MODELLING GREENHOUSE GAS EMISSIONS IN CATTLE: FROM RUMEN TO THE WHOLE-FARM

By

Aklilu Wake Alemu

A Thesis

Submitted to the Faculty of Graduate Studies

of the University of Manitoba

In Partial Fulfilment of the Requirement

For the degree of

DOCTOR OF PHILOSOPHY

Department of Animal Science

University of Manitoba

Winnipeg, Manitoba

Canada, R3T 2N2

DEDICATION

I dedicated this work to my mother Aynalem Fitawta, my wife Etsegenet Asfaw, and my siblings Kuri, Getu, Haregeweyen, Mebrate, Yeshi and Girma.

ACKNOWLEDGEMENTS

I am grateful to acknowledge many people that take parts in the course of my study and those who make a difference in my life as well as my future endeavor. The completion of this thesis would not be possible without the contribution of these people in a various ways

First of all, I would like to thank my supervisors Drs. Ermias Kebreab and Kim Ominski for their support, guidance and encouragement throughout the course of my graduate degree program. Their intelligence and optimism in research inspired me with enthusiasm and the will to succeed in research and academic endeavor. Special thanks to Dr. Mario Tenuta and Dr. Kees Plaizier for being my committee member and providing their comments and advice on my projects. My great appreciation goes to Dr. Luis Orlindo Tedeschi, my external examiner, for taking time in reviewing and providing comments on my thesis.

I would like to thank MITACS ACCELERATE and University of Manitoba Graduate Student Fellowship Program for funding my projects and sponsoring my study.

I would also like to extend my sincere and utmost appreciation for my fellow graduate students, academic, support, technical and research staff at the department of Animal Science for their company and willingness to help in any support I need in the course of my program.

I am always grateful for Dr. Getahun Legesse and Alexander Yitbarek, for their friendship, support and encouragement throughout my program. Many thanks to Elyas, Harold, Michel, Sean, Mohammed and Hamidreza for the great time we had in the gym and basketball court.

My great appreciation also goes to my family who supported me throughout my life. I would like to thank my mother for raising me the best way she can and installing in me all the values of education and hard work. I am thankful for the contribution of my brothers, sisters,

nieces and nephews. Last but not least, I would like to thank my beloved wife Etegenet Worku Asfaw who always there for me and giving me advice and encouragement.

Finally, to God Almighty, I am only here today because of you. Your guidance and love made all of these possible.

FORWARD

This thesis is prepared in a manuscript format containing three manuscripts. Manuscript I is published in Animal Feed Science and Technology, and the authors in this manuscript are A. W. Alemu, J. Dijkstra, A. Bannink, J. France and E. Kebreab. Manuscript II is published in Canadian Journal of Animal Science, and the authors in the manuscript are A. W. Alemu, K. H. Ominski and E. Kebreab. Manuscript III is in preparation to be submitted to Agricultural Systems. The authors in the manuscript are A. W. Alemu, K. H. Ominski, M. Tenuta and E. Kebreab. All the manuscripts are formatted according to the guidelines of Canadian Journal of Animal Science.

ABSTRACT

Mathematical modeling in animal agriculture can be applied at various levels including at the tissue, organ, animal, farm, regional and global levels. The purposes of this research were i) to evaluate models used to estimate volatile fatty acid (VFA) and methane (CH_4) production and assess their impact on regional enteric CH_4 inventory, and ii) to develop a process-based, whole-farm model to estimate net farm GHG emissions. In the first study, four VFA stoichiometric models were evaluated for their prediction accuracy of rumen VFA and enteric CH_4 production. Comparison of measured and model predicted values demonstrated that predictive capacity of the VFA models varied with respect to the type of VFA in rumen fluid which impacted estimated enteric CH_4 production. Moving to a larger scale assessment, we examined the enteric CH_4 inventory from Manitoba beef cattle (from 1990 to 2008) using two mechanistic rumen models that incorporate VFA stoichiometric models: COWPOLL and MOLLY, and two empirical models: Intergovernmental Panel on Climate Change (IPCC) Tier 2 and a nonlinear equation (Ellis). The estimated absolute enteric CH_4 production varied among models (7 to 63%) indicating that estimates of GHG inventory depend on model selection. This is an important consideration if the values are to be used for management and/or policy-related decisions. Development of models at the individual farm component level (animal, soil, crop) does not accurately reflect net GHG emissions generated from the whole production system. We developed a process-based, whole-farm model (Integrated Components Model, ICM), using the existing farm component models COWPOLL, manure-DNDC and some aspects of IPCC to integrate farm components and their associated GHG emissions. Estimates of total farm GHG emissions and their relative contribution using the ICM were comparable to estimates using two

other whole-farm models (Integrated Farm System Model and Holos model). Variation was observed among models both in estimating whole-farm GHG emissions and the relative contribution of the different sources in the production system. Overall, whole-farm models are required to explore management options that will mitigate GHG emissions and promote best management practices. However, for full assessment of the production system, other benefits of the system (*e.g.*, carbon sequestration, ecosystem services), which are not part of current whole-farm models, must be considered.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF ABBREVIATIONS

Ac	Acetate
ADF	Acid detergent fiber
ADG	Average daily gain
ATP	Adenosine triphosphate
Bc	Branched chain volatile fatty acid
BMP	Benificial management practice
Bu	Butyrate
BW	Body weight
C	Carbon
C_b	Bias correlation factor
CCC	Concordance correlation coefficient
CH_4	Methane
CNCPS	Cornell Net Carbohydrate and Protein System
CO_2	Carbon dioxide
CO_2 eq	Carbon dioxide equivalent
CP	Crude protein
DM	Dry matter
DMI	Dry matter intake
DNDC	Denitrification-Decomposition model
EE	Ether extract
EF	Emission factor

GE	Gross energy
GEI	Gross energy intake
GHG	Greenhouse gas
H_2	Hydrogen
IFSM	Integrated Farm System Model
ICM	Integrated Components Model
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardization
LCA	Life cycle analysis
MAFRI	Manitoba Agriculture, Food and Rural Initiative
Mcal	Mega calorie
MSPE	Mean square prediction error
N	Nitrogen
N_2O	Nitrous oxide
NADH	Nicotinamide adenosine dinucleotides
NDF	Neutral detergent fiber
NFC	Non fiber carbohydrate
NH_4^+	Ammonium
NH_3	Ammonia
NO_3^-	Nitrate
NRC	National Research Council
O_2	Oxygen

OM	Organic matter
Pr	Propionate
RMSPE	Root mean square prediction error
SF_6	Sulfur hexafluoride
TDN	Total digestible nutrient
VFA^-	Dissociated (anionic) form of volatile fatty acid
VFA	Volatile fatty acid
VFAH	Undissociated form of volatile fatty acid
Y_m	Methane conversion rate (%GEI)

1. GENERAL INTRODUCTION

As the world population grows and is expected to reach 9 billion by 2050, the demand for food, crop and livestock products, will also be doubled relative to the current year (Alexandratos 2009; Thornton 2010; O'Mara 2011). Consumption of livestock products has increased substantially in the past decade with exponential growth in areas such as Latin America and Asia (Gerber and Steinfeld et al. 2008; Alexandratos 2009). In order to meet the demand, the livestock sector needs to increase milk and meat production. It is well documented that livestock production emits a significant quantity of GHGs that contribute to global warming (Steinfeld et al. 2006; O'Mara 2011). Climate change mitigation has become an international policy (United Nations 1998; IPCC 2006) which makes increasing production (and therefore emissions) at odds with the obligation to reduce greenhouse gas (GHG) emissions. An international team of researchers (Foley et al. 2012) who characterized and documented changes in agricultural lands and their yield over the past 40 years proposed a five point plan for feeding the world while protecting the planet, such as, halting farmland expansion, realizing yield potential, using farm inputs more strategically, balancing diets to meet nutrient requirements and reducing waste. Increasing agricultural resource use efficiency and accelerating technological changes may be potential pathways to implement the proposed plans and help to achieve the world`s demand for food while minimizing GHG emissions (Gerber and Steinfeld, 2008).

