitigation practices in livestock systems: an illustration of a
whole-farm approach

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4.1 ABSTRACT

As agriculture contributes about 10% of Canada's greenhouse gas (GHG) emissions, reducing agricultural emissions would significantly decrease total Canadian GHG output. Evaluating mitigation practices is not always easy because of the complexity of farming systems in which one change may affect many processes and associated emissions. Our objective was to compare the effects of selected management practices on net whole-farm emissions from a beef production system, as estimated for hypothetical farms at four disparate locations in western Canada. We compared whole-farm emissions (Mg CO₂e) per unit of protein output (Mg) of 11 management systems (Table 6) for each farm using a model based, in part, on Intergovernmental Panel on Climate Change (IPCC) equations. Compared to the baseline management scenario, pasturing cattle on alfalfa-grass showed the largest decrease (0.39 to 0.70 Mg CO₂e/Mg protein) in emissions for all locations. Feeding lower quality forage over winter showed the greatest increase in emissions per unit protein on the southern Alberta (S.AB) and northern Alberta (N.AB) farms, with increases of 1.14 and 1.09 Mg CO₂e/Mg protein, respectively. Eliminating the fertilization of forages resulted in the largest increase (2.36 Mg CO₂e/Mg protein) in emissions per unit protein on the Saskatchewan (SK) farm, while reducing the fertilizer rate by half for all crops showed the largest increase (2.26 Mg CO₂e/Mg protein) on the Manitoba (MB) farm. The findings illustrate the importance of considering all GHG's simultaneously, and show that practices which best reduce emissions vary among locations. The findings also suggest merit in comparing emissions on the basis of CO₂e per unit of protein exported off-farm, rather than on the basis of total CO₂e or CO₂e per hectare.

Abbreviations: **ADG**, average daily gain; **AG_{residue N}**, nitrogen in above ground plant residue; **BG_{residue N}**, nitrogen in below ground plant residue; **Bo**, maximum methane producing capacity of manure; **C**, carbon; **CH₄**, methane; **CO₂**, carbon dioxide; **CO_{2e}**, carbon dioxide equivalents; **CP**, crude protein; **DM**, dry matter; **DMI**, dry matter intake; **EF**, emission factor; **EF_{eco}**, ecodistrict emission factor; **E_x**, energy requirement for fuel use or production of machinery and herbicides; **GEI**, gross energy intake; **GHG**, greenhouse gas; **MB**, Manitoba; **MCF**, methane conversion factor; **N**, nitrogen; **N.AB**, northern Alberta; **NE**, net energy; **N₂O**, nitrous oxide; **P**, phosphorus; **PI**, crude protein intake; **PR**, protein retention; **SK**, Saskatchewan; **S.AB**, southern Alberta; **SC**, soil carbon change coefficient; **TDN**, total digestible nutrients; **VS**, volatile solids; **Y_m**, percent gross energy intake lost as CH₄

Keywords: beef production, systems analysis, greenhouse gas, mitigation strategies, management practices

4.2 INTRODUCTION

Canada emits about 2% of the global greenhouse gases (GHG's) (Neitzert et al. 2007). Of these emissions, agricultural carbon dioxide (CO₂), methane (CH₄) and nitrous oxide (N₂O) account for approximately 10% (Janzen et al. 2008). Unlike those in other sectors, GHG's from agriculture are primarily N₂O and CH₄, in roughly equal proportion when expressed in units of CO₂ equivalents (Janzen et al. 2008). Agriculture produces 25 and 66% of Canada's CH₄ and N₂O emissions, respectively (Neitzert et al. 2007). Approximately two thirds of Canada's agricultural emissions are from livestock (enteric fermentation, manure management and pasture, range and paddock manure), with the rest coming from fertilizers (both manure and synthetic fertilizers), plant residue decomposition and cropping practices (such as summerfallow and irrigation) (Collas and Liang 2007). Since 1990, GHG emissions from enteric fermentation and manure management have increased by 39 and 23%, respectively, and N₂O emissions from manure management and agricultural soils have gone up by 32 and 10%, respectively (Collas and Liang 2007). This observed increase in Canadian GHG emissions has been attributed to rising livestock populations and an increased use of synthetic nitrogen (N) fertilizer on farms (Collas and Liang 2007). As agriculture is a significant contributor to the country's total GHG emissions, some of which are increasing, a reduction in the industry's GHG output would aid Canada in reaching its emission reduction targets.

All sources of GHG's within the farming system need to be considered when assessing farm GHG emissions. Complex interactions exist between the processes which produce each GHG (Figure 6). Examining only one GHG ignores these interactions (Robertson and Grace 2004; Gregorich et al. 2005; Schils et al. 2005), and the subsequent

effect that changes in management practices may have on net farm GHG emissions. Moreover, farmers make decisions based on their entire farm – they have to look at how changes in one component may affect viability of other areas. As a result, whole-farm mitigation practices will be more readily accepted and adopted by producers than those that only focus on a small portion of the production system.

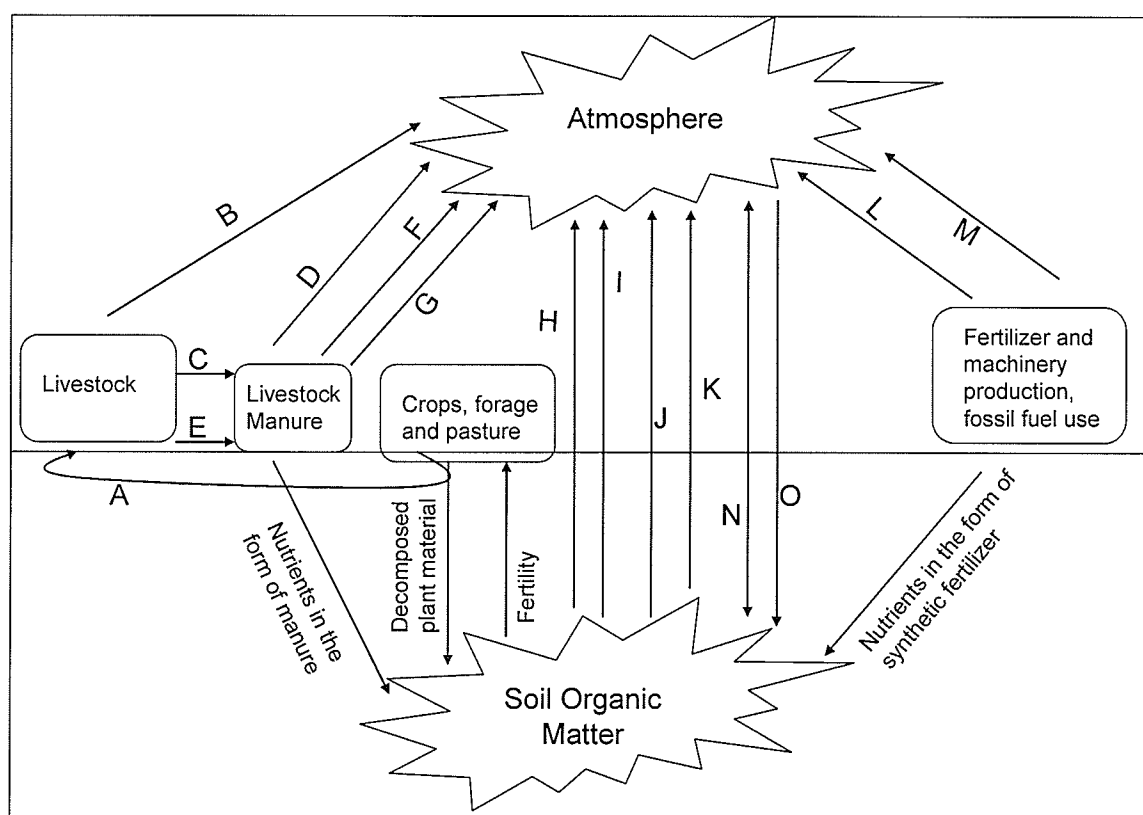


Figure 6. Diagram of the pools and flows contained within the model that was used in this study. Flows are as follows: A = gross energy intake of cattle; B = enteric CH_4 production by cattle; C = volatile solids excretion by cattle; D = CH_4 production from manure; E = nitrogen excretion by cattle; F = direct N_2O emissions from manure; G = indirect N_2O emissions from manure; H = direct N_2O emissions from land application of manure; I = indirect N_2O emissions from land application of manure; J = direct N_2O emissions from crop production; K = indirect N_2O emissions from crop production; L = fossil fuel derived CO_2 emissions from fuel usage and herbicide and machinery production; M = fossil fuel derived CO_2 emissions from fertilizer production; N = soil carbon uptake; O = CH_4 oxidation in agricultural soils.

This paper focuses specifically on beef production systems, as they are significant sources of GHG's, and hence, may represent important opportunities to reduce emissions. In 2005, Canadian beef cattle produced 21 Mt CO₂e in enteric CH₄ and 0.85 Mt CO₂e CH₄ from manure management (Collas and Liang 2007). These systems are sources of three principal GHG's (CO₂, CH₄, and N₂O), and can also be a sink for carbon (C). Our objectives were (i) to demonstrate the importance of looking at the whole-farm GHG emissions (rather than only from individual components); (ii) to compare the effects of changes in management practices on the net whole-farm GHG emissions and productivity from a beef production system; and, (iii) to compare how net responses to selected practices vary among potential sites in western Canada. The underlying goal was to demonstrate the importance of a systems-based approach to reducing GHG emissions, rather than merely to provide estimates of emissions from various practices. For this purpose, we used a simple spreadsheet model, although the findings may apply also to other systems-based modeling approaches.

4.3 MATERIALS AND METHODS

Four hypothetical farms, representing four diverse climatic and soil conditions on the Canadian prairies, were developed to provide the basis for evaluating GHG emissions. One farm was designed to represent each of the following locations: (i) the southwestern region of Manitoba (MB farm), focusing on the area surrounding Deloraine (ii) the south-central region of Saskatchewan (SK farm), based on Swift Current (iii) the south-central area of Alberta, centered around Lethbridge (S.AB farm) and (iv) the Peace River region of northern Alberta (N.AB farm). Site specific climate, yield, fertilizer, and

soil data were used to create the farms (Table 5). Each farm was designed to support the same number and type of cattle to allow comparisons among the farms. All farms included 100 hectares (ha) of barley, 230 ha of alfalfa-grass and 110 ha of pasture (0 ha fallow). The tillage system was assumed to be no-till, having changed from intensive tillage 15 years earlier. For livestock, the production year was split into three periods, with emissions from enteric fermentation and manure management calculated separately for each period. During Period 1 (136 days), from October 16 – February 28th, cattle were confined in corrals, the cows were pregnant but not lactating, calves from the previous spring were retained on-farm and fed forage-based diets (referred to as background calves), and replacement heifers were reared with the other calves. All animals were fed alfalfa-grass hay (11.9, 16.7, 7.1 kg DM/head/day for cows, bulls, background calves, respectively). There were 100 cows, 5 bulls, 43 steers and 42 heifers on farm in Period 1. At the beginning of Period 1, 10 animals were culled from the herd, and the previous year's replacement heifers joined the cow herd. The number of calves does not equal the number of cows, because of an assumed 15% calf loss annually. The background steers and heifers had an average daily gain (ADG) of 0.71 and 0.68 kg/head/day, respectively (Basarab et al. 2005). During Period 2 (92 days), from March 1st until May 31st, cattle were kept in corrals; cows calved and began lactating, replacement heifers were moved into a separate group and, part way through Period 2, background calves were moved to the feedlot. Milk production was assumed to be 7.9 kg/day, milk fat was 4.7% and milk protein content was 3.5% (Butson and Berg 1984). All cattle were fed alfalfa-grass hay (14.5, 16.7, 8.6, 8.4 kg DM/head/day for cows, bulls, replacement heifers and background calves, respectively) except for feedlot cattle, which

Table 5. Description of the hypothetical farms evaluated.

Farm Name	Soil Type*	Average Growing Season Precipitation [†] (mm)	Average Growing Season Potential Evapo-transpiration [†] (mm)	Average Annual Temperature [†] (°C)	Irrigation	Yield [‡] (kg/ha)		Fertilizer Application Rate [§] (kg/ha)					
						Barley	Forages	Barley		Alfalfa-grass		Grass	
								N	P	N	P	N	P
MB	Black Chernozem	369	613	2.9	No	3,155	3,163	90	39	26	37	112	30
SK	Dark Brown Chernozem	264	635	3.7	No	2,253	4,469	62	19	26	26	82	11
S.AB	Dark Brown Chernozem	277	653	5.5	Yes	4,581	10,302	112	45	30	56	191	45
N.AB	Gray Luvisol	285	544	-0.4	No	2,748	5,181	90	39	26	37	112	30

* Landscapes of Canada Working Group (2005)

[†] Marshall et al. (1999)[‡] Yield estimates were obtained from Manitoba Agriculture, Food and Rural Initiatives (2005); Saskatchewan Agriculture and Food (2007a, b); AAFRD (2007a, b, c). All yields were assumed to be dry weights. Yields of grass were the same as yields of alfalfa (Fairey 1991).[§] rate = $0.67 \times (\text{max-min}) + \text{min}$ of the range in AAFRD (2004)

received barley (8.5 kg DM/head/day) and barley straw (2.0 kg DM/head/day).

Replacement heifers had an ADG of 0.68 kg/day; background calves had the same ADG as in Period 1, until they reached the feedlot when ADG was increased to 1.59 kg/day for steers and 1.49 kg/day for heifers (Basarab et al. 2005). During Period 2 there were 100 cows, 10 replacement heifers, 5 bulls, 75 background calves (early in the period) and 75 feedlot cattle (later in the period). During Period 3, from June 1st to October 15th, cows, replacement heifers and bulls were rotationally grazed on a grass pasture (14.0, 9.8, 17.2 kg DM/head/day for cows, replacement heifers and bulls, respectively), while feedlot cattle remained confined and were fed barley (10.3 kg DM/head/day) and barley straw (2.0 kg DM/head/day). The cows were lactating, and cows and replacement heifers were bred. Milk production was assumed to be 5.8 kg/day, with milk fat and protein levels as before (Butson and Berg 1984). Calves were weaned in early October. The ADG of cattle in the feedlot and replacement heifers was the same as that in Period 2 (Basarab et al. 2005). In Period 3 there were 100 cows, 10 replacement heifers, 5 bulls and 75 feedlot cattle. During all periods the manure from cattle kept in corrals was stored in a deep bedding manure handling system (described in Intergovernmental Panel on Climate Change (IPCC) (2006)) and land applied in the fall.

Eleven management options were considered (Table 6); one baseline system, and 10 variations of the baseline. The net emissions of each gas, (Mg CO₂e), as well as emissions per Mg of protein exported off-farm for each practice were compared to those of the baseline management scenario for each of the farms.

Cattle feed requirements and rations were based on Cowbytes software (AAFRD 2003). Crop and forage yields were then used to determine the area of each crop needed

Table 6. Practices evaluated and some assumptions used in calculating emissions.

Designation	Description and input
1. Baseline	
2. Include fallow	7% of the total farm land area is converted to chemical fallow
3. 50% N application rate	N rate (synthetic fertilizer and cattle manure) reduced by half for all land; yields reduced by 25%*
4. No fertilizer to forages	No synthetic fertilizer or cattle manure applied to pasture or hay; yields reduced by 30%*
5. Split manure application	Change from applying all cattle manure in the fall to applying ½ of manure in fall; ½ of manure in spring
6. Spring manure application	Change from applying all cattle manure in fall to applying all manure in spring
7. Alfalfa-grass pasture	<p>Alfalfa-grass pasture used instead of a grass pasture</p> <ul style="list-style-type: none"> • Feed consumed by bulls, cows and replacement heifers: 16.0, 14.0 and 9.6 kg/head/day • Diet total digestible nutrients (TDN) and crude protein (CP): 63.8 and 17% for all grazing animals (AAFRD 2003) • % gross energy lost as CH₄ (Ym): 7.1% for bulls and cows (McCaughy et al. 1999); 4.5% for replacement heifers (McCaughy et al. 1997) • Model inputs for calculating above and below ground residue for pasture were the same as alfalfa-grass hay
8. Low quality forage	<p>Low quality forage (e.g. rain-affected hay) fed in Periods 1 and 2</p> <ul style="list-style-type: none"> • Period 1 bull and cow diet: 16.6 and 11.7 kg/head/day; background calf diet: 5 and 2 kg/head/day of hay and barley, respectively • Period 2 bull diet: 16.6 kg/head/day; cows: 8.9 and 5.2 kg/head/day hay and barley; replacement heifers: 5.3 and 2.8 kg/head/day hay and barley; background calves: 5.9 and 2.4 kg/head/day hay and barley; Period 1 diet TDN and CP: 53.1 and 7.8% for both bulls and cows; Period 1 CP: 9.1% for replacement heifers and background calves; Period 2 TDN: 53.1, 64.1, 63.3 and 61.6% for bulls, cows, replacement heifers, background calves; Period 2 CP: 7.8, 9.5, 9.4 and 9.1% for bulls, cows, replacement heifers and background calves (AAFRD 2003) • Ym: 6.9% for all cattle (Boadi and Wittenberg 2002)

- | | |
|----------------------------|---|
| 9. Lengthen grazing period | Lengthen grazing season by 136 days <ul style="list-style-type: none">• Period 1 feeding activity coefficient changed (IPCC 2006) |
| 10. Use fats | Use fats in finishing diets <ul style="list-style-type: none">• Dry matter intake (DMI) reduced by 14.3%; steer and heifer ADG reduced by 17% (Boadi et al. 2004a)• CH₄ emissions decreased by 20% (additive reduction factor) (Boadi et al. 2004a) |
| 11. Use ionophores | Use ionophores in finishing diets <ul style="list-style-type: none">• CH₄ emissions decreased by 27% for 14 days (additive reduction factor) (Guan et al. 2006)• DMI reduced by 10.4% (Guan et al. 2006) |
-

* Henry Janzen, personal communication, Agriculture and Agri-Food Canada, Lethbridge, AB

to meet the feed requirements of the cattle.