The three primary GHGs produced by livestock agriculture are methane (CH_4), nitrous oxide (N_2O) and carbon dioxide (CO_2). Emissions of CH_4 are generated mainly from enteric fermentation during a normal process of feed digestion and to a lesser extent from manure storage (IPCC 2006; Moss et al. 2000). Methane emitted from ruminant livestock production

accounts for about one quarter of the global anthropogenic CH_4 (Lassey 2008). Nitrous oxide emissions originate from soils as a result of nitrification and denitrification of nitrogen (N) (Norton 2008), and from leaching, runoff and volatilization of N (IPCC 2006; Well and Butterbach-Bahl 2010**)**. Carbon dioxide emissions generated from on- and off-farm energy use also contributes to total farm GHG emissions. Animal agriculture contributes 8-11% to the global anthropogenic GHG emissions (O'Mara 2011) and could increase to 18% when the analysis incorporated emissions from land use changes in the system (Steinfeld et al. 2006; Gill et al. 2010). Due to the significant contribution of emissions from animal agriculture, several governments have implemented policies to accurately quantify and reduce GHG contributions from animal agriculture (United Nations 1998).

Mathematical modelling has been used to quantify GHG emissions associated with food production (France and Thornely 1984; Thornely and France 2007; Dumas et al. 2008). The types of models developed range from simple empirical equation relating the inputs and outputs of a system to complex dynamic mechanistic models developed according to biological hierarchy (Thornely and France 2007). These models play a key role in ruminant feed evaluation (National Research Council - NRC 2001), quantification of nutrient utilization by ruminants (Kebreab et al. 2009) and environmental consequences of ruminant nutrition (Bannink et al. 2010; Ellis et al. 2011). Given the difficulty and cost of conducting experiments, quantification and prediction of GHG emissions from animal agriculture and development of beneficial management practices (BMPs) to minimize emissions (at animal and farm level) may be achieved through the use of mathematical models. The research studies presented in this thesis addresses the role of mathematical models in quantification of enteric CH_4 production and net farm GHG emissions from Canadian beef cattle production systems.

2. LITERATURE REVIEW

2.1. Mathematical Modelling in Ruminant Nutrition

Mathematical modelling can be defined as "the use of equations to describe or simulate processes in a system which inherently applies knowledge and is indispensible for science and society, especially agriculture" (Dumas et al. 2008). Furthermore, Thornley and France (2007) described a mathematical model as "an equation or a set of equations which represents the behaviour of a system". The levels of aggregation upon which the models were developed can vary from the relatively micro-level of tissue and organ such as the rumen to the macro-level of farms and geographical regions. These models play a prominent role in ruminant nutrition, from quantifying nutrient utilization (Kebreab et al. 2006a; France and Kebreab 2008) to setting feeding standards (*e.g.*, NRC, Cornell Net Carbohydrate and Protein Systems, CNCPS).

Some of the general objectives of mathematical modelling in animal agriculture may include (Thornley and France, 2007);

- Integration of knowledge from different field of study (*e.g.*, mathematics, biology, chemistry, economics) to explain complex mechanisms,
- Reduction in the cost of conducting research,
- Organization and description of past quantitative observations,
- Estimation of inventories and conducting predictions for management and policy making decisions,
- Identification of existing gaps in knowledge and directions for further research,
- Assessment of the social, economic and environmental impacts of animal agriculture,

- Development of economically as well as environmentally sound beneficial management practices (BMPs) to minimize contributions of greenhouse gas (GHG) emissions from livestock agriculture to increase its environmental sustainability.

2.1.1. Classification of models

A scheme for classifying models that is widely recognized as a standard and is extensively used by modelers in the field of agriculture and environmental systems has been described by France and Thornley (1984). Models can be classified as; (i) empirical vs. mechanistic, (ii) dynamic vs. static, (iii) deterministic vs. stochastic, and (iv) continuous vs. discrete.

Empirical modelling is essentially a direct description of observational and experimental data. It uses existing data to describe the relationship of observations between one or two variables. In the organizational hierarchy (*e.g.*, tissue, organ, organism), empirical modeling based at a single level describes the behavior of a specific level in terms of the attributes of that level alone without regard to any biological theory. In empirical modelling, mathematical relationships developed are unconstrained by physical laws (*i.e.*, energy conservation or the laws of thermodynamics), by biological information or knowledge of the structure of the system (France and Kebreab 2008). Empirical modelling is often a curve-fitting exercise. If the developed models fit the data well, the equations are useful under the particular conditions for which the data was generated (France and Kebreab 2008). These models may provide a practical tool, however, their inability to incorporate biological components as well as the need for mechanistic explanations has forced researchers to seek models that integrate the underlying biological mechanism using mechanistic explanations.

Mechanistic models are complex, process-based which are constructed on (at least) two organizational levels (Dijkstra et al. 2005; France and Kebreab 2008). As the number of

organizational levels included in the model increase, so does the model complexity and thus it is advisable not to use more than two levels. Mechanistic modelling employs the approach of scientific reductionism, *i.e.*, the description of the system at the upper level (*e.g.*, organism) can be constructed based on the components and their associated process at the lower level (*i.e.*, organ).

Dynamic models incorporate time explicitly and they are generally presented as a set of differential equations with time as an independent variable. Most models in agriculture and ecology are dynamic, describing the time course of events (Dijkstra et al. 2005). Conversely, static models do not contain time as a variable and do not make time-dependent predictions. For example, feed evaluation models are static as they predict the nutrient requirement of an animal of known body weight, sex and production level at a specific time (Tedeschi et al. 2005).

Stochastic models include probabilistic element(s), giving a distribution of outputs to a given set of inputs. Deterministic models, however, provide exact solutions derived from the equation or set of equations. The use of the term deterministic in animal science implies that the solution applies specifically to the average animal in a population (France and Kebreab 2008). Continuous models represent time continuously while in discrete models, time is an integer.

Dynamic mechanistic models that appear in the animal science literature are based on a system of ordinary differential equations. These models are represented using a standard mathematical representation called the rate:state formalism (rate of process = function of state of system) (Thornley and France 2007). According to the formalism, the system under investigation is defined at time t by q state variables (x_1, x_2, …, x_q) that represent properties or attributes of the system (*e.g.*, quantity of substrate, organ or tissue mass). The models then contain q first-order differential equations which describe how the state variables change with time and written as:

$dx_i/dt = f_i(x_1, x_2, \ldots, x_q; P)$; $i = 1, 2, \ldots, q$

where P denotes set of parameters and the function f_i denotes the rate of change of the state variable x_i. The function f_i has the dimension of state variable per unit time and comprises terms which represent the rate of component processes (*e.g.,* substrate utilization). In these types of modelling, differential equations are constructed by direct application of the law of mass conservation and solved using analytical or numerical solutions.

2.2. Overview of Rumen Functions and Modelling Rumen Functions

The fermentation process in the rumen is a complex process involving microbial activity and degradable dietary components. Representing this process using mathematical model is also complex and requires knowledge of rumen microbial consortia, digestion kinetics production and metabolism of VFAs and CH_4 production.