When management options affected yields, the land area needed to produce individual feedstuffs was adjusted. To maintain a consistent land base for comparison between scenarios and locations, farm size for the study was based on the location with the lowest yield and the scenario requiring the highest number of hectares to produce the feedstuffs needed to sustain the cattle.

The amount of manure produced by the livestock on-farm was also determined. Where the manure produced on-farm was insufficient to meet the target N requirements for crops (AAFRD 2004), supplementary synthetic fertilizer (both N and P) was applied.

Enteric CH₄ emissions were calculated for all categories of cattle on the farms. Our CH₄ production equation was based on gross energy intake (GEI) of the cattle, which was calculated based on animal requirements and feed energy values (Table 7). Separate equations were used for calculating GEI and CH₄ production for calves (Table 7). The percent of GEI lost as CH₄ (Y_m) varied among categories of cattle; for example, a Y_m value of 6% GEI (Boadi and Wittenberg 2002) was used for cows and bulls kept in corrals, while a Y_m value of 4% GEI (IPCC 2006) was used for cattle in the finishing phase. Although Y_m values were changed to reflect changes in management, the value used for a given management strategy was consistent across all locations. Methane conversion values for the different animal types and scenarios were taken from IPCC (2006) and from Canadian studies (McCaughey et al. 1997; McCaughey et al. 1999; Boadi and Wittenberg 2002; Ominski et al. 2006). Coefficients for the effect of ionophores and fats on CH₄ production, dry matter intake (DMI), and ADG were based on Boadi et al. (2004a) and Guan et al. (2006).

Table 7. Equations used to calculate enteric CH₄ emissions from beef cattle.

Equation	Definition of Terms
$GEI = (((NE_m + NE_a + NE_l + NE_p) / REM) + (NE_g / REG)) / (TDN / 100)$ <p>[applies to all categories of cattle except suckling calves]</p>	<p>GEI = gross energy intake (MJ/head/day) (IPCC 2006)</p> <p>NE_m, NE_a, NE_l, NE_p = net energy required for maintenance[*], animal activity, lactation, and pregnancy, respectively (MJ/head/day) (IPCC 2006)</p> <p>REM = ratio of net energy available in diet for maintenance to total digestible nutrients consumed (IPCC 2006)</p> <p>NE_g = net energy needed by the animal for growth (MJ/head/day) (IPCC 2006)</p> <p>REG = ratio of net energy available for growth in diet to total digestible nutrients consumed (IPCC 2006)</p> <p>TDN = total digestible nutrients in feed[†] (% TDN)</p>
$CH_{4\text{ enteric}} = (GEI * (Y_m/100) / 55.65 * (1 - (A / 100))) * d * h$ <p>[applies to all categories of cattle except suckling calves]</p>	<p>CH_{4 enteric} = total enteric CH₄ produced (kg CH₄) (adjusted from IPCC 2006)</p> <p>Y_m = CH₄ conversion factor[‡] (% GEI intake lost as CH₄)</p> <p>55.65 = energy content of CH₄ (MJ/ kg CH₄) (IPCC 2006)</p> <p>A = reduction in enteric CH₄ production due to use of feed additives (added to IPCC (2006) equation)[§]</p> <p>d = # of days in period</p> <p>h = # of animals in category</p>
$GEI_{\text{calves}} = C_w * 0.4 * 0.01 * d/2 * 18.45$ <p>[applies to suckling calves]</p>	<p>GEI_{calves} = gross energy intake (MJ/head/day) (Darryl Gibb unpublished)</p> <p>C_w = average weight of beef cow (kg) (Darryl Gibb unpublished)</p> <p>[assumes: calf weighs 40% of dam's weight at weaning; calf consumes 1% of its own body weight (Darryl Gibb unpublished)]</p> <p>energy density of feed = 18.45 MJ/kg DM (IPCC 2006)</p>
$CH_{4\text{ enteric}} = GEI_{\text{calves}} * (Y_m / 55.65) * (1 - (A / 100)) * c$ <p>[applies to suckling calves]</p>	<p>Y_m = same as Y_m used for dams' diet</p> <p>c = # of cows with live calves</p>

^{*} average weight (kg): cows = 591; bulls = 1000; background steers and heifers: 276, 274 in Period 1, 344, 342 in Period 2; replacement heifers: 352 in period 2, 430 in Period 3; feedlot steers and heifers: 393, 385 in Period 2, 518, 487 in Period 3; mature cow and bull weight = same as average weight; finished steer and heifer weight (kg) = 615 and 568

[†]baseline TDN of: 61% for confined cows, bulls, replacement heifers and background calves; 75% for feedlot cattle; 64.1% for grazing cows, bulls, replacement heifers (AAFRD 2003)

[‡]baseline Y_m of: 6% for confined cows, bulls, replacement heifers, background calves (Boadi and Wittenberg 2002); Y_m of 4% for feedlot cattle (IPCC 2006); Y_m of 9.5% for grazing cows and bulls (McCaughy et al. 1999); Y_m of 8.7% for grazing replacement heifers (Ominski et al. 2006)

[§]baseline additive reduction factor = 0

Calculations for CH₄ emissions from cattle manure were based on the amount of volatile solids (VS) excreted, the length of storage (if any), the maximum CH₄ producing capacity (Bo) of manure and a methane conversion factor (MCF) specific for the storage type used. The amount of VS excreted depended largely on GEI and on feed characteristics (Table 8); VS excretion from calves was calculated separately. Straw used as bedding is not included in the total VS or in the estimate of manure CH₄ emissions (IPCC 2006). Since the use of straw is associated with solid manure storage systems, the contribution of the VS in straw to total CH₄ emissions is minimal.

Table 8. Equations used to calculate CH₄ emissions from cattle manure in storage and deposited directly onto pasture by grazing animals (IPCC 2006).

Equation	Definition of Terms
$VS = (GEI * (1 - TDN / 100) + (0.04 * GEI)) * ((1 - 0.08) / 18.45) * h$ <p>[applies to all cattle except suckling calves]</p>	<p>VS = volatile solids[*] (kg/day) 0.04 = urinary energy expressed as a fraction of GE 0.08 = ash content of manure (%/100)</p>
$VS = ((GEI * (1 - TDN / 100) + (0.04 * GEI)) * ((1 - 0.08) / 18.45)) / d * c$ <p>[applies to suckling calves]</p>	<p>VS = volatile solids[†] (kg/day)</p>
$CH_{4_manure} = VS * 0.19 * MCF * d * 0.67$	<p>CH₄_{manure} = CH₄ emitted from a specific manure handling system (kg CH₄) 0.19 = maximum CH₄ producing capacity of manure (m³ CH₄/kg) MCF = CH₄ conversion factor for manure[‡] 0.67 = conversion factor from volume (m³ CH₄) to mass (kg CH₄)</p>

^{*}uses GEI from Table 7 that applies to all cattle except suckling calves

[†]uses GEI from Table 7 that applies to suckling calves

[‡]MCF = 0.17 for deep bedding; 0.01 for pasture (IPCC 2006)

Nitrous oxide emissions from manure are affected by the amount of N excreted in the feces and urine of cattle, which is a function of protein intake and protein retention by cattle (Table 9). Emissions are also affected by the length of the grazing season, which

affects the amount of manure deposited directly onto pasture. Indirect N_2O emissions, which emanate from N lost via leaching or volatilization, are related to the type of manure storage (manure kept in a drylot versus manure deposited on pasture by grazing animals). Leaching losses of zero were assumed for solid manure storage (Table 9). As there are no volatilization or leaching losses of straw N during solid manure storage (IPCC 2006), straw was not included in our estimates of manure storage N_2O emissions.

The N_2O emissions (both direct and indirect) from land application of cattle manure were affected by the amount of N lost as N_2O -N during manure storage, and the season of land application (Table 10). As straw used for bedding represents a source of additional N applied to land, N_2O emissions from straw should be included with emissions resulting from land application of manure (IPCC 2006). For simplicity, we estimated N_2O emissions from straw by making the assumption that no residue was removed from fields.

Our estimates of N_2O emission included N_2O released from N inputs to the land (synthetic fertilizer, plant residue from the previous year, mineralization of native soil organic matter), and considered influences of tillage, fallowing, soil texture, climate, and irrigation (Table 11). We also estimated indirect emissions from leaching, run-off and volatilization. Estimated N_2O emissions from N inputs were calculated for each crop type, whereas others were calculated for the entire farm.

We included CO_2 emissions from fossil fuels used on-farm and for the production of machinery, herbicides and N and phosphorus (P) fertilizers (Table 12). Carbon dioxide emissions from direct fuel use and production of machinery were based on the area of crops, forages, pasture and fallow and the energy use per hectare, with a unique

energy use value associated with each crop type or fallow. A similar equation was used to calculate the CO₂ emissions from herbicide production, with the assumption that it did not include the area of forages or pasture.

Carbon dioxide emissions or removal were calculated in response to recent (within 20 years) changes in land management, including changes in tillage, changes in frequency of fallow or perennial forage in rotations, and the conversion of cropped land to permanent cover (Table 13). Carbon dioxide emissions resulting from straw addition to or removal from the land were not included in our analysis. Uptake of CH₄ by soils was assumed to be constant, at a rate of 0.325 kg CH₄-C/ha/year for all land. Although this value has some uncertainty, its contribution to net whole-farm emissions is almost negligible.

The net whole-farm emissions were expressed both as Mg CO₂ equivalent (CO₂e) per farm and as Mg CO₂e per Mg protein exported off-farm in crop and animal products. Expressing emissions per unit of protein provides a common basis for comparing emissions across different farming systems. We assumed a protein content of 12.5, 14.0 and 7.8%, for barley, alfalfa-grass hay and low quality forage, respectively. Grass pasture and alfalfa-grass pasture were assumed to have protein contents of 12.5 and 17.1%, respectively. The value for grass pasture was an average of early season grass pasture and native pasture (AAFRD 2003). The estimated protein contents of finished steers, finished heifers and cull cows were 16.0, 15.9 and 15.8%, respectively (Berg and Butterfield 1976). The dressing percentage of steers, heifers and cull cows was 59.2, 57.7 and 49.5%, respectively (AAFRD 2007d). Protein in the animal tissue of cull cows and finished

Table 9. Equations used to calculate direct and indirect N₂O emissions from cattle manure in storage and on grazed pastures.

Equation	Definition of Terms
$PI_{solid_calf} = (GEI_{calves} / 18.45) * P$ [applies to suckling calves]	PI_{solid_calf} = calf crude protein intake from solid feed* (kg/head/day) P = % protein in mother's diet†
$PI_{milk_calf} = ((153 * M) / d) * MP$ [applies to suckling calves]	PI_{milk_calf} = calf protein intake from milk* (kg/head/day) M = milk production‡ (kg/day) MP = milk protein content§ (%/100)
$PR_{calf} = (PI_{solid_calf} * 0.2) + (PI_{milk_calf} * 0.4)$ [applies to suckling calves]	PR_{calf} = total protein retention of calves* (kg/head/day); assumes that 20% of solid feed protein and 40% of milk protein is retained (Darryl Gibb, personal communication)
$PR_{fetal} = 5 / d$ [applies to pregnant cattle]	PR_{fetal} = protein retention for pregnancy* (kg/head/day) 5 = amount of protein (kg) deposited in fetus for every gestation cycle (Prior and Laster 1979)
$N_{excreted} = (PI + PI_{solid_calf} + PI_{milk_calf}) / 6.25 - ((PR_{fetal} + PR_{calf}) / 6.25 + PR_{lactation} / 6.38) * h * d$ [applies to all cattle]	$N_{excreted}$ = daily nitrogen excreted by animals (kg/head/day) (IPCC 2006) PI = crude protein intake† (kg/head/day) (IPCC 2006) [kg protein = kg N * 6.25] $PR_{lactation}$ = protein retention in cow's milk (kg/head/day) (IPCC 2006) [kg protein = kg N * 6.38]
$N_2O-N_{manure_direct} = \sum N_{excreted} * EF_{manure_handling_system}$	$N_2O-N_{manure_direct}$ = direct N ₂ O–N from manure (kg N ₂ O–N) (IPCC 2006) $\sum N_{excreted}$ = the sum of N excreted from all animal categories (kg N) $EF_{toredmanure_handling_system}$ = emission factor for release of N ₂ O–N from manure handling system¶
$N_2O-N_{indirect_manure} = N_2O-N_{leaching_manure} + N_2O-N_{volatilization_manure}$	$N_2O-N_{indirect_manure}$ = indirect losses of N ₂ O–N through volatilization, leaching and run-off (kg N ₂ O–N) (IPCC 2006) $N_2O-N_{leaching_manure}$ = manure N lost through leaching and run-off* (kg N ₂ O–N) (IPCC 2006) $N_2O-N_{volatilization_manure}$ = manure N lost through volatilization^ (kg N ₂ O–N) (IPCC 2006)

*equation obtained from Darryl Gibb, personal communication, Agriculture and Agri-Food Canada, Lethbridge, AB.

†baseline crude protein content of: 14% for confined cows, replacement heifers, bulls and background calves; 11.0 and 11.2% for feedlot cattle in Period 2 and 3, respectively; 12.5% for grazing cows, bulls and replacement heifers (AAFRD 2003)

‡milk production (kg/head/day): Period 1 = 0; Period 2 = 7.9; Period 3 = 5.8 (Butson and Berg 1984)

[§]milk protein = 3.5% (Butson and Berg 1984)

^ΦEF_{stored_manure_handling_system} = 0.01 for deep bedding and 0.02 for pasture (IPCC 2006)

[¥]percent of manure N lost during leaching: deep bedding = 0; pasture = calculated (Rochette et al. in press); emission factor for N₂O-N emissions from leaching = 0.0125 kg N₂O-N/kg N leached (IPCC 2001).

^λpercent of manure N lost during volatilization: deep bedding = 30; pasture = 20; emission factor for N₂O-N emissions from volatilization = 0.01 kg N₂O-N/ kg NH₃-N + NO_x-N volatilized (IPCC 2006)

Table 10. Equations used to calculate direct and indirect N₂O emissions from land application of cattle manure.

Equation	Definition of Terms
$\frac{N_2O-N_{land_applied_direct}}{EF_{fertilizer} * TF_{manure_applied}} = N_{left_in_manure_storage} *$	<p>$N_2O-N_{land_applied_direct}$ = direct emissions from land application of manure (kg N₂O-N) (adjusted from IPCC 2006)</p> <p>$N_{left_in_manure_storage}$ = N left in manure storage after losses during storage (kg N) (IPCC 2006)</p> <p>$EF_{fertilizer}$ = emission factor for fertilizer applied to land*</p> <p>$TF_{manure_applied}$ = time of application factor for season of land application of manure[†] (added to IPCC (2006) equation)</p>
$\frac{N_2O-N_{indirect_manure}}{N_{volatilization_manure}} = \frac{N_2O-N_{leaching_manure} + N_2O-N_{volatilization_manure}}{N_{volatilization_manure}}$	<p>$N_2O-N_{indirect_manure}$ = indirect losses of N₂O-N through volatilization, leaching and run-off (kg N₂O-N) (IPCC 2006)</p> <p>$N_2O-N_{leaching_manure}$ = manure N lost through leaching and run-off[‡] (kg N₂O-N) (IPCC 2006)</p> <p>$N_2O-N_{volatilization_manure}$ = manure N lost through volatilization[§] (kg N₂O-N) (IPCC 2006)</p>

* $EF_{fertilizer}$ = 0.007 in MB or N.AB; 0.001 in SK; 0.0167 in S.AB (derived from a compilation of published and unpublished measurement-based Canadian data (Philippe Rochette , Agriculture and Agri-Food Canada – Ste. Foy & Reynold Lemke, Agriculture and Agri-Food Canada – Swift Current))

[†] If spring manure application, $TF_{manure_applied}$ = 1.0 in MB, SK or N.AB; 1.3 in S.AB; If fall manure application, $TF_{manure_applied}$ = 1.1 in MB, SK or N.AB; 1.8 in S.AB

[‡] percent of manure N lost during leaching: SK = 5; MB, N.AB = 10; S.AB = 25 (IPCC 2006); emission factor for N₂O-N emissions from leaching = 0.0125 kg N₂O-N/kg N leached (IPCC 2001)

[§] percent of manure N lost during volatilization: 20 (IPCC 2006); emission factor for N₂O-N emissions from volatilization = 0.01 kg N₂O-N/kg NH₃-N + NO_x-N volatilized (IPCC 2006)

Table 11. Equations used to calculate direct and indirect N₂O emissions from crop production (grain crops, forages and pasture).