2.2.1. Volatile fatty acids

Volatile fatty acids are produced during fermentation of dietary carbohydrates, protein and fat in the rumen and hind gut. The ability of rumen microbes (bacteria, archaea, protozoa, fungi) to convert fiber and non-protein nitrogen compounds to VFAs and microbial protein, which can be utilized by the host, is one reason for the great evolutionary success of ruminants. About 70 – 80% of the ruminant animal`s total caloric requirement is provided by VFAs (Bergman 1990; Dijkstra 1994). The predominant VFAs in the rumen fluid are acetate (Ac), propionate (Pr), butyrate (Bu), isobutrate, valerate, isovalerate, 2-methylbutrate with others usually found in relatively small amounts. Acetate, Pr, and Bu account for more than 95% of VFA produced (Bannink et al. 2006). The pathways for VFA production in the rumen are summarized in Figure 2.1. Before being fermented to VFA, carbohydrates are hydrolyzed to their constituents, hexoses or pentoses. Hexose is metabolized, almost exclusively, in the Embden-Meyerhof-Parnas

pathway into pyruvate, yielding reduced co-factor (nicotinamide adenosine dinucleotides (NADH)) and adenosine tri phosphate (ATP, Fahey and Berger 1988; Moss et al. 2000). In order to complete the fermentation of sugar and maintain redox balance in the rumen, the reduced co-factor (NADH) must be reoxidized to NAD^+ through electron acceptors other than oxygen (CO_2, sulphate, nitrate, fumarate) (Hungate 1966; Bergman 1990). In the process, hydrogen (H_2), which is utilized by H_2-utilizing bacteria (*e.g.*, methanogenes), is produced as an end product. In situations when the produced H_2 is not properly utilized by rumen methanogens, NADH can be reoxidized by the dehydrogenase of fermenting bacteria to produce lactate and ethanol (McAllister et al. 1996; Moss et al. 2000, Figure 2.1). This pathway is typical for ruminants fed grain-based diets that contain rapidly fermentable carbohydrates.

In addition to dietary carbohydrates, dietary proteins and lipids are also fermented to produce VFA in the rumen. The contribution of lipid to VFA production is relatively minimal as inclusion rates in cattle diets are minimal. Branched chain VFAs, such as isobytric, isovaleric, and 2-methylbutyric acid are synthesized from branched chain amino acids, valine, leucine and isolucine, respectively (Moss et al. 2000).

The ratio of VFAs produced in the rumen is a function of the microbial population present in the rumen which are dependent upon the diet and type of carbohydrate fermented (Dijkstra et al. 2008). The Ac:Pr:Bu ratio in the rumen of cattle fed high forage diets is typically 70:20:10. High forage diets encourage the growth of Ac producing bacteria. Conversely, high grain diets, typically high in non-structural carbohydrates, favour Pr producing bacteria and thus, Pr production is increased at the expense of Ac (Bannink et al. 2006).

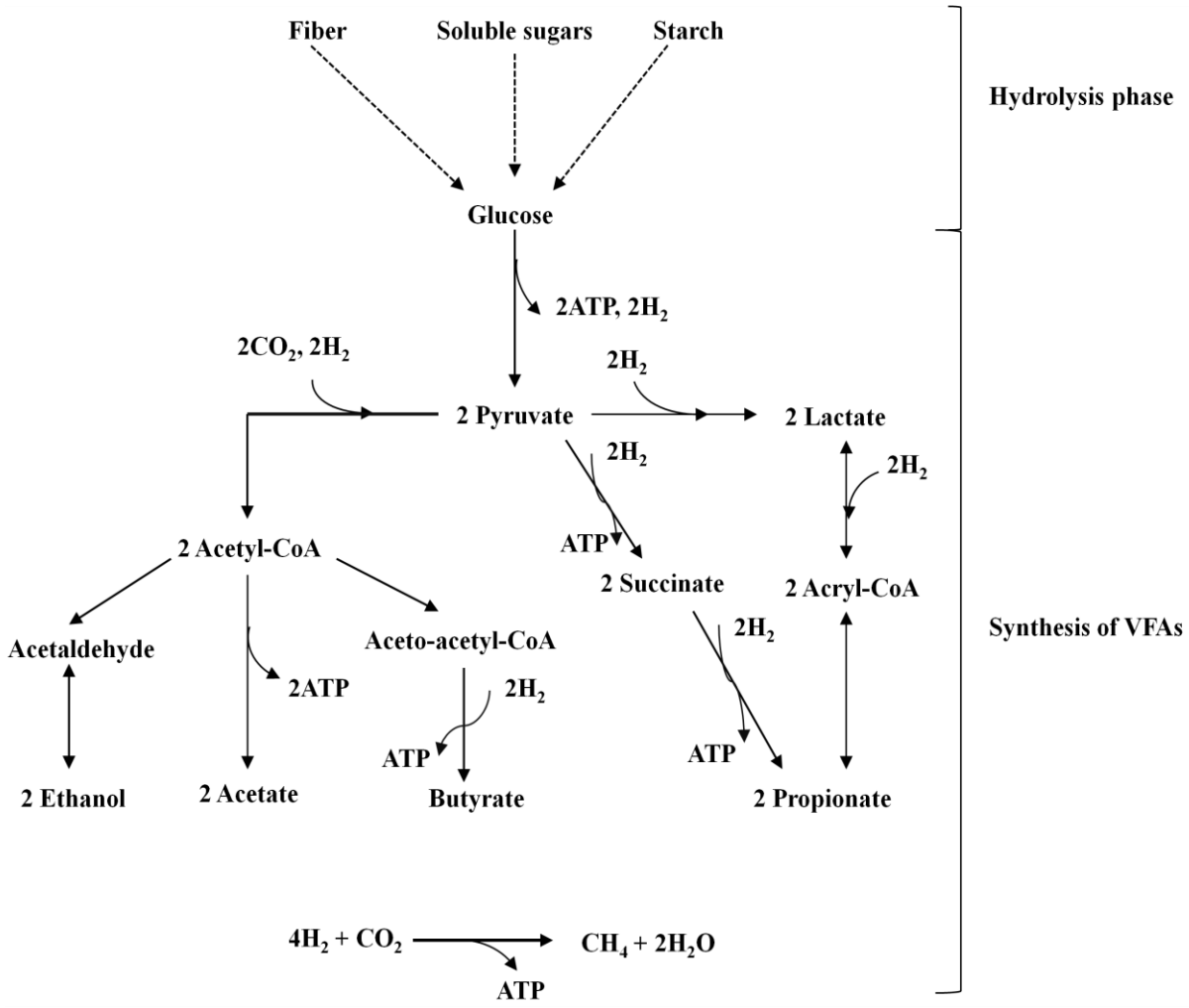

Figure 2.1. Major carbohydrate metabolism pathways in the rumen (modified from Moss et al. 2000 and Kohn and Boston 2000).

2.2.1.1. Absorption of volatile fatty acid from the rumen

The majority of VFAs produced in the rumen are absorbed across the rumen wall, although a small proportion (10 – 20% in sheep and 35% in dairy cows) pass to the omasum and abomasum and are absorbed from these organs (Bergman 1990; France and Dijkstra 2005). The fraction that passes from the rumen depends on the balance between the absorption rate and turnover rate of the liquid phase in the rumen (Huhtanen et al. 2008). Absorption of VFAs occurs by diffusion of undissociated acid (non-ionic form, VFAH) and anions (VFA^-) across the rumen wall (Dijkstra 2004; Aschenbach et al. 2011). The VFAH form is highly lipid-soluble and an efficient way to import acids with high lipophilicity (especially Bu) whereas the VFA^- form is less lipophilic and requires a specific active transport mechanism such as transport proteins (Aschenbach et al. 2011). Moreover, the VFA^-/HCO_3^- exchange is important for acids with less lipophilicity, such as Ac and is highly driven by CHO_3^- imported from the blood via Na^+/HCO_3^- co-tarnsport (Aschenbach et al. 2011).