Equation	Definition of Terms
$N_{\text{fert}} = ha * N$	$N_{\text{fert}} = \text{N inputs from synthetic fertilizer}^*$ (kg N)
$N_{\text{min}} = \Delta C_{\text{soil}} * 1/10$	$N_{\text{min}} = \text{N inputs from mineralization of soil organic matter (kg N) (IPCC 2006) [assumes ratio of mineralized C/mineralized N = 10]}$ $\Delta C_{\text{soil}} = \text{average annual loss of soil C}^{\dagger}$ (kg)
$AG_{\text{residue_N}} = ((Y - (mc * Y)) * (AG_{\text{residue_ratio}} / Y_{\text{ratio}})) * (1 - \text{residue_removed} / 100) * AG_{\text{residue_N_conc}}$	$AG_{\text{residue_N}} = \text{N in above ground plant residue (kg N/ha)}$ $Y = \text{crop yield}^{\dagger}$ (kg/ha); $mc = \text{moisture content of crop}^{\S}$ (w/w) $AG_{\text{residue_ratio}} = \text{above ground residue ratio}^{\Phi}$; $Y_{\text{ratio}} = \text{yield ratio}^{\Psi}$ $\text{Residue_removed} = \% \text{ of plant residue removed from the field}^{\Lambda}$ $AG_{\text{residue_N_conc}} = \text{N concentration of above ground residue}^{\Omega}$ (kg N/kg)
$BG_{\text{residue_N}} = (P * ((Y - (mc * Y)) * (BG_{\text{residue_ratio}} / Y_{\text{ratio}}))) * BG_{\text{residue_N_conc}}$	$BG_{\text{residue_N}} = \text{amount of N in below ground plant residue (kg N/ha)}$ $P = 1 \text{ if annual crop; } 0.2 \text{ if perennial [assumes perennial crop terminated in every 5 yrs]}$ $BG_{\text{residue_ratio}} = \text{below ground residue ratio}^{\Xi}$; $BG_{\text{residue_N_conc}} = \text{N concentration of below ground residue}^{\sigma}$ (kg N/kg)
$N_{\text{residue}} = (AG_{\text{residue_N}} + BG_{\text{residue_N}}) * ha$	$N_{\text{residue}} = \text{N inputs from crops returned to soil (kg N)}$ $ha = \text{area of all crops, perennial forages and improved pasture (ha)}$
$N_2O\text{-}N_{\text{inputs}} = (N_{\text{fert}} + N_{\text{min}} + N_{\text{residue}}) * EF_{\text{eco}}$	$N_2O\text{-}N_{\text{inputs}} = \text{N}_2\text{O-N emissions from N cropping inputs (kg N}_2\text{O-N) (Rochette et al. in press)}$ $EF_{\text{eco}} = \text{ecodistrict emission factor (kg N}_2\text{O-N/kg N) (Rochette et al. in press)}$
$N_2O\text{-}N_{\text{fallow}} = (N_S - N_F) * EF_{\text{eco}} * ha_F$	$N_2O\text{-}N_{\text{fallow}} = \text{N}_2\text{O-N emission from fallow (kg N}_2\text{O-N) [this assumes N mineralization on fallow = difference in recommended N rate for wheat on stubble}^{\omega} \text{ and wheat on fallow}^{\tau}]$
$N_2O\text{-}N_{\text{direct}} = N_2O\text{-}N_{\text{inputs}} + N_2O\text{-}N_{\text{tillage}} + N_2O\text{-}N_{\text{soil texture}} + N_2O\text{-}N_{\text{irrigation}} + N_2O\text{-}N_{\text{fallow}}$	$N_2O\text{-}N_{\text{direct}} = \text{direct N}_2\text{O emissions (kg N}_2\text{O-N) (Rochette et al. in press)}$ $N_2O\text{-}N_{\text{tillage}}, N_2O\text{-}N_{\text{soil texture}}, N_2O\text{-}N_{\text{irrigation}} = \text{N}_2\text{O-N associated with tillage, soil texture, and irrigation, respectively}$
$N_2O\text{-}N_{\text{indirect_crops}} = N_2O\text{-}N_{\text{leaching_crops}} + N_2O\text{-}N_{\text{volatilization crops}}$	$N_2O\text{-}N_{\text{indirect_crops}} = \text{indirect losses of N}_2\text{O-N through volatilization, leaching and run-off (kg N}_2\text{O-N)}$

$N_2O-N_{\text{leaching_crops}} = \text{N lost through leaching and run-off}^{\#} \text{ (kg } N_2O\text{-N) (Rochette et al. in press)}$

$N_2O-N_{\text{volatilization_crops}} = \text{N lost through volatilization}^{\ddagger} \text{ (kg } N_2O\text{-N) (IPCC 2006)}$

* See Table 5 for N fertilizer application rate. Synthetic fertilizer is only used on those hectares that did not receive cattle manure (calculated from animal feed characteristics).

† The sum of $CO_2-C_{\text{tillage}} + CO_2-C_{\text{frequency}} + CO_2-C_{\text{cover}} + CO_2-C_{\text{forage}}$ (Table 13)

‡ See Table 5 for crop yields for each crop on each farm.

§ Moisture content of: barley = 0.12; alfalfa and alfalfa-grass = 0.13; grass = 0.13 (Janzen et al. 2003)

Ⓢ Above ground residue ratio: barley = 0.47; alfalfa and alfalfa-grass = 0.10; grass = 0.12 (Janzen et al. 2003)

* Yield ratio: barley = 0.38; alfalfa and alfalfa-grass = 0.40; grass = 0.18 (Janzen et al. 2003)

^ 0 residue is removed from the field for both barley and alfalfa-grass hay. In fact, straw is removed from the barley field, but to include emissions from straw used for bedding, we input 0 residue removed.

Ⓜ Above ground residue N concentration: barley = 0.007; alfalfa and alfalfa-grass = 0.015; grass = 0.016 (Janzen et al. 2003)

Ⓛ Below ground residue ratio: barley = 0.15; alfalfa and alfalfa-grass = 0.50; grass = 0.70 (Janzen et al. 2003)

Ⓞ Below ground residue N concentration: barley = 0.010; alfalfa and alfalfa-grass = 0.015; grass = 0.010 (Janzen et al. 2003)

Ⓝ N fertilizer application rate for spring wheat (kg/ha) – stubble: MB = 71; SK = 54; S.AB = 95; N.AB = 71 (rate is 2/3 of the range for the crop and soil type given in AAFRD (2004) – calculation shown in Table 5)

† N fertilizer application rate for spring wheat (kg/ha) – fallow: MB = 28; SK = 17; S.AB = 17; N.AB = 28 (rate is 2/3 of the range for the crop and soil type given in AAFRD (2004) – calculation shown in Table 5)

Ⓢ Fraction of N lost via leaching was calculated using equations from Rochette et al. (in press); the emission factor for leached N was 0.0125 (IPCC 2001)

‡ Fraction of N lost via volatilization was 0.1 and the emission factor was 0.01 (IPCC 2006)

Table 12. Equations used to calculate the fossil fuel-derived CO₂ emissions from machinery, herbicide and fertilizer.

Equation	Definition of Terms
$CO_{2_fuel} = E_{fuel} * ha_i * 81$	CO_{2_fuel} = emissions from fuel use (kg CO ₂) E_{fuel} = energy requirement [*] (GJ/ha) ha_i = area of all crops, all perennial forages and excess improved pasture used for hay (ha)
$CO_{2_machinery} = E_{machinery} * ha_{ij} * 70$	$CO_{2_machinery}$ = emissions from use of fossil fuels in machinery production (kg CO ₂) $E_{machinery}$ = energy requirement [†] (GJ/ha) ha_{ij} = area of all crops, all perennial forages and excess improved pasture used for hay (ha)
$CO_{2_herbicide} = E_{herbicide} * ha_{iii} * 43$	$CO_{2_herbicide}$ = emissions from use of fossil fuels in herbicide production (kg CO ₂) $E_{herbicide}$ = energy requirement [‡] (GJ/ha) ha_{iii} = area of crops (ha)
$CO_{2_N_fertilizer} = N_{rate} * ha * 2.372$	$CO_{2_N_fertilizer}$ = emissions from use of energy to make N fertilizer (kg CO ₂) N_{rate} = N fertilizer application rate (kg/ha) ha = area fertilized with synthetic N fertilizer (ha)
$CO_{2_P_fertilizer} = P_{rate} * ha * 0.4223$	$CO_{2_P_fertilizer}$ = emissions from use of energy to make P fertilizer (kg CO ₂) P_{rate} = P fertilizer application rate (kg/ha) ha = area fertilized with synthetic P fertilizer (ha)

^{*} E_{fuel} for crops: MB and N.AB = 1.43; SK = 1.42; S.AB = 0.97; E_{fuel} for fallow: MB and N.AB = 0.93; SK and S.AB = 0.34

[†] $E_{machinery}$ for crops: MB, SK and N.AB = 0.48; S.AB = 1.27; $E_{machinery}$ for fallow: MB and N.AB = 0.28; SK and S.AB = 0.04

[‡] $E_{herbicide}$ for crops: MB, SK and N.AB = 0.46; S.AB = 0.24; $E_{herbicide}$ for fallow: MB and N.AB = 0.60; SK and S.AB = 0.78

E_{fuel} , $E_{machinery}$ and $E_{herbicide}$ are assumed values, based, in part, on E. Smith (personal communication)

Table 13. Equations used to calculate the CO₂ emissions or removal resulting from soil carbon change.

Equation	Definition of Terms
$CO_2-C_{tillage} = ha * SC_{tillage} * 1000$	$CO_2-C_{tillage}$ = change in soil C from changes in tillage (kg CO ₂ -C) for pertinent area (ha) $SC_{tillage}$ = soil carbon coefficient for changes in tillage practices (Mg C/ha) [*] (Boehm et al. 2004)
$CO_2-C_{frequency} = (ha_{crop} + ha_{fallow}) * ((1 - fallow_{present}/100) - (1 - fallow_{past}/100)) * SC_{frequency} * 1000$	$CO_2-C_{frequency}$ = change in soil C caused by changes in cropping frequency (kg CO ₂ -C) ha_{crop} = area of crops, perennial forages and improved pasture (ha); ha_{fallow} = current area of fallow (ha) $fallow_{present}$ = percentage of land now in fallow [†] ; $fallow_{past}$ = percentage of land in fallow 20 years ago $SC_{frequency}$ = soil carbon coefficient for reducing the percentage of fallow [‡] (Mg C/ha) (Frick et al. 2000)
$CO_2-C_{cover} = ha_{converted} * SC_{cover} * 1000$	CO_2-C_{cover} = change in soil C caused by changes in the amount of permanent cover (kg CO ₂ -C) $ha_{converted}$ = area of cropped land converted to permanent cover [§] (ha) SC_{cover} = soil carbon coefficient for conversion of cropped land to permanent cover ^Φ (Mg C/ha) (Boehm et al. 2004)
$CO_2-C_{forage} = (ha_{crop} + ha_{fallow}) * (forage_{present} / 100 - forage_{past} / 100) * SC_{cover} * 1000$	CO_2-C_{forage} = change in soil C caused by changes in the frequency of perennial forages in the crop rotation (kg CO ₂ -C) $forage_{present}$ = current percentage of perennial forages in the rotation (not including improved pasture) [‡] $forage_{past}$ = percentage of perennial forages in the rotation 20 years ago

^{*} $SC_{tillage}$ = -0.19 for SK and S.AB; -0.365 for MB and N.AB

[†] current fallow area is the same as 20 years ago

[‡] $SC_{frequency}$ = -0.6 for SK and S.AB; -0.4 for MB and N.AB

[§] no land has been converted to permanent cover in the past 20 years

^Φ SC_{cover} = -0.313 for SK and S.AB; -0.899 for MB and N.AB

[‡] current percentage of perennial forages in the rotation is the same as it was 20 years ago

steers and heifers, as well as protein in any grain, forages or pasture which were not fed to cattle were included in the protein exported off-farm. The pasture was rotationally grazed, with some portions of the grazing system to be left ungrazed. Where excess grass was produced, we assumed it was cut and baled for hay. Emissions were measured using protein exported off-farm rather than protein produced on-farm per se, because exported protein represents the net farm production.

Methane and N_2O were converted to CO_2e using a global warming potential of 23 and 296, respectively (Ramaswamy et al. 2001). All calculations were performed using Microsoft Excel[®] software.

4.4 RESULTS AND DISCUSSION

Net whole-farm emissions (Mg CO_2e per farm) varied widely among farms. The MB and N.AB farms had lowest emissions, with average net emissions for all scenarios of approximately 286 and 291 Mg CO_2e , respectively (Appendix I). The SK farm had average emissions of approximately 465 Mg CO_2e , and the S.AB farm had average net emissions of about 1072 Mg CO_2e . When expressed per unit of productivity, however, emissions for MB, SK, S.AB and N.AB were 4.54, 4.41, 2.80 and 2.00 Mg CO_2e per Mg protein, respectively (Appendix I).

The largest contributor to net emissions was CH_4 on the MB, SK and N. AB farms and N_2O on the S.AB farm (Figure 7). The CH_4 emissions were similar across all locations. The N_2O emissions were similar for the MB and N.AB farms, with SK having slightly lower N_2O emissions. The S.AB farm had N_2O emissions that were several-fold greater than those of the MB, SK and N.AB farms, reflecting the higher soil moisture

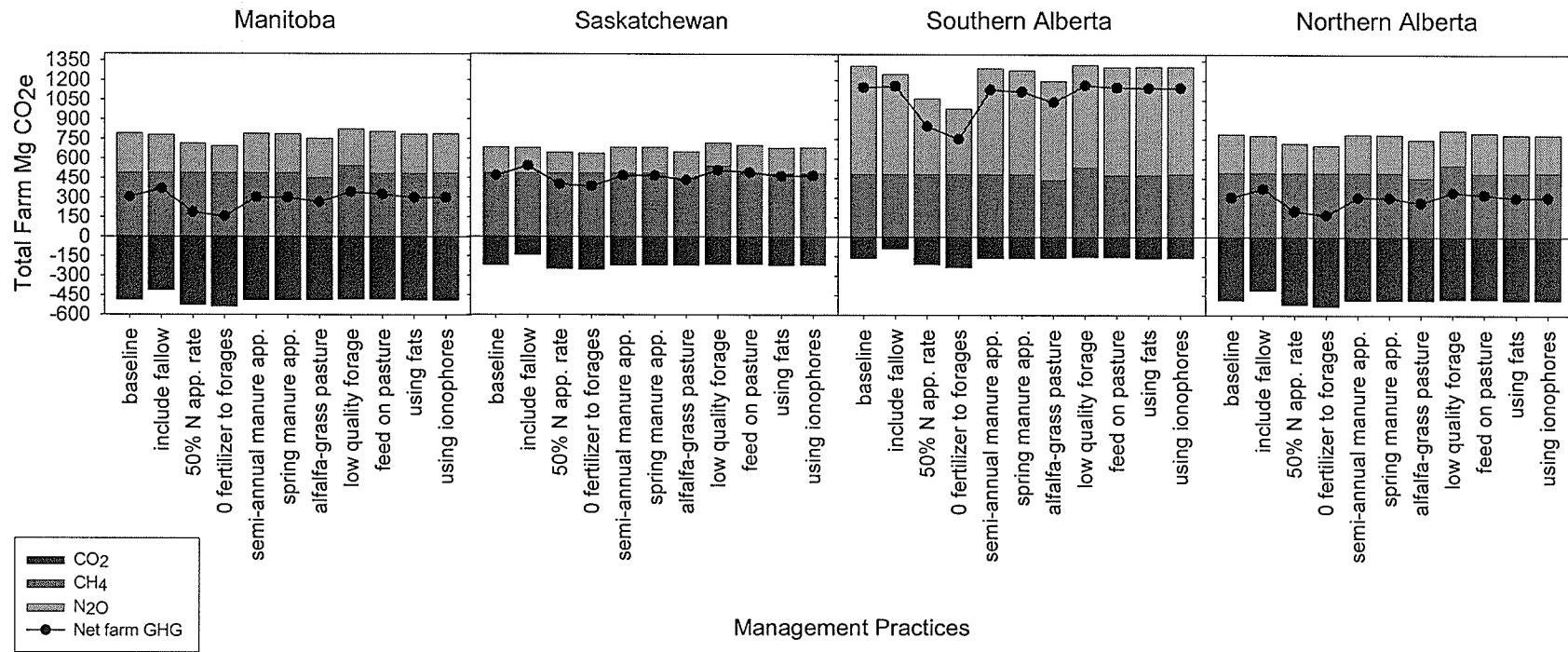


Figure 7. Greenhouse gas emissions (Mg CO₂e) for each management practice on each of the four hypothetical farms.

resulting from irrigation on the S.AB farm. The S.AB farm also had high CO₂ emissions, due to more intensive energy use associated with irrigation. The negative CO₂ emissions on the MB and N.AB farms were much higher than those on the other farms because their soils (Black Chernozem and Gray Luvisol) tend to show higher rates of C accrual with improved practices (Boehm et al. 2004).

Reducing fertilizer rates appreciably reduced overall emissions from farms. For example, the largest reductions in total emissions per farm (Mg CO₂e) were achieved by eliminating fertilization of forages, at all locations (Figure 7). Applying less N fertilizer (either by reducing the fertilizer application rate by half or eliminating fertilization of forages) significantly lowered N₂O production, and reduced overall emissions. This also lowered CO₂ emissions from fuel use and machinery and synthetic fertilizer production, as fewer hectares of pasture were cut and baled for hay and less synthetic fertilizer was applied.