During absorption of VFAs through the rumen wall, the rumen epithelium metabolizes small amounts of Pr (5 to 10%), most of the Bu (77%) and valerate (68%) but none of the Ac and isobutyrate (Kristensen et al. 2000a; b; Kristensen 2005). The ketone-body resulting from Bu metabolism provides the first four carbon units for short- and medium-chain fatty acid synthesis in the mammary gland. Propionate is the only VFA capable of being used for gluconeogenesis through conversion to oxaloacetate *en route* to glucose and accounts for 65-80% of the net glucose supply in lactating dairy cows (Reynolds et al. 2003). Acetate serves as an energy source and for lipogenesis in the mammary gland and adipose tissue (Bergman 1990; Reece 2004). Overall, the efficiency of VFA utilization in ruminants depends not only on the total VFA produced but also on their molar proportions.

In general, the profile of VFA formed in the rumen has both economic and environmental consequences. The efficiency of energy utilization, composition of animal products (milk and meat; Sutton 1985) and the amount of enteric CH_4 produced depend on the VFA profile in the rumen. Hence, understanding of the VFA profile in the rumen could enable us to manipulate the rumen ecosystem in order to enhance productivity while reducing its negative outcomes such as lactic acidosis, rumen dysfunction, ketosis, laminitis and enteric CH_4 production.

2.2.1.2. Measuring volatile fatty acid production in the rumen

There are two major conventional groups of techniques that have been used to quantify the rate of total and individual VFA production in the rumen: i) *in vivo* and ii) *in vitro* measurement techniques (France and Dijkstra 2005).

2.2.1.2.1. In vivo measurement technique

In vivo technique includes both non-tracer methods (*i.e.,* portal-arterial difference in VFA concentration and perturbation of the steady state) and tracer methods that use isotopes (radioactive isotope ^{14}C or stable isotope ^{13}C, Hungate 1966; France and Dijkstra 2005). The portal-arterial difference in VFA concentration is a technique that uses an estimate of VFA production obtained by measuring the amount of VFAs directly absorbed or appearing in the blood stream (portal vein; Barcroft et al. 1944). The difference between VFA concentration in the venous blood draining the rumen and that in arterial blood provides a measure of the amount entering the blood from the rumen. However, the method fails to account for individual VFAs metabolized in the rumen wall and by the rumen microbes on their way to the portal vein (Bergman 1975; Kristensen et al. 2000a, b). An alternative to this is the perturbation of the steady state method (Martin et al. 2001) in which the rate of production of VFA is calculated from the change in rumen VFA concentration when one or mixture of acids is infused (López et

al. 2003; France and Dijkstra 2005). The limitation of the technique is that the infused acids modify the rumen environment including rumen pH and rumen osmolality, which affect the fractional absorption rate of VFAs from the rumen and thus, the validity of this technique is questionable (France and Siddons 1993).

The non-tracer techniques have been replaced by tracer techniques where the isotope form of individual VFA or mixture of VFAs is administered into the rumen by continuous infusion at constant rate (Hegarty and Nolan 2007). The data is then interpreted using either a single-pool (total VFA) or an interchanging three-pool model (Ac, Pr, Bu) that considers the interconversion of carbon between individual VFAs (Bergman et al. 1965; Leng and Brett 1966). Although a steady state condition is normally assumed, interpretation of tracer data using dynamic modelling allows their adaptation to non-steady state condition (France and Dijkstra 2005). The rate:state equations used in the tracer technique depend on mass conservation principles (France et al. 1987). However, the tracer technique also has the following limitations: (i) it does not consider hindgut VFA production (Dijkstra 1994), and (ii) the technique incorporates the exchange of carbon among VFAs but the exchange of VFA carbon with the carbon pool of the rumen microbial system is much larger than represented by exchange between VFAs because rumen microbes metabolize significant fractions of Ac, Pr and isobutyrate produced in the rumen (Kristensen 2001; 2005).

2.2.1.2.2. In vitro measurement technique

The methodological problems caused by the complexity of VFA metabolism associated with *in vivo* measurements, and the higher material and analytical costs associated with the application of isotope dilution technique forced researchers to seek for an alternative technique of measuring VFA production. *In vitro* techniques of estimating VFA production have received attention in the

past (*e.g.*, Carroll and Hungate, 1954; Hume 1977; Corley and Murphy 2004). The method provides an opportunity to simulate rumen conditions and estimate the production of VFA, gas and microbial protein production from different diets using an "artificial rumen" (Carroll and Hungate 1954). The rate of production of individual or total VFAs is calculated from the increase in acid concentration obtained by incubating the samples for different periods. Compared to *in vivo* methods, the lack of VFA absorption in the *in vitro* technique makes the quantitative measurements of VFA easier, however, the technique is questionable in representing *in vivo* situations.

2.2.1.3. Prediction of volatile fatty acid production in the rumen

Due to the cost of performing routine fermentation experiments that require isotopes and the methodological constraints related to VFA production measurement techniques, efforts have been made to predict VFA production in the rumen using various modelling approaches. As such, different empirical and mechanistic VFA models have been developed (Dijkstra et al. 2008).

Empirical VFA model: these models assume that the composition of diet is related to the molar proportions of VFAs in the rumen and thus VFA molar proportion are predicted from feed chemical compositions (*e.g.*, Friggens et al. 1998). However, these models do not consider the digestibility differences among feed ingredients that affect VFA production (Dijkstra et al. 2008). Starch from corn, for example, is more resistant to rumen degradation compared to starch from barley and as such, a higher production of Pr occurs with barley (Sutton et al. 1985). In an attempt to incorporate digestibility, Nozière et al. (2010; 2011) developed an empirical model using meta-analysis of literature data which includes intake level, site and extent of digestion of OM, starch and neutral detergent fiber (NDF) as covariates. Rumen degradability of NDF, starch

and protein used in the model were estimated using *in sacco* techniques, which could question its application to *in vivo* situations.

Volatile fatty acid stoichiometry model: this approach considers the substrates (carbohydrate and protein) fermented in the rumen to be partly fermented and incorporated into the microbial biomass (Dijkstra et al. 2008). Stoichiometric coefficients are developed to describe the partitioning of C between each VFA (Ac, Pr, Bu and branched chain (Bc)) for each fermented substrate in the rumen. The assumptions made in VFA stoichiometric models include: (i) molar proportion of VFAs in the rumen fluid represents the proportions of VFAs produced, (ii) an identical fractional rate of absorption for all types of VFA, and (iii) rumen bacteria are considered as a single pool. However, several studies have questioned the validity of these assumptions (Sutton et al. 1985; Dijkstra et al. 1993; Dijkstra 1994; López et al. 2003).

The VFA stoichiometric modelling approach was first introduced by Koong et al. (1975) and later modified by Murphy et al. (1982), Bannink et al. (2006) and Sveinbjörnsson et al. (2006) with the aim of increasing prediction accuracy. Extant dynamic mechanistic whole-rumen models (*e.g.*, Dijkstra et al. 1992; Baldwin 1995; Danfær et al. 2006) use VFA stoichiometric models to estimate VFA production. The VFA models of Murphy et al. (1982) and Bannink et al. (2006) developed stoichiometric coefficients for five fermented substrates (soluble carbohydrate, starch, protein, cellulose and hemicelulose) from roughage and concentrate diets to estimate the corresponding VFA production.

Another mechanistic approach of estimating VFA production based on application of thermodynamics laws has also been introduced (Nagorcka et al. 2000; Kohn and Boston 2000; Offner and Sauvant 2006). Unlike the previous stoichiometric VFA modelling approach, the fermentation coefficients of thermodynamics approach are based on both the type of fermented

substrates and the different fermentation pathways characterizing the different microbial groups in the rumen (amylolytic and fibrolytic bacteria and protozoa). Evaluation of the model by Offner and Sauvant (2006) suggest that even though the predicted VFA profile was satisfactory, simulations of parameters such as pH and redox potential were less reliable. They acknowledged that in addition to kinetics and population ecology, other driving forces need to be considered in the model.