But when productivity was taken into account (expressing GHG emissions per unit protein), reducing fertilizer rates often *increased* emissions. For example, reducing the fertilizer application rate by half resulted in the largest increase in emissions per Mg protein on the MB farm, because of the decline in productivity. Similarly, eliminating fertilization of forages led to the largest rise in emissions per Mg protein on the SK farm (Figure 8). As a consequence of lower yields, more land is needed to meet the feed requirements of the cattle, leaving fewer hectares for other crops with saleable value. While reducing the amount of N fertilizer applied does reduce net emissions, it can increase emissions per unit protein because of suppressed yields (e.g. the MB and SK farms). The increase in emissions per unit protein resulting from decreased fertilizer

rates which was observed on the MB and SK farms were less apparent on the S.AB and N.AB farms, presumably due to the higher yields at the latter locations. On the S.AB and N.AB farms, reducing fertilizer application rates by half increased emissions per unit protein, relative to that of the baseline scenario, whereas not applying any fertilizer to forage or pasture land showed a very slight decrease in emissions per unit protein, relative to that in the baseline scenario, for both the S.AB and N.AB farms.

Including 30 ha of fallow (and reducing the area of barley by the same amount) showed the greatest increase in net farm emissions (Mg CO₂e) on the MB, SK and N.AB farms (Figure 7). Including fallow also increased emissions per unit protein (Figure 8) because fallow reduced C sequestration (Grant et al. 2004). Nitrous oxide emissions declined because of reduced fertilizer use, but the N₂O decrease was outweighed by increased CO₂ emission. As less land was in production, less protein was available for export from the farm, making emissions per unit protein higher. The resulting increase in GHG output demonstrates the importance of studying farming systems as a whole as opposed to their component parts.

Pasturing cattle on an alfalfa-grass mix as opposed to grass only was the most effective practice for reducing emissions at each of the four locations when estimating emissions based on farm productivity (Figure 8). The reduction in Mg CO₂e per Mg of protein occurred primarily through reduced enteric CH₄ emissions. The higher nutritive value of alfalfa, compared to grass, increases the rate of digestion. The increased dietary protein content and microbial N synthesis associated with alfalfa increases the amount of non-ammonia N that reaches the small intestine. This, along with the improved energy

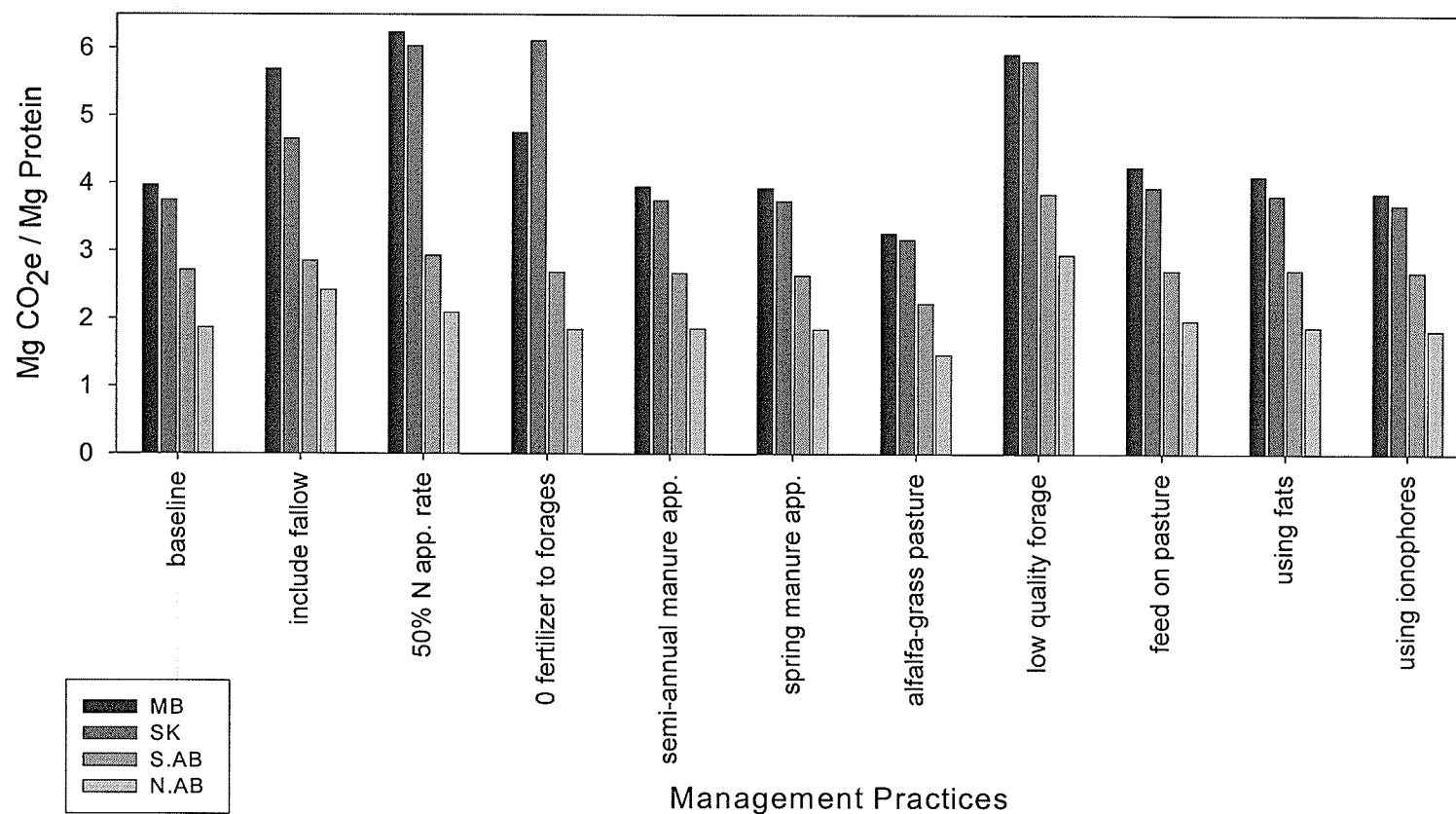


Figure 8. Greenhouse gas emissions per unit of protein exported off-farm (Mg CO₂e/Mg protein) for each management practice on each of the four hypothetical farms.

utilization associated with legume diets, improves overall digestive efficiency (Frame and Laidlaw 2005).

Feeding lower quality forage over the winter tended to increase both Mg CO₂e (Figure 7) and Mg CO₂e/Mg protein (Figure 8), though the responses varied somewhat among farms. Low quality forage increases CH₄ production because it slows the rates of passage and ruminal fermentation, creating a high acetate:propionate ratio, which is conducive to CH₄ production (Boadi et al. 2004a). Slower passage rates reduce DMI, resulting in less fermentable substrate in the rumen, and therefore increase CH₄ production (Johnson and Johnson 1995; McAllister et al. 1996). Conversely, the N₂O emissions were decreased due to the lower protein content of the feed, and consequently, the lower amounts of N present in the manure during storage and land application. However, the decrease in N₂O emissions was not enough to counteract the large increase in enteric CH₄ emissions.

The timing of manure application had limited effect on emissions. For the SK farm, the model estimated no change in emissions for either splitting the land application of cattle manure between spring and fall or land applying manure in the spring only compared to the baseline management scenario (where all manure is land applied in the fall). For the other farms, land applying ½ of the manure in the fall and the other ½ in the spring and land applying all of the manure in the spring decreased net farm emissions (Figure 7) and emissions per unit protein (Figure 8) when compared to the baseline scenario, but reductions were small. Decreased emissions resulting when ½ or all of the cattle manure was land applied in spring, as opposed to full manure application in fall,

may be attributed to lower amounts of manure-N being present on the land during spring thaw, when most of the annual N_2O emissions occur (Nyborg et al. 1997).

Feeding cows and bulls on pasture from the middle of October until the end of February increased both net farm emissions (Figure 7) and emissions per unit of protein (Figure 8) at all farms, except S.AB where there was no change. Enteric CH_4 emissions increased because cattle which are fed hay while on fall/winter pasture have higher GEI's than confined cattle, but the lower MCF for manure deposited on pasture (versus stored) resulted in a net decrease in CH_4 emissions. The MCF for stored manure is higher than for manure deposited directly on pasture because the anaerobic conditions and accumulation of organic matter that occur in manure storage are conducive to CH_4 production (IPCC 2006). Manure N_2O emissions increased because the N_2O emission factor for manure deposited directly on pasture is greater than that for manure kept in storage, because of greater aeration in pasture-deposited manure (Jun et al. 2002). A shorter confinement period reduced the amount of cattle manure available for land application, resulting in the use of more synthetic fertilizer, and increased N_2O and CO_2 emissions. The overall effect of this practice, therefore, reflected the net effect of reduced CH_4 emissions and enhanced N_2O and CO_2 emissions.

Adding fats to finishing diets slightly reduced overall net emissions ($\text{Mg CO}_2\text{e}$) for all farms (Figure 7). Fats in the diet reduce enteric CH_4 production because they provide an alternative hydrogen sink to CO_2 (Czerkawski et al. 1966), inhibit certain ruminal microbial populations (NRC 2001) and reduce fibre digestibility (McGinn et al. 2004). Fatty acids serve as an alternative hydrogen sink to CO_2 , preventing the hydrogen from attaching to the CO_2 to form CH_4 (Czerkawski et al. 1966). Some fat will remain

unsaturated (NRC 2001), accumulate in the rumen and impede the microbes' ability to break down fibre (NRC 2001), reducing the fibre digestibility of forages and decreasing the amount of CH₄ produced (McGinn et al. 2004). Including fats in diets, however, can reduce ADG (Boadi et al. 2004a), and, by extending the length of time animals were in the feedlot, result in more feed consumed and less feed available for sale off-farm. Keeping the animals in the feedlot longer also increased the CH₄ and N₂O emissions from manure because more manure was produced. This increased emissions per unit protein by varying degrees at all locations (Figure 8). Including fats in finishing diets does not always lead to a reduction in DMI and ADG (Mir et al. 2002), but our results are based on Boadi et al. (2004a), which may be the study most relevant to our setting. Clearly, this potential mitigation practice merits more research attention, particularly because of apparent responses in both CH₄ and productivity, and the apparent sensitivity of that response to rate of amendment.

Including ionophores in the diet showed a slight decrease in emissions per unit protein when compared to the baseline at all locations. Rumen microorganisms can adapt to ionophores and, after a short period, the rumen bacteria start producing CH₄ again (Chen and Wolin 1979). As a result of this adaptation, the model showed only a 0.76 Mg CO₂e per Mg protein decrease in CH₄ production (Figure 8). Most of the observed decrease in overall emissions from this practice reflected reduced N₂O emission from a decrease in manure N (less manure was produced as the animals have a lower DMI when consuming ionophores with a high concentrate diet (Guan et al. 2006)). Ionophores are not a long-term solution for reducing enteric CH₄ emissions because of methanogens'

ability to adapt to them, and the increasing public pressure to reduce the use of antimicrobials in animal production (Beauchemin et al. 2008).

Our study shows the importance of looking at the beef production system as a whole, considering all gases, and all facets of the farm. For example, use of low quality forages on the MB farm increased CH_4 emissions by approximately 56 Mg CO_2e but reduced N_2O emissions by almost 25 Mg CO_2e . Focusing on only one of the gases might distort the benefits of the practice. In another example, feeding cows and bulls on pasture from the middle of October until the end of February increases enteric CH_4 emissions, but these are more than offset by a decrease in manure CH_4 ; in this instance, evaluating only enteric CH_4 emissions might yield the wrong recommendation, as there was a net decrease in CH_4 emissions. Similar examples where emissions from one source increased while others decreased are the scenarios where (i) fallow was included and (ii) where the pasture was alfalfa-grass rather than grass. All of these examples stress the importance of considering whole-farm emissions when examining the benefit of a proposed mitigation practice.

The location of the farm plays an important role in determining the relative importance of the various GHG's. For example, the analysis demonstrates that CO_2 and CH_4 contribute large, almost equal, proportions of the net farm CO_2e on the MB and N.AB farms, and therefore, management practices focusing on reducing emissions of these gases would lead to the largest reductions in GHG output. Reductions in CH_4 emissions would be the most effective means of reducing emissions on the SK farm, as CH_4 is the SK farm's largest contributor to emissions. However, for the S.AB farm, N_2O

is the largest GHG contributor, and practices aimed at reducing N₂O emissions might be most urgent there.

The effectiveness of practices in reducing GHG emissions varied, depending on location. For example, changing the timing of manure application had less effect on the SK farm than on the MB, S.AB and N.AB farms. Also, eliminating fertilization of forages showed small decreases in emissions per unit protein on the S.AB and N.AB farms but relatively large increases in emissions on the MB and SK farms. These observations suggest that there can be no universally-applicable set of best management practices – the practices that best reduce emissions depend on the location of the farm.

The variability of the effectiveness of practices among sites suggests the possibility of using resources more efficiently by focusing on those sites that are most responsive. For example, our analysis, if true, suggests that mitigation efforts on farms like MB might merit more attention than similar practices on farms like S.AB.

Our analysis suggests, further, that the net overall effect of mitigative practices, at least those evaluated, was often small. For example, the practices evaluated at the SK site reduced net emissions by 17%, at best, relative to the baseline scenario. In part, this observation may reflect the multiple processes of GHG formation in farming systems. By virtue of the numerous processes, and the complexity of their interactions, it may be possible to appreciably reduce emissions from one source, but that benefit may be partly offset or at least dwarfed by emissions from another source.

Expressing emissions in terms of Mg CO₂e per Mg protein changed the ranking of the different locations. The MB farm, which had the lowest net farm emissions (Mg CO₂e), was the highest emitter when farm productivity was accounted for, and the S.AB

farm, which had the highest net farm emissions, was the second lowest GHG producer when productivity was considered because of the higher crop yields.

Perhaps more importantly, the relative effectiveness of various practices in reducing emissions depended on whether those emissions were expressed as Mg CO₂e or as Mg CO₂e per Mg protein exported off-farm. Reporting emissions in terms of farm productivity is more pertinent, we propose, because it more accurately describes the environmental cost of that product. Choosing mitigation practices based simply on total net emissions might be misleading, because it could favor practices that reduce productivity. For example, in the scenario where the N application rate was cut in half for all crops, the net emissions (Mg CO₂e) decreased at all four farms, but emissions per Mg protein increased as a result of lost productivity. Similar results were observed for the scenario where zero fertilizer was applied to forages and pasture on the MB and SK farms. Comparing management practices based on emissions per unit of protein identifies those practices and conditions with highest efficiency – those that yield highest productivity per unit of GHG emission. Furthermore, expressing emissions per unit protein may minimize ‘leakage’ – the phenomenon whereby reduced emissions from lower productivity at one site are merely shifted to sites elsewhere that provide compensatory production.

Farms are complex ecosystems, and estimating net emissions requires numerous assumptions and empirical equations, many of which have significant uncertainty. Our estimates are still approximate, and subject to further research and measurements. With more research and scientific advances, many of the implicit assumptions and coefficients will no doubt change, perhaps modifying the predicted values. For example, rates of N₂O

emissions from grazed pastures may be particularly tenuous and in need of further research. These uncertainties, however, do not diminish the value of this analysis in illustrating the importance of trying to capture whole-farm influences when evaluating proposed mitigation practices. Moreover, such whole-farm analyses help to identify those facets of farms that merit the most urgent scientific scrutiny.

4.5 CONCLUSION

The model used in our study will benefit from enhanced understanding that will come with further research. At present, it demonstrates, despite its uncertainties, that a whole-system approach is essential for evaluating the practices that best reduce emissions. Further, it shows that the ‘best’ practice depends on where the farm is located; there may be no universally-applicable set of best management practices. Finally, this exercise illustrates that expressing emissions per unit of protein might prevent the recommendation and subsequent utilization of mitigation practices that reduce emissions at the expense of restricted productivity.

5.0 MANUSCRIPT II

Estimating greenhouse gas emissions from a beef pasture system in southern Manitoba
using measurements and a systems-based model

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5.1 ABSTRACT

Livestock production accounts for approximately 59% of Canada's agricultural greenhouse gases (GHG's). Because livestock farms are such large GHG emitters, it is important to investigate management strategies that can reduce net emissions from the entire production system. In this study, a model was used to estimate annual net farm emissions from various management strategies including emissions of methane (CH_4) from livestock and manure, nitrous oxide (N_2O) emissions from soil and manure (direct and indirect), and carbon dioxide (CO_2) from energy use and soil carbon change. Whole farm emissions were expressed as carbon dioxide equivalents (CO_2e) per hectare and also as CO_2e per unit of protein exported off farm to examine the impact of management strategy on farm productivity. The first objective was to evaluate the effectiveness of two proposed mitigation strategies using two systems-based approaches: i) predicted emissions derived from an Intergovernmental Panel on Climate Change (IPCC)-based model and (ii) measured emissions derived from a multidisciplinary field study. The second objective was to identify those facets of whole-farm emissions most in need of further study. The measured emissions were garnered from a field study designed to examine GHG emission associated with three fertility treatments applied to grazed forage: no liquid hog manure (control); 242 kg total N/ha in a spring application of liquid hog manure (full); 121 kg total N/ha in each of a spring and fall application of liquid hog manure (split). The measured values and the model output showed the following ranking among treatments in net emissions per ha: full > split > control. The measured values, expressed per unit protein showed the following rankings: split \approx full > control (although

split was slightly higher than full). The model predictions, expressed per unit protein showed a similar ranking, although a larger difference was predicted between the split and full treatments: split > full > control. Emissions (both net emissions per ha and per unit protein) from a hypothetical synthetic fertilizer treatment, included in the model estimates, were intermediate to the predicted control and manured treatments. Thus, trends in emissions were similar between predicted and measured approaches, although individual flux estimates varied appreciably. The model was able to create an emissions estimate for a complete year based on long-term means, as opposed to actual measurements which may be taken from only a portion of the year and which are subject to highly variable local conditions during the period of measurement and between years. The predicted and measured enteric CH₄ emissions are comparable; however, much less certainty exists for N₂O, as there is a large discrepancy between the predicted estimates and measured values. Some components of the system have a negligible impact on the overall emissions from the system, while others are of much greater consequence. Future research should be directed particularly towards components, such as enteric CH₄ and soil N₂O, that have larger impacts on overall system GHG emissions.