2.2.2. Enteric methane

As described in Figure 2.1, the fermenting microorganisms hydrolyze protein, starch and plant cell-wall polymers in to VFAs, H_2, NH_3 and CO_2 and the rumen methanogens produce CH_4 by using H_2 and formate as substrates (McAllister et al. 1996). Formation of CH_4 (methanogenesis) in the rumen is the major way of H_2 elimination (McAllister et al. 1996; Moss et al. 2000). The collaboration between fermenting species and H_2-utilizing bacteria through 'interspecies H_2 transfer" prevents the accumulation of H_2 in the rumen (Ellis et al. 2008).

The majority of CH_4 in ruminants is produced in the rumen (87%) and the rest is produced in the large intestine (13%) (Torrent and Johnson 1994). The amount of CH_4 production in the rumen is a function of the VFA profile (Moss et al. 2000). The H_2 produced during the conversion of hexose to Ac or Bu is mainly utilized by methanogens. However, production of Pr is a competitive pathway of H_2 utilization that reduces enteric CH_4 production in the rumen because both Pr-producing and CH_4-producing organisms in the rumen utilize H_2 as a substrate (Ellis et al. 2008, Figure 2.1). Moss et al. (2000) reported a strong negative correlation ($r^2 = 0.77$) between enteric CH_4 and Pr production.

Despite its importance in rumen fermentation process through H_2 elimination, CH_4 production is a loss of energy for the animal (2 to 12% of GE intake, Johnson and Johnson,

1995). Therefore, until recently, research on enteric CH_4 production has focused on CH_4 emissions from an energetic efficiency standpoint (*e.g.,* Moe and Tyrrell 1979; Johnson and Johnson 1995). However, with rising interests in the effect of enteric CH_4 on global warming and its implication on climate change (O'Mara 2011), research interest has been directed towards quantification and reduction of enteric CH_4 production (*e.g.,* Boadi et al. 2004c; Ominski et al. 2006; Grainger et al. 2007; Waghorn and Hegarty 2011).

2.2.2.1. Modelling enteric methane production

Measuring enteric CH_4 emissions from animals requires complex and often expensive equipment. Mathematical models are, therefore, widely used because they provide quick estimate of CH_4 emissions with minimum cost. Inventory of CH_4 from rumen fermentation at a regional, national and/or global scale is conducted through the use of mathematical models (Bannink et al. 2011; Alemu et al. 2011; Sejian et al. 2011). Models used to estimate enteric CH_4 production can be categorized into two principal groups: statistical (empirical) and mechanistic models.

2.2.2.1.1. Statistical (empirical) models

Statistical models have long been used as a tool to describe the empirical relationship between the animal and enteric CH_4 production (*e.g.,* Kriss 1930; Bratzler and Forbes 1940). The models relate animal and/or dietary factors to CH_4 output directly. Some of the commonly used statistical models for prediction of enteric CH_4 emissions from beef and dairy cattle are summarized in Table 2.1.

Table 2.1. Empirical models used to predict enteric CH_4 emissions from beef and dairy animals.

Equations[z]	R^2	n	References
CH_4 (MJ d^{-1}) = −2.07 + 2.636 × DMI (kg d^{-1}) − 0.105 × DMI^2 (kg d^{-1})	...	...	Axelsson (1949)
CH_4 (MJ d^{-1}) = 5.447 + 0.469 × (energy digestibility at maintenance intake, % of GE) + multiple of maintenance × [9.930 − 0.21 × (energy digestibility at maintenance intake, % of GE)/100 × GEI, MJ d^{-1}]	0.71	615	Blaxter and Clapperton (1965)
CH_4 (MJ d^{-1}) = 0.341 + 0.511 × NSC (kg d^{-1}) + 1.74 × HC (kg d^{-1}) + 2.652 × CEL (kg d^{-1})	0.67	404	Moe and Tyrrell (1979)
CH_4 (L d^{-1}) = 47.8 × DMI _ 0.76 × DMI^2 _ 41 (kg d^{-1})	0.75	315	Yan et al. (2000)
Linear: CH_4 (MJ d^{-1}) = 5.93 + 0.92 × DMI (kg d^{-1})	0.60	159	Mills et al. (2003)
Nonlinear: CH_4 (MJ d^{-1}) = a – (a + b)$e^{-cMEI\ (MJ/d)}$	0.81	159	Mills et al. (2003)
CH_4 (MJ d^{-1}) = GEI (MJ d^{-1}) x Y_m (%GEI)	-	-	IPCC (2006)
Linear: CH4, MJ d^{-1} = 2.72 + 0.0937 × MEI (MJ d^{-1}) + 4.31 × CEL (kg d^{-1}) − 6.49 × HC (kg d^{-1}) − 7.44 × Fat (kg d^{-1}).	0.54	872	Ellis et al. (2009)
Nonlinear: CH4, MJ d^{-1} = 10.8 × [1 − $e^{\{-[-0.034 \times (NFC/NDF) + 0.228] \times DMI,\ kg/d\}}$].	0.42	872	Ellis et al. (2009)

[z]a = Theoretical maximum CH_4 output (kg d^{-1}), b = Minimum CH_4 output (kg d^{-1}), c = Shape parameter calculated as [0.0011 x starch (g kg^{-1} DM)/acid detergent fiber (ADF) (g k^{-1}g DM)] + 0.0045, CEL = Cellulose, DMI = Dry matter intake, GE = Gross energy, GEI = Gross energy intake, HC = Hemicellulose, MEI = Metabolizable energy intake, NDF = Diet neutral detergent fiber concentration, NSC = Diet non-fiber carbohydrate concentration [100 – (crude protein (%) – fat (%) - NDF (%), - ash (%))], Y_m = CH_4 conversion factor (6.5±1% for dairy cow and grazing beef cattle, 3±1% for feedlot cattle).

Statistical models are important for rapid diet evaluation or large-scale inventory purposes. However, various studies questioned the prediction accuracy of the models and their application for enteric CH_4 inventory when they are applied outside the production system from which they were developed (Mills 2008; Ellis et al. 2010). Furthermore, empirical models could imply cause and effect where none actually exist, especially when the aim is to develop the strongest possible correlation for a given set of data (Holter and Young 1992; Mills 2008). In an attempt to increase enteric CH_4 prediction accuracy of statistical models, Mills et al. (2003) and Ellis et al. (2009) developed non-linear models. Moreover, application of empirical models with complex rumen digestive processes such as estimation of emissions or the impact of mitigation strategies is challenging.

As part of empirical modelling approach, IPCC has recommended equations for calculating enteric CH_4 emissions (IPCC 2006). Depending on the quality of the established database, the IPCC operates at three levels (Tiers 1, 2, 3) to estimate GHG emissions. Tier 1 methodology is the simplest calculation that uses default EF (kg CH_4 $head^{-1}$ yr^{-1}) value to estimate enteric CH_4 production. The default EF value proposed for North American beef cattle is 53 kg CH_4 $head^{-1}$ yr^{-1} (IPCC 2006). Tier 2 methodology calculates CH_4 production based on GE intake (GEI) of the animal and default CH_4 conversion factor (Y_m, % GEI). The default Y_m values proposed by IPCC (2006) are 6.5±1 for beef and dairy cattle and 3±1 for feedlot cattle. Tier 3 methodology is a more complex approach where calculation of CH_4 production is based on sophisticated models that consider detailed dietary and animal information.

2.2.2.1.2. Mechanistic models

The mechanistic methanogenesis model mainly depends on H_2 balance in the rumen (Figure 2.2). The inputs to the H_2 pool (mol d^{-1}) include: (i) H_2 produced during substrate fermentation

(carbohydrate and protein) into Ac and Bu (lipogeneic VFA), and (ii) H_2 produced as microbial populations utilize amino acid for growth. Outputs from the pool were comprised of: (i) H_2 utilized for production of Pr and valerate (glucogenic VFA), (ii) H_2 utilized for microbial growth on non-protein N, and (iii) biohydrogenation of ingested unsaturated fatty acids. Methane production is estimated from the H_2 balance (model inputs minus outputs). The model considers H_2 as a zero pool which means all the produced H_2 that is not utilized in the above proceeses are available for CH_4 production through methanogenesis (Mills et al. 2001). Production of H_2 in the model assumes 2 mol of H_2 produced per mol of Ac and Bu and 0.58 mol of H_2 produced for microbial growth on amino acids. Methane production is estimated from H_2 balance in the rumen (H_Y) as CH_4 (MJ d^{-1}) = (H_Y/4) x 0.883 assuming 4 moles of H_2 required for the production of 1 mol of CH_4, and 0.883 is the heat combustion of CH_4 in MJ mol^{-1} (Mills et al. 2001).

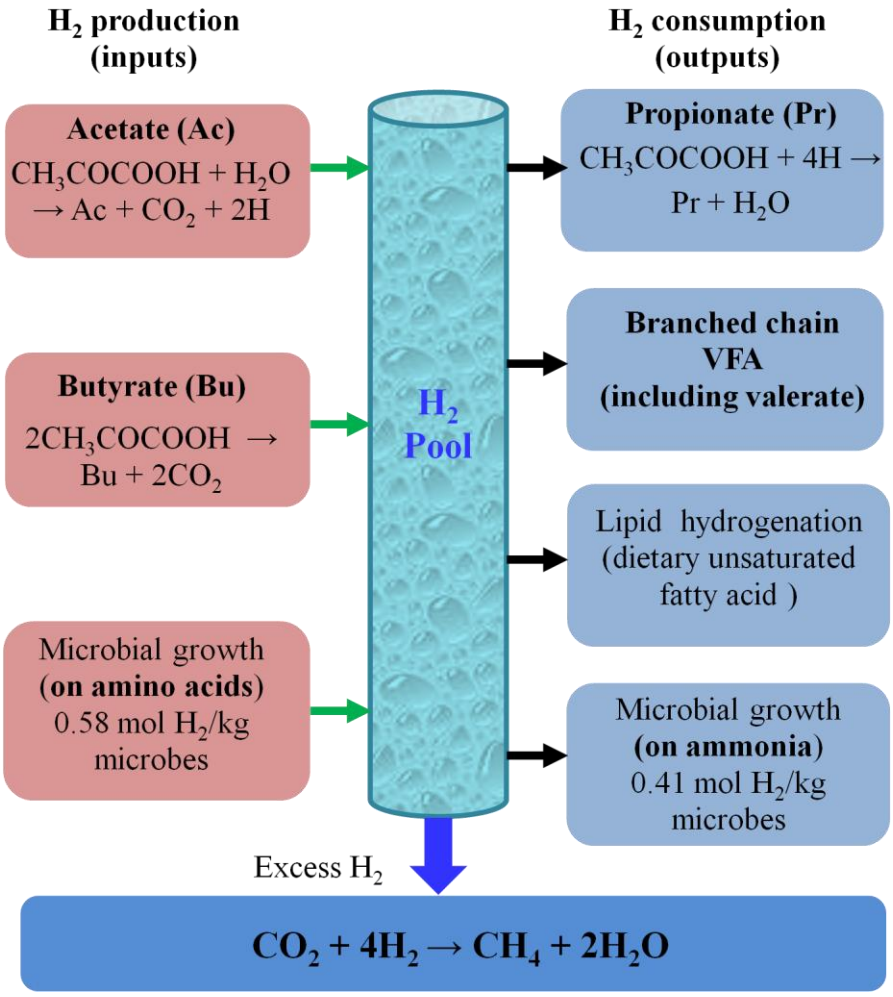

Figure 2.2. Rumen methanogenesis (H_2 gas balance) model.

2.3. Overview of Soil Emissions Models

Agricultural soil is a dynamic biological system that stores and releases GHGs. In most terrestrial ecosystems soil microorganisms play a key role in production and consumption of CO_2, CH_4 and N_2O (Li 2007). Soil microbes gain energy by breaking the carbon bond of dissolved organic compound which involves an electron transfer from the dissolved organic carbon to electron acceptors (O_2, NO_3^-, manganese (Mn^{4+}), iron (Fe^{3+}), sulfate or H_2). Under aerobic conditions, the microbes use O_2 as an electron acceptor and release CO_2 which in most cases, leads to decomposition of soil organic carbon (Li 2007). However, under anaerobic conditions where soil O_2 partial pressure is lower, the activity of decomposers is decreased and the activity of soil denitrifiers increases. Through the sequential reduction of NO_3^- to eventually N_2 by denitrifiers, some N_2O may escape before it has been further reduced. In addition to denitrification, emissions of N_2O could result from other processes such as nitrification (a process that produces NO_3^-) and chemodenitirification (Norton, 2008). If soil has been under anaerobic conditions for a long time (*e.g.*, several days), the activity of methanogens, which use H_2 as an electron acceptor, is activated and results in CH_4 production (Li 2007).

As GHG emissions from soil are the result of microbial processes, they are characterized by high spatial and temporal variability. As such, measurement of soil GHG emissions for inventory is impractical and many countries apply the IPCC default methodologies that implement fixed EFs to calculate GHG emissions from agricultural soils. However, the IPCC approach has been reported to have higher degree of uncertainty associated with EFs (*e.g.,* Smith et al. 2002). Therefore, various process-based mechanistic models have been developed to simulate fluxes of C and N in the atmosphere, vegetation and soil to estimate trace gas emissions on a daily basis. These types of models include DAYCENT (Del Grosso et al. 2002),

Denitrification–Decomposition or DNDC (Li et al. 1992a, b; Li 2007), ecosys (Grant and Pattey 2003) and Agricultural Production Systems sIMulator or APSIM (Thorburn et al. 2010). These models contain different key submodels to represent plant production and various biochemical processes in the soil including decomposition of litter and soil OM, mineralization of nutrients, soil water content and temperature by layer, N gas emissions from nitrification and denitrification and CH_4 oxidation in non-saturated soils. The DAYCENT and DNDC models have been utilized to estimate emission inventories from agricultural soils at local and global scales (Smith et al. 2008; Del Grosso et al. 2009; Abdalla et al. 2010). Model evaluation using soil N_2O emissions based on Canadian experimental data have indicated that the DNDC model was a better estimator than IPCC methodology (Smith et al. 2002) or the DAYCENT model (Smith et al. 2008).

2.4. Whole-farm GHG Analysis Using Systems Modelling Approach

The concept of whole-farm system modelling was developed as a consequence of the limitations of component-based research and sectoral approach in assessing farm GHG emissions which failed to consider the interrelationships among the farm systems (*e.g.*, animals, soil and crop, Crosson et al. 2011). Over the past few decades, a considerable amount of research has been conducted to quantify GHG emissions from various sources within the production system, such as enteric fermentation (Moss et al. 2000; Grainger et al. 2007), housing (Amon et al. 2001; Ellis et al. 2001), manure removal, storage and treatment system (Boadi et al. 2004b; Amon et al. 2006; Berg et al. 2006) and feed production (Liebig et al. 2010). Because most of these studies have investigated the individual components of the production chain, the interaction between C and N cycle in the system and the tradeoffs between emissions of the different GHGs are not considered. Therefore, various studies have proposed an integrated whole-farm approach of

GHG analysis (Janzen et al. 2006; Schils et al. 2007). In addition to quantifying the net farm GHG emissions, whole-farm modelling provides an opportunity to investigate the various GHG mitigation strategies from the entire farm. It is paramount to ensure that the application of management practices to one component of the farm will not increase emission from other farm components. In general, whole-farm modelling plays an important role in developing new mitigation technologies including BMPs available for farmers.