Abbreviations: **ASH**, manure ash content; **C**, carbon; **CH₄**, methane; **CO₂**, carbon dioxide; **CO₂e**, carbon dioxide equivalents; **CP**, crude protein; **DMI**, dry matter intake; **EF**, emission factor; **GHG**, greenhouse gas; **ha**, hectare; **N**, nitrogen; **NI**, nitrogen intake; **N₂O**, nitrous oxide; **TDN**, total digestible nutrients; **TS**, total solids; **VS**, volatile solids

Keywords: systems-based measurements, systems-based model, methane, nitrous oxide, carbon dioxide, beef cattle, grazing

5.2 INTRODUCTION

Canada produced 747 Mt CO₂e of greenhouse gases (GHG's) (carbon dioxide (CO₂), methane (CH₄) and nitrous oxide (N₂O)) in 2005, of which agriculture accounted for roughly 10% (Janzen et al. 2008). Livestock production is responsible for approximately 59% of Canada's agricultural greenhouse gases (GHG's) (Neitzert et al. 2007). Because livestock farms are such large GHG emitters, it is important to investigate management strategies that may be utilized to reduce emissions.

As there are many interactions and feedbacks between different areas of the farm, the net effect of farming practices can only be quantified by considering whole farms and taking a systems approach to measuring GHG emissions (Stewart et al. 2008). As such, models may be the only practical way to estimate net emissions from whole farms (Janzen et al. 2006). The development of models can also identify areas where data and understanding are limiting. The ability of a model to accurately predict net emissions may be assessed by comparing predicted values to measurements taken from actual farming systems. As prolonged measurement of all GHG emissions from entire farms is rarely possible, predicted values are compared to measurements from specific components of the farm, often for relatively short time periods.

Most models now available focus on one GHG or one component of the grazing system. For example, some models focus only on emissions from soil (Del Grosso et al. 2000; Li 2000), while others focus only on direct emissions from livestock (Dijkstra et al. 1992; Mills et al. 2001; Mills et al. 2003). Some systems-based models exist; but, these models were developed to estimate GHG emissions from one particular farming system, such as a dairy farm (Olesen et al. 2006). Very few models are able to integrate, under

various management practices, grazing animals and the manure they produce with the manure applied to pasture as fertilizer (Kebreab et al. 2006). Progress was reported by Stewart et al. (2008), who illustrated how a simple model might be used to capture net whole farm emissions from such livestock farms.

A research study conducted at a site near La Broquerie, MB used a systems approach to measure GHG's, including those from grazing animals, manured and non-manured pasture soil, feces and synthetic urine. Although it was not possible to measure all the gases from all sources continuously, the study did provide a broad spectrum of measured emission data that can be used alongside Intergovernmental Panel on Climate Change (IPCC) models to estimate net GHG emissions at the farm level. A related study by Wiens et al. (2008) looked at the energy cost of applying liquid hog manure to pastures.

Our first objective was to evaluate the effectiveness of two proposed mitigation strategies using two systems-based approaches: i) predicted emissions derived from an IPCC-based model and ii) measured emissions derived from a multidisciplinary field study. Our second objective was to demonstrate, based on an evaluation of measured and predicted values, the key areas where knowledge is lacking and enhancements are most urgent.

5.3 MATERIALS AND METHODS

Data was collected from a research trial (Ominski et al. 2008) conducted from 2004 – 2006, which consisted of three fertility treatments differing in the amount and timing of land-applied liquid hog manure: a 'control' treatment, receiving no manure; a

‘full’ treatment, receiving manure at a rate of 242 kg total nitrogen (N)/ha in spring; and a ‘split’ treatment, receiving manure in both spring and fall (121 kg total N/ha per application). Each fertility treatment had two replicates for the grazed and hayed paddocks; however, predicted emissions were only estimated for the grazed replicates in this study. In the research trial, measurements of N₂O emissions were taken from the hayed paddocks only. Canadian Council on Animal Care guidelines were followed in the care and management of the steers (CCAC 1993).

Enteric CH₄ was measured using the sulphur hexafluoride technique (Johnson et al. 1994), as validated by Boadi et al. (2002) and used extensively in the literature (Grainger et al. 2007; Hegarty et al. 2007; Pinares-Patiño et al. 2007). Nitrous oxide and CH₄ emissions from the soil were measured using a chamber technique similar to those of Hutchinson and Livingston (1993) and Mosier (1989), and widely used elsewhere (Carter and Ambus 2006; Maljanen et al. 2007; Saggar et al. 2007). (Appendix II outlines the calculations used to determine 2006 soil emissions; 2004 and 2005 soil emissions were from Ominski et al. 2008). Emissions of N₂O and CH₄ from soils were measured in hayed paddocks to avoid chamber disturbance by the grazing cattle, and hence, do not include emissions from feces and urine. To compensate, emissions were measured separately from samples of feces collected from the grazed paddocks applied to soil in the hayed paddocks and synthetic urine applied to soil in an un-grazed alleyway connecting the site to an adjacent pasture (Ominski et al. 2008).

Emission factors for N₂O release from feces and urine (Table 14) were calculated separately for each treatment based on chamber measurements (Appendix III). The urine data included emissions measurements from 2005 only, whereas the fecal data included

measurements from 2004 and 2005. Equations (4) and (5) (Kebreab et al. 2001) were used to estimate the amount of N excreted in urine and feces (g N/day) from N intake (NI; g N/day).

$$\text{urinary N} = -0.70 + 0.30 * \text{NI} \quad (4)$$

$$\text{fecal N} = 32.0 + 0.28 * \text{NI} \quad (5)$$

Nitrogen intake was based on measured animal dry matter intake (DMI) and forage crude protein (CP) content. Fecal and urine N₂O emissions per treatment were then estimated using the calculated emission factors and N excretion (Appendix III).

An emission factor for CH₄ production from feces of grazing animals (Table 14) was calculated separately for each treatment based on chamber measurement data from 2004 and 2005 (Appendix IV). The amount of volatile solids (VS) excreted in the feces was calculated using an Intergovernmental Panel on Climate Change (IPCC) (2006) equation (6):

$$\text{VS} = \text{DMI} * (1 - \text{TDN}) * (1 - \text{ASH}) \quad (6)$$

where DMI is the dry matter intake of the cattle (kg/head/day), TDN is the total digestible nutrients of the feed (%/100), ASH is the ash content of the manure (% DM) (Table 15). The VS were then used to calculate total solids (TS), assuming that manure is 84.6% VS (Allan and Embleton 2001). Fecal CH₄ emissions per treatment were estimated from emission factors and total amount of feces excreted (Appendix IV).

Carbon dioxide emissions from energy use during land application of liquid hog manure were estimated from the IPCC (2006) emission factor for diesel fuel (0.0741 kg CO₂ per MJ energy used) and the energy requirements for applying liquid hog manure to a pasture with a slurry wagon (Wiens et al. 2008) (Appendix V).

Table 14. Emission factors (EF) for estimation of CH₄ production from feces and N₂O release from feces and urine data collected over a three-year period.

Treatment	Feces CH ₄ EF (g CH ₄ -C / kg feces)	Feces N ₂ O EF (mg N ₂ O-N / g N)	Urine N ₂ O EF (mg N ₂ O-N / g N)
Control	0.064	0.074	5.05
Split	0.072	0.238	3.15
Full	0.088	0.820	3.15

The model used in this study, described by Stewart et al. (2008), considers CH₄ emissions from livestock and manure (both feces and urine), direct and indirect N₂O emissions from soil and manure, CO₂ emissions from energy use and soil carbon (C) change. These emissions are then integrated to estimate annual net farm emissions for various management practices, expressed as carbon dioxide equivalents (CO₂e) and CO₂e per hectare (ha). Whole farm emissions are also calculated in CO₂e per unit of protein exported off farm to examine the impact on farm productivity.

Our study also considered an additional hypothetical treatment (synthetic fertilizer), as addition of some nutrients to these pastures is necessary to avoid eventual depletion. The model inputs for the synthetic fertilizer scenario (Table 15) were comparable to the ‘full treatment’, except synthetic fertilizer was used instead of hog manure, at the same rate of N application. Plant residue that remained on the pastures at the end of the grazing season was not removed from the site, and acted as a N input to the soil.

The system evaluated included the following parameters: backgrounded cattle grazing for 82 days, the pasture utilized by these animals, and the management strategy employed to fertilize the pasture land. The model was used to estimate enteric

fermentation for the duration of the grazing season and annual N_2O and CH_4 emissions from soil and manure deposited by grazing cattle. Each of the three fertility treatments was entered into the model individually. Data collected from 2004 to 2006 was averaged for each treatment and was used in the model (Table 15).

Table 15. Model inputs were derived from measured data collected over a three-year period for each of three fertility treatments.

Model Input	Control	Split	Full	Data Reference ¹
Treatment area (ha)	16.2	8.1	8.1	
Pasture dry matter yield (kg/ha)	1199	2618	2021	
Animal start weight (kg)	279.9	281.5	281.5	
Animal average weight (kg)	320.0	313.7	324.4	
Animal final weight (kg)	360.1	345.9	367.3	
Protein exported as cattle growth (kg/ha)	20.5	56.8	71.7	Berg and Butterfield 1976; Ominski et al. 2008
Protein content of forage (%)	9.5	16.2	17.8	
Total digestible nutrients of forage (%)	64.50	66.48	69.69	Calculated using equation from the Cornell Nutrition Conference (1979) as cited by the Bodycote Testing Group and data from Ominski et al. 2008
Dry matter intake (kg/head/day)	6.73	6.72	6.60	
Average daily gain (kg/head/day)	1.17	0.81	1.11	
Ash content of cattle manure (% DM)	18	16	17	Determined from manure samples using the AOAC method ^{2,3}
CH_4 produced (% gross energy)	6.6	6.0	6.0	

intake)			
Average stocking density (#head/ha)	1.4	4.7	4.4
Length of grazing season (days)	82	82	82
Spring liquid hog manure application rate (kg N/ha)	0	121	242
Fall liquid hog manure application rate (kg N/ha)	0	121	0

¹Data reference is Ominski et al. (2008) unless otherwise indicated.

²Association of Official Analytical Chemists (AOAC) 1990, method no. 942.05

³Manure samples were dried and then ground. A portion of the sample (1-2 g) is put into a porcelain crucible and put into the muffle furnace at 550°C for 12 hours. The remaining residue is weighed, and the ash content is determined using the following equation:

$$((W3 - W1)/W2) * 100 = \% \text{ ASH} \quad (7)$$

where W1 = weight of the crucible, W2 = weight of the crucible and manure sample before placing in the muffle furnace, W3 = weight of the crucible and remaining residue after being removed from the muffle furnace, and ASH = manure ash content.

The climate inputs (long term means of growing season precipitation (401 mm) and potential evapo-transpiration (609 mm)) were obtained from Marshall et al. (1999). The fraction of N lost through volatilization during land application of liquid hog manure was assumed to be 25% (Prairie Provinces' Committee on Livestock Development and Manure Management 2006). Emissions during storage of the liquid hog manure (uncovered earthen storage system) used as fertilizer were also included in the predicted GHG emissions estimates (Appendix VI). The moisture content of hog manure and amount of liquid hog manure applied to each treatment used in determining manure storage emissions were taken from Ominski et al. (2008). The VS and TS used were from Allan and Embleton (2001). Annual CH₄ production (m³) during the storage of liquid hog manure was calculated based on average monthly temperatures (Vergé et al. 2006) and converted to kilograms (kg) of CH₄ using a conversion factor of 0.67 kg

CH₄/m³ (IPCC 2006). Monthly storage N₂O emissions were calculated using emission factors and fractions lost through volatilization and leaching (IPCC 2006) and summed to yield annual estimates. Where specific emission factors or data were not available from the measured data (e.g. the grazed land methane conversion factor and the emission factor for leaching of manure deposited on pasture by grazing cattle), default values from Stewart et al. (2008) were used. As indicated in Table 16, not all gases were measured or included from all sources (e.g. CH₄ and N₂O emissions from hog manure storage were not measured, and urine CH₄ emissions were measured, but not included because urine is not expected to increase CH₄ emissions as it is not a large source of C for methanogens (Jarvis et al. 1995)); hence the total predicted and measured emissions cannot be directly compared. However, the emissions derived using both prediction and measurement techniques can be ranked as a means of assessing the mitigation potential of the management strategies, and the shortcomings of each technique can be identified.

Net emissions were expressed both as Mg CO₂e/ha and also as Mg CO₂e/Mg protein, where Mg protein refers to protein exported from each treatment at the end of the grazing season. The amount of exported protein was calculated from the total weight gain of all animals on each treatment pasture and the protein content of that gain. The total emissions for each treatment (Mg CO₂e), divided by the amount of protein exported, yielded an estimate of emissions per unit protein (Mg CO₂e/Mg protein) for each treatment. The protein content of the liveweight gain was assumed to be 18.8% (Berg and Butterfield 1976). Liveweight gain was determined using start and finish weight data described in Table 15.

Table 16. Description of sources and sinks of GHG emissions in the predicted and measured emissions.

Emission Source/Sink	Measured Emissions ¹	Predicted Emissions ¹
Enteric CH ₄		
- respired CH ₄	Y	Y
- flatulated CH ₄	N	Y
Soil CH ₄	Y	Y
Manure CH ₄		
- cattle feces	Y	Y
- cattle urine	N	Y
Hog manure storage CH ₄	N	Y
Hog manure storage N ₂ O	N	Y
Manure N ₂ O		
- cattle feces and urine	Y	Y
Soil N ₂ O		
- soil N ₂ O (late spring–early fall)	Y	Y
- soil N ₂ O (fall - spring of next year)	N	Y
- land application of liquid hog manure	Y	Y
Energy use, as measured by CO ₂		
- land application of imported/liquid hog manure	Y	Y

¹Y = included; N = not included

All CH₄ and N₂O emissions were converted to CO₂e, using a global warming potential of 23 for CH₄ and 296 for N₂O (Ramaswamy et al. 2001). All calculations were performed using Microsoft Excel[®].

5.4 RESULTS AND DISCUSSION

For the model scenarios, the dominant GHG varied among treatments. In the control, full and split scenarios, CH₄ was most important, whereas in the synthetic fertilizer scenarios, N₂O was dominant (Table 17).

Table 17. Contribution of CO₂, CH₄ and N₂O as a percent of total emissions (Mg CO₂e/ha) for each modeled scenario and measured treatment.¹

Treatment	GHG	Modeled	Measured ²
Control (% of total Mg CO ₂ e/ha)	CO ₂	0	0
	CH ₄	72	89
	N ₂ O	28	11
Full (% of total Mg CO ₂ e/ha)	CO ₂	2	9
	CH ₄	63	60
	N ₂ O	35	31
Split (% of total Mg CO ₂ e/ha)	CO ₂	4	11
	CH ₄	58	73
	N ₂ O	39	16
Synthetic Fertilizer (% of total Mg CO ₂ e/ha)	CO ₂	17	NA
	CH ₄	30	NA
	N ₂ O	53	NA

¹Totals may not add up to 100% due to rounding

²NA = not available

The model output suggested the following ranking of treatments: control (0.72 Mg CO₂e/ha) < synthetic fertilizer (4.40 Mg CO₂/ha) < split (8.24 Mg CO₂e/ha) < full (8.90 Mg CO₂e/ha). The higher emissions from the split and full scenarios resulted largely from CH₄ and N₂O emissions during storage of the liquid hog manure which was subsequently used to fertilize the pastures (Table 18).

Fertilizing pastures (with either liquid hog manure or synthetic fertilizer), resulted in significant CO₂ emissions from energy use during application and, in the synthetic fertilizer scenario, from fertilizer production (Table 18). These emissions were higher in the split than in the full treatment, and highest with synthetic fertilizer.

In the control and synthetic fertilizer scenarios, the majority of CH₄ emissions were generated by enteric fermentation. In the scenarios with hog manure, however, CH₄ emissions were dominated by manure storage. In all of the scenarios, CH₄ release from soil, feces and urine was minimal (Table 18).

Table 18. The net emissions (Mg CO₂e/ha) for each modeled scenario, by emission source.