2.4.1. Steps in whole-farm modelling

Whole-farm GHG emissions models may be categorized as system analysis models or life cycle analysis (LCA) models (Crosson et al. 2011). The approaches used by both methodologies are similar although the LCA approach is more formalized (International Organization for Standardization (ISO) 2006a, b). According to ISO standards 14040 and 14044 (ISO 2006a, b) there are four main phases in LCA study: (i) definition of goal and scope, (ii) inventory analysis, (iii) impact assessment, and (iv) interpretation of results. According to Crosson et al. (2011), the corresponding phases for systems analysis models include conceptual framework definition, model development, model application and interpretation of results, respectively.

2.4.1.1. Definition of goal and scope/conceptual framework

The first step in whole-farm modelling is to define goal and scope of the farming system of interest (Figure 2.3). According to ISO (ISO 2006a) definition of this goal includes an explanation of the objective and aim of the study as well as its intended audience. Most whole-farm studies in the literature aim to quantify the environmental impact of the current production system, which is defined as attributional LCA, in contrast to a consequential LCA, where the objective is to quantify the environmental impact resulted from marginal change in the level of output from a product (ISO 2006b)

Defining the scope of the study includes describing the system under study and its function, the boundary of the system, functional unit and allocation procedure, data requirement and assumptions made (ISO 2006a). Defining the system boundary determines the parts of the system that are included and excluded in the assessment. Figure 2.3 illustrates the entire production process (cradle to retail) of a primary product (*e.g.*, meat, milk) and its associated GHG emissions. The cradle to retail system boundary is divided into two sub-systems: cradle to farm-gate and farm-gate to retail. Generally, whole-farm system studies in livestock production endeavor to assess the cradle to farm-gate process and its associated GHG emissions. As much as 70-90% of emissions in the total chain may occur before the product leaves the farm-gate (Gerber et al. 2010; Ledgard et al. 2010) and thus the farm gate is the most commonly used boundary for whole-farm systems studies. Greenhouse gas emissions from external farm inputs (*e.g.*, materials, energy and chemicals), called pre-chain emissions, have been included in most whole-farm studies (Johnson et al. 2003; Beauchemin et al. 2010; Foley et al. 2011).

In a whole-farm GHG analysis, the functional unit (quantified product or service of the system) is used as a reference unit to express its environmental impact using an allocation approach (ISO 2006a). Functional units used to allocate farm GHG emissions for beef production systems, include live and/or carcass weight (Casey and Holden 2006a, b; Ogino et al. 2007; Beauchhemin et al. 2010; Foley et al. 2011), weight gain (Phetteplace et al. 2001), protein produced (Stewart et al. 2009) and total farm area occupied (Flessa et al. 2002).

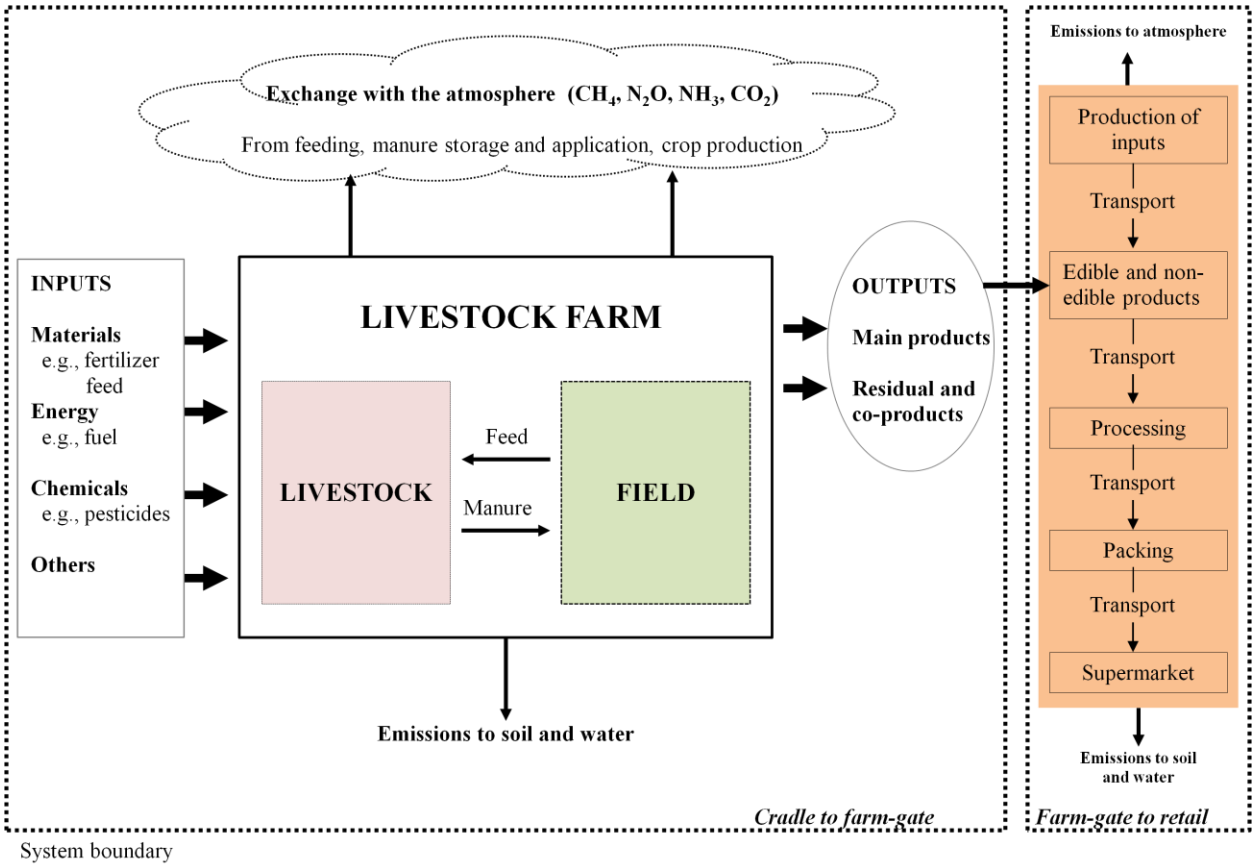

Figure 2.3. Illustration of the systems analysis/life cycle analysis of livestock products in a production system from source to market (modified after Gerber et al. 2010 and Hermansen and Kristensen 2011).

As agricultural systems produce more than one product and services (multi-functional output system, Figure 2.3) assigning functional units for these types of production systems is challenging. However, appropriate attribution of environmental impacts to each product from the system known as co-product allocation is necessary (Kristensen et al. 2011). The ISO 14044 standard (ISO 2006b) recommends that allocation by splitting a multi-functional process in such a way that it can be described as two separate processes each with a single output be avoided. Kristensen et al. (2011) reported that the allocation procedure used to divide farm GHG emissions between meat and milk has a significant impact on the estimated emissions from each product in Danish dairy production system. Cederberg and Stadig (2003) discussed four approaches for a co-product allocation: (i) no allocation in which all emissions are attributed to milk with no allocation for the sold animals (*e.g.*, Shills et al. 2005); (ii) economic allocation where farm emissions are allocated between the two products based on the annual income received from each (*e.g.*, Kristensen et al. 2011; McGeough et al. 2012); (iii) cause-effect biological allocation where emission allocation is based upon the energy required to produce or the energy available from each product (*e.g.*, McGeough et al. 2012); (iv) system expansion in which the system is expanded to include the alternative method of producing the product (*e.g.*, for dairy production systems, emissions related to meat produced from the systems are considered, Kristensen et al. 2011).