Emission Source	Control	Full	Split	Synthetic Fertilizer
CO ₂				
CO ₂ – energy	0.00	0.15	0.30	0.74
Total CO₂	0.00	0.15	0.30	0.74
CH ₄				
CH ₄ – cattle	0.52	1.32	1.27	1.32
CH ₄ – soil	-0.01	-0.01	-0.01	-0.01
CH ₄ – cattle feces	0.01	0.02	0.02	0.02
CH ₄ – cattle urine	<0.01	<0.01	<0.01	<0.01
CH ₄ – hog manure storage	0.00	4.31	3.45	0.00
Total CH₄	0.52	5.65	4.74	1.34
N ₂ O				
N ₂ O – hog manure storage	0.00	1.04	1.04	0.00
N ₂ O – cattle manure	0.10	0.67	0.68	0.67
N ₂ O – soil ¹	0.11	1.39	1.48	1.65
Total N₂O	0.20	3.10	3.20	2.32
Total Emissions	0.72	8.90	8.24	4.40

¹Soil N₂O includes direct and indirect N₂O emissions

As with the CH₄ emissions, the storage of hog manure was the largest contributor to N₂O emissions in the full and split scenarios. In the synthetic fertilizer scenario, the N₂O emissions were mostly comprised of emissions resulting from land application of synthetic fertilizer. Nitrous oxide emissions were very small in the control scenario (Table 18).

The measured estimates of net emissions (Mg CO₂e/ha) showed the following ranking of the three treatments: control (0.40 Mg CO₂e/ha) < split (1.47 Mg CO₂e/ha) < full (1.76 Mg CO₂e/ha). The differences in net emissions between treatments were largely attributed to differences in enteric CH₄ production and soil N₂O release (Table 19).

In the full and split treatments, the CO₂ emissions arose from energy use during land application of liquid hog manure. Increased fuel use for the split manure applications (Wiens et al. 2008) led to the slightly higher energy-derived emissions for the split treatment (Table 19).

In all three treatments, enteric fermentation produced most of the CH₄ emissions, while fecal CH₄ emissions were minimal. Soil CH₄ production contributed little to the total CH₄ emissions (Table 19).

In the manured treatments, N₂O emissions were largely a result of land application of liquid hog manure to pastures. Nitrous oxide emissions from the control treatment were very small (Table 19).

Expressing emissions per unit of exported protein (Mg CO₂e/Mg protein) allows practices to be compared based on their productivity. It acknowledges explicitly that reduced emissions are not necessarily feasible or desirable if they occur at the expense of

productivity. In the system under study, the only protein removed was that accumulated in the animal tissues, as measured by weight gain of the animals.

Table 19. The net emissions (Mg CO₂e/ha) for each measured treatment, by emission source.

Emission Source	Control ^{1,2}	Full ^{1,2}	Split ^{1,2}
CO ₂			
CO ₂ – energy	0.00	0.162	0.164
Total CO₂	0.00	0.162	0.164
CH ₄			
CH ₄ – cattle	0.31	0.98	0.97
CH ₄ – soil	0.04	0.08	0.10
CH ₄ – cattle feces	<0.01	<0.01	<0.01
CH ₄ – cattle urine	NI	NI	NI
CH ₄ – hog manure storage	NA	NA	NA
Total CH₄	0.36	1.06	1.07
N ₂ O			
N ₂ O – hog manure storage	NA	NA	NA
N ₂ O – cattle manure	0.01	0.04	0.03
N ₂ O – soil ³	0.03	0.49	0.21
Total N₂O	0.04	0.54	0.24
Total Emissions	0.40	1.76	1.47

¹NI = emissions were measured in the research trial, but were not included

²NA = not available

³Soil N₂O includes direct and indirect N₂O emissions

The modeled full and split scenarios had estimates of emissions per unit protein of 124 and 145 Mg CO₂e/Mg protein, respectively, much higher than those in the synthetic

fertilizer (61 Mg CO₂e/Mg protein) and control scenarios (35 Mg CO₂e/Mg protein) (Figure 9). Much of the difference between treatments was attributable to emissions from liquid hog manure storage.

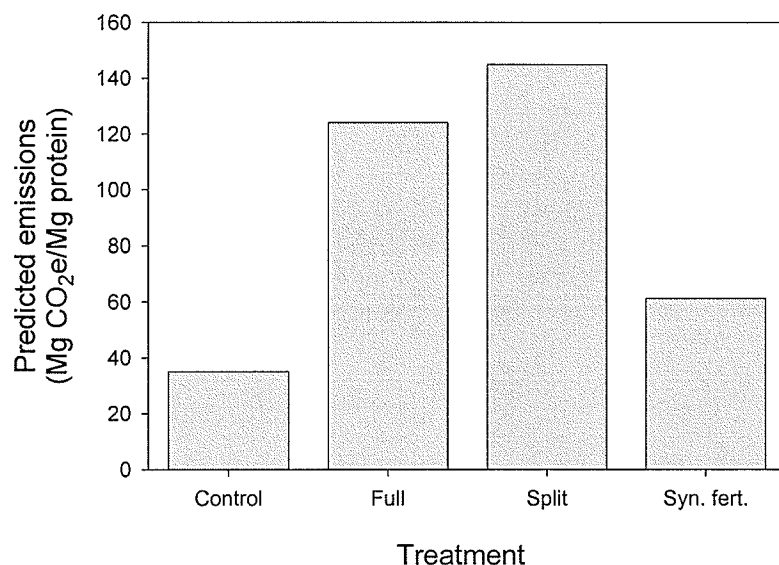


Figure 9. Emissions per unit protein removed (Mg CO₂e/Mg protein) for each modeled scenario (Syn. fert. = hypothetical synthetic fertilizer treatment).

The measured estimates of emissions per unit protein for the full and split treatments were very similar: 25 and 26 Mg CO₂e/Mg protein, respectively. The control treatment, estimated at 20 Mg CO₂e/Mg protein, had the lowest measured estimate of emissions per unit protein (Figure 10).

In both the measured and the predicted approaches, the control scenario had the lowest net emissions (Mg CO₂e/ha and Mg CO₂e/Mg protein). Although useful for comparison, this treatment would rarely be recommended. Without nutrients returning to the system, the soil's nutrient reserves would eventually be depleted, and productivity of the system would suffer. This finding implies that short-term estimates are not

necessarily reliable as indicators of sustainability, unless long-term implications are also considered.

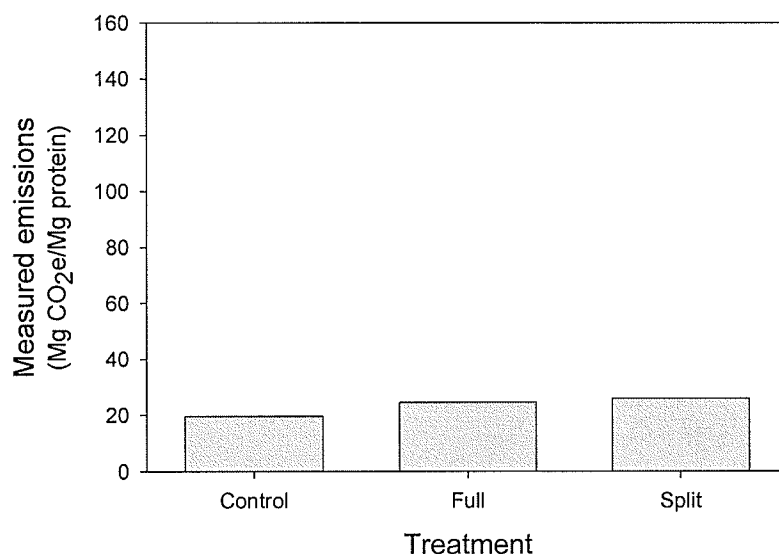


Figure 10. Emissions per unit protein removed (Mg CO₂e/Mg protein) for each measured treatment.

The predicted emissions included CH₄ and N₂O released during the storage of the liquid hog manure, whereas the measured values did not. This discrepancy regarding where the system boundary lies was responsible for the majority of the difference observed between the predicted and measured emissions.

In any systems analysis, a critical decision is where to draw the boundary of the system. For example, we included emissions from storage of liquid hog manure in the predicted emissions. It could be argued that these emissions should not be included in this system – they are accounted for on the hog farm, and should not penalize the practice of using manure as fertilizer. Further, the emissions from liquid hog manure storage, some might contend, will exist whether or not the manure is used to fertilize a pasture,

and that using livestock manure avoids emissions from synthetic fertilizer otherwise required. Nevertheless, the stored hog manure, we contend, is part of the system, and including it in our analysis allows us to investigate ways of reducing overall emissions by adjusting the way manure is used (e.g. timing of applications). Further, we have included a hypothetical 'synthetic fertilizer' treatment to estimate the net impact of replacing hog manure with fertilizer. To maintain consistency in our system boundaries, we have included the emissions associated with production of the synthetic fertilizer used.

Overall, aside from the liquid hog manure storage emissions, the measured CH₄ fluxes correspond closely to model estimates (Mg CO₂e/ha) (Tables 18 and 19). The small differences in enteric CH₄ could reflect minor problems with enteric CH₄ collection equipment (for example, broken hoses or clogged nosepieces) during measurement, or the unresponsiveness of the model to selective grazing. As forage availability was high during the course of the grazing season, the steers may have been able to graze selectively – choosing plants with higher levels of digestibility, crude protein and soluble carbohydrates, and lower levels of acid detergent fiber (Coleman and Barth 1973).

The measured estimates of CH₄ emissions from manure deposited directly on pasture by grazing cattle were much lower than those predicted by the model. Static chamber measurements of soil emissions were taken from the hayed paddocks of the research trial, rather than the grazed pastures, possibly contributing to the differences observed between the measured emissions values and the predicted emissions. As well, emissions from feces and urine were measured from cattle manure patties placed in the hayed paddocks and synthetic urine patches in an un-grazed alleyway connecting the research site to an adjacent pasture. Although CH₄ emissions from the urine patches

were measured, they were not included in the measured emissions for our analysis because urine CH_4 made up such a minute proportion of our predicted emissions and other authors have previously demonstrated that urine CH_4 contributions to total emissions are negligible (Jarvis et al. 1995; Yamulki et al. 1999). A second factor contributing to the discrepancy might be the difference in time period: the model estimates CH_4 release over the course of a year from the volatile solids deposited during the grazing season, whereas the emission factors used to estimate measured CH_4 release from cattle feces were derived from a few fecal CH_4 emissions over the course of approximately one month in each of 2004 and 2005 (Appendix IV). The measured estimates, therefore, may underestimate cumulative emissions. Another factor contributing to the discrepancy could be inaccuracy of the model equation at high manure ash contents (16 – 18%), as the equation was designed with ash contents in the range of 8% (IPCC (2006) default value).

The model predicted that the soil acted as a small net sink for CH_4 . However, because the model predictions of soil CH_4 are based on the treatment area and a constant value for CH_4 oxidation, the model does not have the ability to truly estimate CH_4 emissions from soil or to incorporate environmental factors. The model prediction is similar to that observed by Saggar et al. (2007), a study in which a version of the DNDC model adapted for New Zealand was used to estimate emissions from a pasture grazed by sheep. However, measurements obtained in the current study showed net emissions of CH_4 in 2004 and 2005, contrary to that observed by Saggar et al. (2007). Precipitation during the 2004 and 2005 growing seasons was unusually high, presumably leading to the net CH_4 emissions observed (Rath et al. 1999). The water table at the research trial site

was high enough to reach into the root zone where the available C was, contributing to the observed net CH₄ emissions (Ominski et al. 2008).

Very large differences were observed between the measured and predicted N₂O estimates (Mg CO₂e/ha) (Tables 18 and 19), partly because N₂O emissions from hog manure storage were included in the predicted estimates. Furthermore, the measured estimates of N₂O emissions from the soil and hog manure included emissions from only part of the annual cycle (mid-May to mid-November in 2004, early April to early November in 2005, mid-April to early October in 2006), whereas the predicted estimates included annual N₂O emissions. A large proportion of annual N₂O emissions can occur during winter and early spring, especially during snow melt (Wagner-Riddle and Thurtell 1998; Pattey et al. 2007). These emissions, though they may occur when the animals are not grazing, must be included in any system analysis. In our study, the measured N₂O emissions are much lower than predicted emissions largely because measurements did not commence until April or May (depending on trial year) and, therefore, did not include the off-season fluxes. Differences between predicted and measured emissions may also be attributed to the technique used to measure soil emissions. Chambers are more suited to mitigation studies, where the goal is to evaluate if differences exist between treatments rather than to determine emission factors (McGinn 2006). This, along with the short collection period of N₂O emissions from feces and urine, could explain why the measured emission estimates were lower than those predicted by the model. Nitrous oxide release from feces was only measured for one month at the end of the grazing season in both of 2004 and 2005, while N₂O release from synthetic urine patches was only measured for two months during the 2005 grazing season.

Our model assumed that soil C change was negligible in all treatments because the land had already been under pasture for about 20 years. This assumption may have slightly underestimated the response of soil C. For example, measurements of root C at the study site showed appreciably higher values in manured treatments (3588 kg C/ha in the full treatment) compared to the control treatment (2695 kg C/ha). No significant change in soil C was observed, however, perhaps reflecting the short duration of the study. Soil C changes slowly, and differences among treatments are often detectable above natural variability only after several years or more (Ellert et al. 2000). The response of soil C to these treatments, therefore, remains a significant shortcoming in our ability to estimate net emissions from these systems.

Differences in predicted and measured estimates of CO₂, CH₄ and N₂O may be attributed also to differences in the duration of time examined; that is, the model predicted annual emissions, applicable to long-term average conditions, whereas the measured emissions were taken from a relatively limited land-base, at a specific point in time, during which temperature and precipitation differed from the long-term average.

The data set utilized in this study was extensive, consisting of soil GHG release and enteric CH₄ production measurements over a period of three years. As the goal of the research trial was to measure GHG emissions from a pasture system, CH₄ and N₂O release from urine and feces, as well as CO₂ generated through energy use, were also quantified. Even efforts of this magnitude, however, still did not capture all emissions from the full annual cycle of beef production system. By using the model developed in Stewart et al. (2008) to predict emissions from similar treatments, we were able to identify gaps in the research trial data set (such as the lack of N₂O emissions during late

winter/early spring). It is also apparent that despite the large number of measurements garnered from the research data, it is difficult to achieve an accurate assessment of net emissions at the farm level solely through measured data.

Data garnered from each individual component in the system has a different level of precision and accuracy associated with it, creating varying levels of uncertainty in our estimates of GHG emissions (Table 20). These levels of uncertainty are affected by our ability to measure or predict the emissions from each source. Sometimes our ability to predict emissions from a systems-component is higher than our ability to measure those emissions because of spatial and temporal variability. Models have the capacity to reduce the impact of spatial and temporal variability on their predictions. In other cases, measurements may yield more definitive results than even the best models, particularly where understanding of mechanisms is limited, or where environmental conditions at the study site deviate from those for which the model was developed.

However, when trying to decide where future research should focus, it is important to realize where in the system these efforts would be most warranted. Some components of the system (such as soil CH_4) for which we have little understanding have minimal impact on the overall total emissions from the system, while others (such as soil N_2O) are of much greater consequence (Table 20). Therefore, research should be shunted away from the zero or low importance areas and directed towards components that have large impacts on overall system GHG emissions and where most benefit from further study would be realized.

Table 20. The status of our general ability to measure or predict GHG emissions from beef production systems, and the relative importance of each component.

Emission Source/Sink	Ability to Measure ¹	Ability to Predict ¹	Relative importance ²
Enteric CH ₄	Strong	Strong	High
Manure CH ₄ from grazed areas	Weak	Intermediate	Negligible
Manure N ₂ O from grazed areas	Weak	Intermediate	Low
Manure storage CH ₄	Intermediate	Intermediate	Med ³
Manure storage N ₂ O	Weak	Intermediate	Low ³
Soil CH ₄	Weak	Weak	Negligible
Soil N ₂ O	Weak	Weak	High
Indirect N ₂ O from soil and manure	Weak	Weak	Low
Energy use CO ₂	Strong	Strong	Low
Soil C change	Intermediate	Intermediate	High

¹Indicators are rough approximations of our ability to measure or predict GHG emissions.

Indicator values of uncertainty are: Strong: <10%; Intermediate: 10-40%; Weak: >40%.

²Estimated values of: High: >35%; Medium: 21-35%; Low: 5-20%; Negligible: <5% of total emissions on typical beef production farms, similar to those described in Stewart et al. (2008) and Manuscript II.

³The relative importance of manure storage CH₄ and N₂O are medium and low, respectively, when hog manure is not used as a fertility source. The relative importance of both manure storage CH₄ and N₂O are high when hog manure is used as a fertility source.

5.5 CONCLUSION

Similar trends in emissions were observed in the modeled scenarios and the measured treatments when expressed as Mg CO₂e/ha and Mg CO₂e/Mg protein. Both estimated that the control treatment, although not a sustainable practice, released the fewest GHG, while the full treatment resulted in the greatest production of GHG (Mg CO₂e/ha). The full and split treatments were estimated to generate the largest amounts of

GHG's per unit of protein in both the model output and the measurements. Compared to measured estimates, the model predicted a much larger difference between each of the treatments.

An evaluation of the model predictions and the emissions from the measured data has demonstrated that challenges exist and additional research is required in the following areas: i) spring-thaw N₂O emissions; ii) further model development to better accommodate manure management practices, such as timing of manure application; iii) more accurate emission factors for calculation of N₂O emissions from soil, feces and urine in pasture systems; iv) more accurate methods to determine the distribution of feces and urine excreted by grazing animals; and v) how emissions are affected by animal-to-animal variation.

Despite these challenges, our study showed the importance and value of adopting a systems approach to estimating the benefits of proposed practices on net emissions. The evaluation of the measured emissions with the model predictions emphasizes the need for models when performing a systems analysis and demonstrates the difficulty of obtaining systems-based data. The model was able to create an emissions estimate which encompassed a complete year and was based on long-term means, as opposed to measurements taken from only a portion of the year which were subject to highly variable local conditions during the brief period for which measurement occurred.

The different levels of uncertainty associated with each system component should not be viewed in a negative light, but, used in conjunction with the relative importance of each component, as an opportunity to learn where we need to concentrate our research efforts in the future. Further, our study shows that neither models nor measurements are

the preferred method of estimating GHG emissions from farms; rather the two approaches are complementary and should be conducted concurrently for their mutual improvement.

6.0 GENERAL DISCUSSION

This study has demonstrated the complexity of conducting systems-based research, and the inter-connectedness of beef production systems. The following paragraphs discuss the different methods of reporting GHG's, the importance of using a systems-based approach, the importance and challenges associated with using models, and how future research is dependant upon the combined use of both models and measurements.

6.1 EXPRESSING EMISSIONS PER UNIT OF PROTEIN EXPORTED

The traditional strategy used to examine net GHG emissions is to convert all gases to carbon dioxide equivalents. The challenge with this strategy is that it is difficult to compare management practices that alter plant and animal productivity, and further, it is hard to determine the effects of management change on total farm productivity. For example, if both forage yields and animal growth are increased, how do we quantify the net increase in farm productivity when we use different units to measure changes in plant and animal productivity? There is no common unit of measure upon which the comparison can be based. In essence, we are comparing apples to oranges.

Expressing emissions per unit of protein exported from the system is a novel approach to reporting farm GHG emissions. The unit 'protein' is common to both plant products and animal tissue, and provides a ruler with one scale which we can use to measure total farm productivity. We can now compare apples to apples. Protein may be more useful than dry matter yield, because protein is related to food quality.

Evaluating GHG emissions in terms of the amount of protein exported from the farm allows the effects that alterations in management practices or farm location have on farm productivity to be observed and, most importantly, to be quantified in a meaningful way. By accounting for farm productivity, this approach to reporting GHG emissions has the potential to prevent or reduce the likelihood of making recommendations that result in increased GHG emissions through reduced farm productivity, and, therefore, increasing the need for more animals or hectares in production on other farms.

Other units, such as energy or lysine, could also be used to report GHG emissions. It is important that one measure of productivity be decided upon, and used by everyone to report agricultural GHG emissions. It is imperative that the unit selected be applicable to all farming types so that comparisons of GHG emissions can be made between livestock and grain farming operations.

6.2 IMPORTANCE OF SYSTEMS ANALYSIS

It is crucial that we examine every component of the beef production system when estimating the net GHG emissions. Performing a systems analysis, whether modeling hypothetical systems or taking measurements from existing farms, illustrates the true impact that a management practice has on GHG emissions, as some practices increase emissions of one GHG while decreasing those of another. The effect that location has on the production of GHG's can be included when a systems approach is used because it incorporates site specific variables, such as climate, yield and soil type. The magnitude of the effect of a change in management is not consistent across the

Canadian prairies. For example, opportunity for mitigating GHG's may be greater in moister regions, with high productivity, than in arid regions where productivity is limited.

In a systems analysis, emissions are designated to specific categories. Sometimes, debate exists as to which category emissions should be included in. For example, should the emissions from animal manure used as fertilizer be attributed to the livestock or the cropping sector? Currently, all manure emissions are included in the emissions associated with livestock production. However, by attributing all manure emissions to the livestock sector, we may be reducing the opportunity to reduce net agricultural GHG emissions that would arise from using animal manure as a fertility source in place of synthetic nitrogen and phosphorus fertilizer. If manure emissions were included in the cropping sector, grain and forage producers could reduce their net farm emissions by using animal manure.

Studying beef production farms as whole systems helps us to gain an understanding of how the processes interact or oppose one another so that we can improve our ability to model and measure GHG fluxes from all farm types and other ecosystems. A systems approach can also lead us to a greater understanding and appreciation of the non-commodity ecosystem services that exist in, and are essential to, a productive agricultural system.

Perhaps most importantly, systems analysis teaches us how to better manage our agricultural lands in a sustainable way. It is important to realize that what is 'best' management for achieving reduced GHG emissions and improved sustainability is not the same in all regions. Through use of this complex, but rewarding, research technique, we

may learn how to reduce GHG emissions while maintaining and improving other ecosystem services and agricultural production.

Systems analysis can be performed to estimate which regions in a country are the most efficient for producing certain agricultural commodities; some areas are more efficient at raising cattle, others at producing grain. Perhaps our comparison should be wider than political boundaries. Is it in our best interests to reduce agricultural production in areas of the world which are perhaps not the most efficient in their country, but when compared to other regions of the world, they are very efficient? Reducing global GHG emissions needs to be a global endeavour; however, political barriers exist which hinder this global objective. Political policy and national differences prevent food from being produced only in the regions where emissions would be the lowest. As no country wants to be entirely dependant upon other countries for their food supply, this is an obstacle to reducing global GHG emissions that we will have to accept and find ways to deal with as best we can, in hopes that it can be overcome.

6.3 IMPORTANCE OF MODELS

Models allow us to predict the impact of management changes on the GHG emissions for a particular farm type or for farms of the same type in different geographical locations. Setting up research trials to evaluate the effects of all the possible combinations of management practice changes for all regions of Canada is not possible – there are not enough researchers, time or resources available to carry out that work. Therefore, we must rely on models (based on a few thorough systems-based research trials) to provide us with the majority of our comparisons of GHG emissions

from various management scenarios. However, models do not displace measurements; instead, they are intended to make collecting field data more efficient and effective.

The effects of time can be incorporated into a systems analysis when models are used. Time is an important factor for determining the impact of management on some farm components, such as soil C sequestration. The ability to use long term averages for inputs such as climate and yield is an intrinsic value of models that reduces the variability due to present conditions that are unavoidable in measurements obtained from field data.

Models are a feasible option for acquiring annual GHG emissions estimates, as most models predict emissions on an annual basis. This represents one of the substantial advantages to using models, as measuring GHG emissions during Canadian winters and spring freeze/thaw cycles are difficult. As well, the immense number of measurements required to obtain an accurate estimate of GHG production over the course of the year is expensive, time consuming and requires the collaboration of a large team of researchers consisting of people with strengths in all major components of the system.

6.4 CHALLENGES OF USING MODELS

Although models are essential in our pursuit of effective mitigation strategies, they are sometimes difficult to use. It is a difficult task to design an efficient model. A good model needs to be complex enough to include the many different emissions sources from the system, but it also needs to be transparent enough that the results are simple to interpret and the model remains simple to use.

Often, a model has been developed for a particular system or set of variables, and adapting the model so it is appropriate for use in another situation may prove difficult.

The algorithms used in the model to describe the original system may not be appropriate for the second system, causing the model to yield inaccurate predictions of net GHG emissions.

Finding a data set that contains enough detail from which emission factors can be derived is difficult. A large number of measurements over a sufficient area and time are needed before an emission factor that is representative of the region and the average climate can be developed. Even once such a data set has been compiled, the resulting emission factor tends to be relevant only to the specific conditions/area under which the measurements were taken. For example, if a soil N₂O emission factor was developed from N₂O measurements taken in a pasture populated by grass species only and grazed by lactating cows, this emission factor would not give an accurate estimate of the N₂O emissions from a legume-based pasture grazed by yearling steers, as the mechanisms of gas production will be altered by the different plant species and animal types present.

Because minute changes in soil type, topography or micro-climate can dramatically affect the GHG's released from soils, it is challenging to find emission factors that are able to represent regions, or sometimes countries, for use in a model. Frequently, model users do not have emission factors which match the system being modeled. Instead, they use emissions factors based on the best available and most compatible data. For this reason, a large number of assumptions are inherent to any model estimates of GHG production and consumption.

6.5 FUTURE RESEARCH WORK

Model predictions of GHG emissions can be evaluated by comparing them to measurements of GHG emissions from existing farm systems, and vice versa. Where there is good agreement between model predictions and actual measured emissions (such as with enteric CH_4 production), we can conclude that we have a good understanding of the processes that affect the production and uptake of GHG's in that component of the beef production system. However, for system components where there is little or no agreement between model predictions and actual measurements (such as with soil N_2O), our understanding is lacking, and it is in these areas of the system that future research efforts should be focused.

Some system components contribute a larger percentage of total emissions than others. These relatively more important areas of the system are where we should focus our research efforts, because, as the areas of most importance, a change in these areas will have the greatest impact on net emissions. For example, the proportion of total emissions that soil CH_4 contributes is minimal. Therefore, any change in soil CH_4 uptake or release will have negligible effects on the total emissions. As resources, time and the number of systems-based researchers is limited, it is important that we focus our efforts on those areas of the highest relative importance, where we can potentially have the most impact on net GHG emissions, rather than on systems components that contribute almost nothing to the total farm emissions. Of these areas of high relative importance, we must center our research on the areas that we understand the least. As systems-based research is required to get a systems understanding of these components, there will need to be a marked increase in the number of studies using the systems approach in the near future.

Bringing together our strengths from both modeling and measuring will help us improve our estimates of GHG fluxes. In some instances, we have a moderate understanding of the mechanisms responsible for GHG emission and a limited capacity to measure these emissions. The reverse is also true. By combining the areas where we can model emissions effectively with those components which we can accurately and reliably measure, we may be able to generate a comprehensive estimate of GHG emissions for systems.

As our ability to understand systems processes improves and the accuracy of our predictions and measurements increase, it will be intriguing to see which management practices become the 'best' for reducing GHG emissions. Will the management practices deemed to be the most effective at reducing GHG emissions be economically viable without a carbon trading system or other monetary incentive? How similar or different will the 'best' management practices be in different areas of the Canadian prairies? And perhaps the most profound question of the three posed here is the last: Is it possible that the management practices we recommend today as being environmentally sound and beneficial in terms of reducing greenhouse gas emissions will be found to actually *contribute* to the rise in greenhouse gas emissions? Hopefully the answer to this final question is a definitive 'no', but to ensure that we do not begin to recommend and employ detrimental management practices, it is essential that more systems-based research be carried out, most urgently in those areas in which our understanding is weakest.

7.0 CONCLUSION

This research has demonstrated that a systems-based approach is essential for evaluating the mitigation potential of different management strategies. The examination of emissions of a single GHG could lead to an incorrect conclusion about the ability of a management practice to reduce GHG emissions, as some management practices decrease emissions of one GHG while increasing emissions of others. The 'best' strategy for reducing GHG emissions was not the same for each region of the Canadian prairies. It is important that when recommending management practices to farmers, we do not generalize; the management practice that best reduces GHG emissions has to be determined for each individual farm type and region.

The trends observed in the modeled scenarios were similar to the trends in the measurements when emissions were expressed both as Mg CO₂e/ha and Mg CO₂e/Mg protein. The model predicted higher emissions from each of the treatments, and a much wider gap between the control and manured treatments than the measured estimates. Lower emissions from the measured data illustrated the difficulty of obtaining field data from all sources/sinks of GHG's in beef production systems. A large number of measurements taken from a large area and over a prolonged period of time are needed before an emission factor that is representative of the region and average climate can be developed. Such data sets require significant resources, further emphasizing the need for models in systems research.

The units of measure used to report net GHG emissions affect the impact that a given management practice has on GHG emissions. Practices that reduced the amounts of fertilizer applied to crops and forage decreased whole-farm emissions ($\text{Mg CO}_2\text{e}$) on all hypothetical farms examined, but resulted in a reduction in farm productivity as measured by increased emissions per unit of protein ($\text{Mg CO}_2\text{e}/\text{Mg protein}$) on the MB and SK farms. Recommendation and subsequent utilization of mitigation practices that reduce emissions at the expense of restricted productivity can be prevented by expressing emissions per unit of protein.

Different levels of uncertainty associated with individual GHG's can be identified by how closely we are able to match measured and predicted estimates of emissions. By examining the proportion of total emissions contributed by each system component, we can estimate their relative importance. To improve our ability to estimate GHG emissions from agricultural systems, future research needs to incorporate the use of both models and measurements to focus on the system components of highest relative importance and those with the greatest uncertainty.

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9.0 APPENDICES

This section contains supplementary material for the results and discussion of Manuscript I (Appendix I) and the materials and methods of Manuscript II (Appendix II – VI).

9.1 APPENDIX I

9.1.1 Greenhouse gas emissions in Mg CO₂e and Mg CO₂e per Mg protein for the MB farm

Manitoba							
<i>Practice number</i>	<i>Practice Name</i>	CO ₂	CH ₄	N ₂ O	Total	Mg CO ₂ e/Mg protein	Net change in Mg CO ₂ e/Mg protein from baseline
CO ₂ equivalents (Mg)							
1	Baseline	-488.48	489.52	302.07	303.11	3.97	0
2	Include fallow at 7% of current farm size	-412.68	489.52	290.38	367.22	5.69	1.72
3	Cut fertilizer rate by half	-526.65	489.52	224.99	187.87	6.24	2.26
4	Apply zero fertilizer to forages and pasture	-539.11	489.52	205.27	155.68	4.76	0.79
5	Apply half of manure in fall; half in spring	-488.48	489.52	300.76	301.80	3.95	-0.02
6	Apply all manure in spring	-488.48	489.52	299.44	300.48	3.94	-0.03
7	Use an alfalfa-grass pasture	-488.43	449.26	305.71	266.54	3.27	-0.70
8	Feed a low quality forage	-481.49	545.64	277.45	341.60	5.92	1.95
9	Increase grazing season	-481.26	484.71	320.94	324.39	4.25	0.28
10	Put fats in finishing diets	-488.48	485.08	302.62	299.22	4.11	0.14
11	Put ionophores in finishing diets	-488.02	488.76	300.63	301.38	3.87	-0.10

9.1.2 Greenhouse gas emissions in Mg CO₂e and Mg CO₂e per Mg protein for the SK farm

Saskatchewan							
<i>Practice number</i>	<i>Practice Name</i>	CO ₂	CH ₄	N ₂ O	Total	Mg CO ₂ e/Mg protein	Net change in Mg CO ₂ e/Mg protein from baseline
CO ₂ equivalents (Mg)							
1	Baseline	-219.34	486.50	202.08	469.24	3.75	0
2	Include fallow at 7% of current farm size	-140.23	486.50	197.56	543.83	4.66	0.91
3	Cut fertilizer rate by half	-247.85	486.50	163.54	402.19	6.03	2.28
4	Apply zero fertilizer to forages and pasture	-254.89	486.50	156.46	388.08	6.12	2.36
5	Apply half of manure in fall; half in spring	-219.34	486.50	201.89	469.05	3.75	0.00
6	Apply all manure in spring	-219.34	486.50	201.70	468.86	3.75	0.00
7	Use an alfalfa-grass pasture	-219.31	446.24	208.29	435.22	3.18	-0.57
8	Feed a low quality forage	-212.47	541.59	181.60	510.72	5.82	2.07
9	Increase grazing season	-212.26	481.79	223.84	493.37	3.95	0.19
10	Put fats in finishing diets	-219.34	482.07	202.49	465.21	3.83	0.08
11	Put ionophores in finishing diets	-218.88	485.75	200.87	467.74	3.69	-0.06

9.1.3 Greenhouse gas emissions in Mg CO₂e and Mg CO₂e per Mg protein for the S.AB farm

Southern Alberta							
<i>Practice number</i>	<i>Practice Name</i>	CO ₂	CH ₄	N ₂ O	Total	Mg CO ₂ e/Mg protein	Net change in Mg CO ₂ e/Mg protein from baseline
CO ₂ equivalents (Mg)							
1	Baseline	-162.48	480.47	829.20	1,147.20	2.72	0
2	Include fallow at 7% of current farm size	-90.70	480.47	767.13	1,156.91	2.86	0.14
3	Cut fertilizer rate by half	-212.32	480.47	581.43	849.58	2.93	0.22
4	Apply zero fertilizer to forages and pasture	-233.72	480.47	506.38	753.13	2.69	-0.03
5	Apply half of manure in fall; half in spring	-162.48	480.47	813.49	1,131.48	2.68	-0.04
6	Apply all manure in spring	-162.48	480.47	797.77	1,115.77	2.64	-0.07
7	Use an alfalfa-grass pasture	-162.46	440.21	758.15	1,035.90	2.23	-0.48
8	Feed a low quality forage	-154.89	533.48	784.79	1,163.37	3.86	1.14
9	Increase grazing season	-154.61	475.96	826.48	1,147.83	2.72	0.00
10	Put fats in finishing diets	-162.76	476.04	829.23	1,142.50	2.73	0.01
11	Put ionophores in finishing diets	-162.19	479.72	826.03	1,143.56	2.70	-0.02

9.1.4 Greenhouse gas emissions in Mg CO₂e and Mg CO₂e per Mg protein for the N.AB farm

Northern Alberta							
<i>Practice number</i>	<i>Practice Name</i>	CO ₂	CH ₄	N ₂ O	Total	Mg CO ₂ e/Mg protein	Net change in Mg CO ₂ e/Mg protein from baseline
CO ₂ equivalents (Mg)							
1	Baseline	-484.09	494.04	295.94	305.90	1.87	0
2	Include fallow at 7% of current farm size	-408.29	494.04	286.63	372.38	2.42	0.56
3	Cut fertilizer rate by half	-520.79	494.04	227.21	200.47	2.09	0.23
4	Apply zero fertilizer to forages and pasture	-532.83	494.04	209.36	170.57	1.84	-0.02
5	Apply half of manure in fall; half in spring	-484.09	494.04	294.63	304.58	1.86	-0.01
6	Apply all manure in spring	-484.09	494.04	293.31	303.26	1.85	-0.02
7	Use an alfalfa-grass pasture	-484.05	453.78	294.89	264.61	1.47	-0.39
8	Feed a low quality forage	-477.10	551.72	269.24	343.86	2.95	1.09
9	Increase grazing season	-476.87	489.08	312.09	324.30	1.98	0.11
10	Put fats in finishing diets	-484.09	489.61	296.49	302.01	1.88	0.02
11	Put ionophores in finishing diets	-483.63	493.28	294.36	304.02	1.84	-0.03

9.2 APPENDIX II

9.2.1 Calculation of 2006 soil N₂O emissions

Steps 1 – 8 were done for each treatment.

Step 1: The 2006 N₂O-N emissions data were in units of $\mu\text{g N}_2\text{O-N/m}^2/\text{s}$, where m^2 is square metres and s is seconds. Each measurement was multiplied by 3600 s/h to get the emissions in $\mu\text{g N}_2\text{O-N/m}^2/\text{h}$, where h is hour.

Step 2: As there were two replications of each treatment (control, full and split) for each sampling interval, the average N₂O-N emission of the two replications for each treatment was calculated for each sampling interval, to give one N₂O-N emission for each sampling interval.

Step 3: For each sampling interval, the N₂O-N emission was assumed to be the value at the time in the middle of the x and x+1 sampling intervals. Emissions were converted to $\mu\text{g N}_2\text{O-N/m}^2$ using the following equation:

$$\mu\text{g N}_2\text{O-N/m}^2 = h * \mu\text{g N}_2\text{O-N}/(\text{m}^2 * \text{h}) \quad (\text{A2. 1})$$

where h is the number of hours from the middle of sampling interval x to the middle of sampling interval x+1.

Step 4: The emissions were then converted to $\text{mg N}_2\text{O-N/m}^2$. The $\text{N}_2\text{O-N}$ emissions from each sampling interval were summed to give the total $\text{N}_2\text{O-N}$ emissions ($\text{mg N}_2\text{O-N/m}^2$).

Step 5: The total $\text{N}_2\text{O-N}$ emissions ($\text{mg N}_2\text{O-N/m}^2$) were multiplied by the treatment area ($162,000 \text{ m}^2$ for control; $81,000 \text{ m}^2$ for full and split) to give total $\text{N}_2\text{O-N}$ emissions ($\text{mg N}_2\text{O-N}$).

Step 6: The $\text{N}_2\text{O-N}$ emissions ($\text{mg N}_2\text{O-N}$) were converted to N_2O (mg).

Step 7: The N_2O emissions were then converted to $\text{Mg N}_2\text{O}$. Emissions of $\text{Mg N}_2\text{O}$ were converted to $\text{Mg CO}_2\text{e}$, using a CO_2e of 296.

Step 8: The N_2O emissions, in terms of $\text{Mg CO}_2\text{e}$, were divided by the number of hectares in the treatment (control: 16.2 ha; full and split: 8.1 ha), to give $\text{Mg CO}_2\text{e/ha}$.

9.2.2 Calculation of 2006 soil CH_4 emissions

The 2006 soil CH_4 emissions were calculated using steps 1 – 8 above, substituting $\text{CH}_4\text{-C}$ and CH_4 for $\text{N}_2\text{O-N}$ and N_2O , respectively, and a CO_2e of 23 for 296.

Note: The calculated 2006 soil emissions of N_2O and CH_4 correspond closely with those in Ominski et al. 2008).

9.3 APPENDIX III

9.3.1 Calculation of emission factors for N₂O release from urine patches on soil in hayed paddocks

Step 1: Ominski et al. (2008) created treatments of synthetic urine differing in nitrogen concentration to represent the urine of animals grazing pastures with different forage nitrogen contents. The synthetic urine was applied to the Hytek soil in patches (1 litre of urine applied to approximately 0.05 m²), to represent urination events by the grazing cattle. The treatments used in this study were the Low and High synthetic urine treatments. The Low synthetic urine treatment had a low urine nitrogen concentration (4.4 g N/litre urine), and was intended to represent the Hytek Control treatment. The High synthetic urine treatment had a high urine nitrogen concentration (10.66 g N/litre urine), and was intended to represent the Full and Split Hytek treatments. Ominski et al. (2008) measured N₂O emissions from five replicates of each of the synthetic urine treatments. Four gas samples were collected in 15 minute intervals from each chamber every time samples were taken. Replication 3 was removed because of an error with treatment identification. N₂O emissions were reported by Ominski et al. (2008) in µg N₂O-N/(m²/h).

Steps 2 – 5 were done for both the Low and High synthetic urine treatments.

Step 2: For each sampling interval, the N₂O emission was assumed to be the value at the time in the middle of the x and x+1 sampling interval. The N₂O emissions from each of the four replications were averaged to give one N₂O measurement per sampling interval. Emissions were converted to µg N₂O-N/m² using the following equation:

$$\mu\text{g N}_2\text{O-N/m}^2 = h * \mu\text{g N}_2\text{O-N}/(\text{m}^2 * h) \quad (\text{A3. 1})$$

where h is the number of hours from the middle of sampling interval x to the middle of sampling interval x+1.

Step 3: The N₂O emissions from each sampling interval were summed to give the total N₂O emissions (µg N₂O-N/m²). The total N₂O emissions were converted to g N₂O-N/m².

Step 4: The amount of urine N applied to the soil (g N /m²) was calculated using the following equation:

$$\text{g N /m}^2 = (\text{L/m}^2) * (\text{g N /L}) \quad (\text{A3. 2})$$

where L/m² is the amount of urine applied to the soil and g N/L is the nitrogen concentration of the synthetic urine for that treatment.

Step 5: The emission factor for N₂O-N production (g N₂O-N/g N) from urine patches was calculated using the following equation:

$$\text{g N}_2\text{O-N/g N} = (\text{g N}_2\text{O-N/m}^2) \div (\text{g N/m}^2) \quad (\text{A3. 3})$$

9.3.2 Calculation of emission factors for N₂O emission from a sample of feces deposited by grazing animals

Step 1: Ominski et al. (2008) collected fecal samples from each of the Hytek treatments. A 1.05 kg (wet weight) fecal sample was placed on the hayed paddock of the same treatment that the sample originated from, and emissions were measured from two replications per treatment. The fecal samples for each treatment were placed into gas collection collars of the same size. On sampling days, four gas samples were taken at 15 minute intervals, and emissions were taken recorded as $\mu\text{g N}_2\text{O-N}/(\text{m}^2/\text{h})$. The nitrogen concentration of feces from each treatment was different (control treatment: 16.7 g N/kg feces; split treatment: 20.6 g N/kg feces; full treatment: 25.1 g N/kg feces) (Ominski et al. 2008).

Steps 2 – 7 were done for each treatment.

Step 2: As the amount of manure applied per chamber (chr) was known, we scaled the emissions reported by Ominski et al. (2008) down from m^2 to the area of the chamber (323.65 cm^2), to give $\mu\text{g N}_2\text{O-N}/(\text{chamber}/\text{h})$.

$$\mu\text{g N}_2\text{O-N}/(\text{chr}/\text{h}) = \mu\text{g N}_2\text{O-N}/(\text{m}^2*\text{h}) * 0.03 \quad (\text{A3. 4})$$

where the chamber area is 3% of the reported area (m^2).

Step 3: The wet weight (kg) of the fecal sample given by Ominski et al. (2008) was converted to dry matter based on the moisture content (%) of the manure from each Hytek treatment, using the following equation:

$$\text{dry matter (kg)} = 1.05 \text{ kg} * (1 - m) \quad (\text{A3. 5})$$

where 1.05 kg is the wet weight of the manure sample in each treatment and m is the moisture content of the manure sample, which is specific for each Hytek treatment Ominski et al. (2008).

Step 4: For each sampling interval, the N_2O emission was assumed to be the value at the time in the middle of the x and $x+1$ sampling intervals. The N_2O emissions from the two replications were averaged to give one N_2O measurement per sampling interval. Emissions were converted to $\mu\text{g N}_2\text{O-N/chamber}$ using the following equation:

$$\mu\text{g N}_2\text{O-N/chr} = h * (\mu\text{g N}_2\text{O-N/(chr*h)}) \quad (\text{A3. 6})$$

where h is the number of hours from the middle of sampling interval x to the middle of sampling interval $x+1$.

Step 5: The N_2O emissions from each sampling interval were summed to give the total N_2O emissions ($\mu\text{g N}_2\text{O-N/chr}$). The total N_2O emissions were converted to $\text{g N}_2\text{O-N/chr}$.

Step 6: The amount of N applied per chamber (g N /chr) was calculated using the following equation:

$$\text{g N /chr} = (\text{kg feces /chr}) * (\text{g N/kg feces}) \quad (\text{A3. 7})$$

where kg feces/chr is the amount of feces (dry matter) applied per chamber and g N/kg feces is the dry matter nitrogen concentration of the feces for that treatment.

Step 7: The emission factor for N₂O-N production (g N₂O-N/g N) from feces was calculated using the following equation:

$$\text{g N}_2\text{O-N/g N} = (\text{g N}_2\text{O-N/chr}) \div (\text{g N/chr}) \quad (\text{A3. 8})$$

9.3.3 Calculation of net N₂O emissions produced from urine patches on soil in hayed paddocks

Steps 1 – 5 were done for each Hytek treatment.

Step 1: Animal protein intake (g/head/day) was calculated using the following equation:

$$\text{protein intake} = \text{DMI} * \text{CP} \quad (\text{A3. 9})$$

where DMI is dry matter intake (kg/head/day) and CP is the protein content of the forage (%/100).

Step 2: Animal protein intake (g/head/day) was divided by 6.25 to get the nitrogen intake (NI) (g N/head/day). NI was then put into equation (4) from Kebreab et al. (2001) to calculate the amount of N excreted in the urine (urinary N) (g N/head/day).

$$\text{urinary N} = -0.70 + 0.30 * \text{NI} \quad (\text{A3. 10})$$

Step 3: Total N excretion in the urine (g N) was calculated using the following equation:

$$\text{Total N excretion} = \text{urinary N} * \# \text{ head} * 82 \quad (\text{A3. 11})$$

where # head is the average stocking density of the treatment and 82 is the number of days in the grazing season.

Step 4: Total N₂O-N emissions (g N₂O-N) for each treatment were calculated using the following equation:

$$\text{g N}_2\text{O-N} = \text{total N excretion (g N)} * \text{g N}_2\text{O-N/g N} \quad (\text{A3. 12})$$

Step 5: Emissions of N₂O-N were converted to N₂O and then to Mg CO₂e, using a CO₂e of 296.

9.3.4 Calculation of net N₂O emissions produced from a sample of feces deposited by grazing animals

The N₂O emissions from feces were calculated using the same steps as the N₂O emissions from urine, with one difference. The equation used to calculate urinary N was replaced with equation (5), also from Kebreab et al. (2001), which calculates fecal N excretion.

$$\text{fecal N} = 32.0 + 0.28 * \text{NI} \quad (\text{A3. 13})$$

9.4 APPENDIX IV

9.4.1 Calculation of emission factors for CH₄ emission from a sample of feces deposited by grazing animals

Steps 1 – 5 are the same as Steps 1 -5 in “Calculation of emission factors for N₂O emission from a sample of feces deposited by grazing animals” of Appendix III, substituting CH₄-C for N₂O-N.

Steps 3 – 6 were done for each Hytek treatment.

Step 6: The emission factor for CH₄-C production (g CH₄-C/kg feces) from feces was calculated using the following equation:

$$\text{g CH}_4\text{-C/kg feces} = (\text{g CH}_4\text{-C/chr}) \div (\text{kg feces/chr}) \quad (\text{A4. 1})$$

9.4.2 Calculation of net CH₄ emissions produced from a sample of feces deposited by grazing animals

Steps 1 - 3 were done for each Hytek treatment.

Step 1: The amount of volatile solids (VS) excreted in the manure (kg) by the cattle over the course of the grazing season is calculated using the following equation:

$$\text{VS} = [\text{DMI} * (1 - \text{TDN}) * (1 - \text{ASH})] * \# \text{ h} * 82 \quad (\text{A4. 2})$$

where DMI is the dry matter intake of the cattle (kg/head/day), TDN is the total digestible nutrients of the feed (%/100), ASH is the ash content of the

manure (% DM), # h is the average stocking density of the treatment and 82 is the number of days in the grazing season.

Step 2: The volatile solids constitute only 84.6% of total solids (TS) (total DM of feces produced). Therefore, TS (kg) were calculated using the following equation:

$$TS = VS / 0.846 \quad (A4. 3)$$

Step 3: Total CH₄-C emissions (g CH₄-C) for each treatment were calculated using the following equation:

$$g \text{ CH}_4\text{-C} = TS * (g \text{ CH}_4\text{-C/kg feces}) \quad (A4. 4)$$

Step 4: Emissions of CH₄-C were converted to CH₄ and then to Mg CO₂e, using a CO₂e of 23.

9.5 APPENDIX V

9.5.1 Calculation of CO₂ emissions from energy use during land application of liquid hog manure using a slurry wagon system

Steps 1 - 3 were done for only the full and split Hytek treatments, as the control treatment had zero energy use as it did not receive any liquid hog manure.

Step 1: The IPCC (2006) effective CO₂ emission factor for gas/diesel oil was converted from kg CO₂/TJ to kg CO₂/MJ, giving an emission factor of 0.0741 kg CO₂/MJ.

Step 2: Energy use per treatment (MJ) was calculated using the following equation:

$$\text{MJ/treatment} = \# \text{ ha} * (\text{MJ/ha}) \quad (\text{A5. 1})$$

where # ha is the number of hectares in the treatment and MJ/ha is the energy use per ha.

Step 3: Carbon dioxide emissions released from energy use were calculated using the following equation:

$$\text{CO}_2 \text{ (kg)} = (\text{MJ/treatment}) * (\text{kg CO}_2/\text{MJ}) \quad (\text{A5. 2})$$

9.6 APPENDIX VI

9.6.1 Calculation of CH₄ emissions from liquid hog manure storage

Step 1: The total manure stored (kg) was calculated by multiplying the manure application rate (kg/ha) by the area of the pasture (ha) (it was assumed that all manure in storage was used to fertilize the pastures at the research site).

Step 2: The total solids (TS) applied (kg DM) was calculated using the following equation:

$$\text{TS (kg DM)} = \text{total manure applied (kg)} * (1 - (\text{mc}/100)) \quad (\text{A6.1})$$

where mc is the moisture content of the hog manure (%).

Step 3: The total volatile solids (VS) applied (kg) was calculated by multiplying the total solids content of the manure by the percent of VS in the manure.

Step 4: I assumed that an equal amount of the VS was added to the manure storage each month. In the 'full' treatment, it was assumed that the manure storage was emptied once a year, in mid-May. In the 'split' treatment, it was assumed that the manure storage was emptied twice per year, once in mid-May and again in mid-October. The timing of land application, and emptying of manure storage, corresponds with the timing of manure application at the research site. The annual amount of CH₄ (m³) produced from liquid hog manure storage was calculated separately for the 'full' and 'split' treatments

using the methodology outlined in Vergé et al. 2006. In the calculations, I used a monthly VS addition of 1/12 of the total VS applied (calculated in the previous step), a maximum CH_4 producing capacity of the manure of 0.48 (IPCC 2006), and average monthly temperatures from (Marshall et al. 1999).

Step 5: The annual CH_4 production (m^3) determined in Step 4 was multiplied by 0.67 to convert annual CH_4 production from volume to kg of CH_4 .

Step 6: Emissions of CH_4 were converted to Mg CO_2e using a CO_2e of 23.

9.6.2 Calculation of N_2O emissions from liquid hog manure storage

Step 1: The total nitrogen (N) stored (kg) was calculated by multiplying the N application rate (kg/ha) by the area of the pasture (ha) (it was assumed that all manure in storage was used to fertilize the pastures at the research site).

Step 2: Nitrous oxide emissions from storage of liquid hog manure were calculated using emission factors from the IPCC (2006). The emission factor for direct N_2O emissions from liquid hog manure without a natural crust cover is zero. The fraction of N leached from liquid manure storage is zero, therefore indirect N_2O emissions from leaching are zero. The IPCC (2006) estimates that 48% of stored N is lost via volatilization. The emission factor for the volatilization of manure N is 0.01. Monthly N_2O emissions from hog manure storage are calculated using the following equation:

$$\text{monthly N}_2\text{O (kg N}_2\text{O-N)} = \text{N in storage (kg)} * R * EF \quad (\text{A6.2})$$

where N is storage is the amount of N in the manure storage (kg), R is the fraction of N volatilized and EF is the emission factor for volatilization of manure.

Step 3: I assumed that an equal amount of the total N applied was added to the manure storage at the end of each month, which was 1/12 of the total N applied (from Step 1). The assumed amount of manure N added to the manure storage each month was greater than 1/12 of the total N applied, as 48% of the added N was lost via volatilization each month. This was necessary to ensure that the required amount of N was available in the manure storage for land application. In the ‘full’ treatment, it was assumed that the manure storage was emptied once a year, in mid-May. In the ‘split’ treatment, it was assumed that the manure storage was emptied twice per year, once in mid-May and again in mid-October. The timing of land application, and emptying of manure storage, corresponds with the timing of manure application at the research site.

Step 4: The monthly manure storage N₂O emissions were summed to give an annual estimate of emissions.

Step 5: Emissions of N₂O-N were converted to N₂O and then to Mg CO₂e, using a CO₂e of 296.