2.4.2. Current whole-farm models

In the past few years there has been an increase in the development of whole-farm models, in part, due to the increased interest in developing whole-farm GHG mitigation strategies. As a result, various whole-farm models have been developed including DairyNZ`s Whole Farm Model: WFM (Wastney et al. 2002), Farm ASSEssment Tool: FASSET (Berntsen et al. 2003),

Dairy Wise (De Haan et al. 2007), Sustainable and Integrated Management System for Dairy Production: $SIMS_{DAIRY}$ (Shills et al. 2007), Farm SIMulation: FarmSim (Shills et al. 2007), Holos (Little et al. 2008), Integrated Farm System Model: IFSM (Rotz et al. 2011a) and MELODIE (Chardon et al. 2012). These models are designed either for practical farm application as a decision support system for farm management (*e.g.*, IFSM, Holos, FASSET) or for research purposes (*e.g.*, MELODIE) to perform long term simulations.

The common features for some of the aforementioned whole-farm models are described in Table 2.2. A common feature of all the models is their ability to calculate CH_4 and N_2O emissions from the production system. The IFSM and Holos model are developed based on data collected from North American production systems.

Table 2.2. General characteristics of some current whole-farm models

Models	DairyWise	FarmGHG	$SIMS_{DAIRY}$	FarmSim	FASSET	MELODIE	Holos	IFSM
Model type	Empirical	Empirical	Semi-mechanistic	Semi-mechanistic	Semi-mechanistic	Mechanistic	Empirical	Semi Mechanistic
CH_4 emissions	X	X	X	X	X	X	X	X
N_2O emissions	X	X	X	X	X	X	X	X
NH_3 emissions	X	X	X	X	X	X		X
NO_3 leaching	X	X	X	X	X	X	X	X
CO_2 emissions	X	X		X	X	X	X	X
C sequestration				X	X	X		†
P cycling	X		X			X		X
Pre chain emissions	X	X		X			X	X
Economics	X		X		X	X		X
Time step	Monthly	Monthly	Monthly	Daily	Daily	Daily	Yearly	Daily
Farm components	Cattle, feed, manure, fields	Cattle, feed, manure, fields	Cattle, manure, fields	Cattle, manure, field (grassland)	Cattle, pig, feed, manure, fields	Cattle, pig, feed, manure, fields	Cattle, pig, sheep, poultry, manure, fields	Cattle, feed, manure, field
Reference	De Haan et al. (2007)	Olesen and Schelde (2008)	Del Prado et al. (2011)	Saletes et al. (2004)	Berntsen et al. (2003)	Chardon et al. (2012)	Little et al. (2008)	Rotz et al. (2011)

$SIMS_{DAIRY}$ =Sustainable and Integrated Management System for Dairy Production; FarmSim = Farm SIMulation; IFSM = Integrated Farm System Model; FASSET = Farm ASSEssment Tool; MELODIE = French acronym for 'object oriented model of animal farms to evaluate their environmental impacts.
†The IFSM calculates the C balance in the production system using a simple and robust approach considering the long term change in soil carbon is zero.

2.4.3. Assessment of whole-farm methodologies

Despite the importance of the whole-farm approach of assessing GHG emissions, there are also challenges in implementation, specifically obtaining input data and comparison of outputs obtained using whole-farm methodologies. Input data for whole-farm models could be obtained either from farm level measurement (Lovett et al. 2008; O'Brien et al. 2010) or a regional/national statistics (Phetteplace et al. 2001; Stewart et al., 2009; Beauchemin et al. 2010) or combination of the two (Basset-Mens et al. 2009). Even though the use of on-farm measured data represents the actual situations of the farm, the availability and quality of those data is questionable (Crosson et al. 2011). Data on manure handling, transportation and application rate, for example, are rarely measured in most beef production systems (Petersen et al. 2007). Conversely, the use of regional/national statistics data as representative farm data could also introduce major error into the whole-farm GHG analysis due to differences in management practice among the farms. Furthermore, the use of EF values from different sources and the use of simple empirical models in a whole-farm analysis could also be a source of error. Ellis et al. (2010) evaluated various empirical models that are used to estimate enteric CH_4 in existing whole-farm models and observed lower prediction accuracy for enteric CH_4 production which could be related to the aforementioned limitations of the models (section 2.2.2.1.1).

Another challenge associated with comparison of outputs is differences between models in the assumed system boundary (Figure 2.3) and functional unit. Most whole-farm analysis of beef production incorporate cradle to farm-gate (on-farm and pre-chain) emissions while other studies considered farm-gate to retail emissions (Gerber et al. 2010; Peters et al. 2010). Stewart et al. (2009) investigated total farm emissions from a beef production system by including pre-chain emission from manufacturing of machinery. Results were reported using protein as a functional

unit, whereas for a similar production system, Beauchemin et al. (2010) conducted GHG analysis excluding emissions from manufacturing of machinery and used weight (live and carcass) as a functional unit.

Further, in implementing a whole-farm approach, allocation of GHG emissions for co-products could be a challenge especially if the boundary is extended beyond the farm-gate (Peters et al. 2010).

2.5. Greenhouse Gas Sources and Emissions from Beef Production Systems

Beef cattle production is the major contributor of GHG emissions from the agricultural sector in Canada (Environment Canada 2011). In 2009, the livestock sector contributed 61% of the agricultural emissions and about 74% of the livestock sector`s contributions came from beef cattle production (Environment Canada 2011). The major GHGs produced from animal agriculture include CH_4, N_2O and CO_2.

2.5.1. Methane

Globally, CH_4 emitted by the world's farmed ruminant livestock accounts for about 37% of world`s anthropogenic CH_4 emissions (Martin et al. 2010). Methane emissions associated with ruminant production arise primarily from enteric fermentation and to a lesser extent, from manure storage. Agricultural soils serve as a sink as well as source for CH_4 in the production system (Ellert and Janzen 2008; Tenuta et al. 2010).

2.5.1.1. Methane produced from enteric fermentation

Enteric CH_4 from ruminant production is one of the major GHG that accounts for about 11-17% of CH_4 generated globally (Beauchemin et al. 2009a). Canadian beef production contributed about 80% of the total national enteric CH_4 emissions in 2009 (Environment Canada 2011). The amount of CH_4 produced depends largely on the amount of feed consumed (Reynolds et al.

2009) and chemical composition and characteristics of the diet (Benchaar et al. 2001). Beauchemin et al. (2009a) reported that daily emission of enteric CH_4 per head of beef cattle typically ranges from 0.05 to 0.5 kg. Similarly, the IPCC Tier 1 reported 0.15 kg as an average daily emission of enteric CH_4 for North American beef cattle.

Various mitigation strategies have been proposed to minimize enteric CH_4 production from ruminants. These strategies can be grouped into three major categories: (i) nutritional management, such as concentrate supplementation, diet modification and grazing management; (ii) chemical and biotechnological suppression of ruminal methanogenesis, such as defaunation, supplementation of ionophores, fat and organic acids supplementation, use of archaeal viruses and immunization; and (iii) genetic selection of animals for lower enteric CH_4 production. Mitigation options have been the subject of several published reviews (*e.g.*, McAllister et al. 1996; Boadi et al. 2004c; Kebreab et al. 2006b; Ominski and Wittenberg, 2006; Beauchemin et al. 2009a; Kobayashi 2010; Martin et al. 2010; Buddle et al. 2011; Clark et al. 2011; Sejian et al. 2011).

2.5.1.2. Methane produced from manure storage

Manure storage is the second most important source of CH_4 emissions from animal agriculture. Based on its solid content, manure managed in Canada may be categorized as solid (≥20% solid content), semi-solid (5-20% solid) and liquid manure (≤5% solid content), and manure produced from most beef production systems falls under the solid manure category (Beaulieu 2004).

Manure CH_4 is generated through reaction similar to that of enteric fermentation. Anaerobic environment, neutral pH, available electron acceptor, optimal temperature (30-40^{0}C), sufficient nutrients availability (N, P, K, S) and substrate with high OM content are the favorable conditions for methanogenesis to take place in stored manure (Conrad 1989; Kebreab et al.

2006b). Moreover, the amount of CH_4 produced from stored animal manure varies with type of animal, diet, manure management and climatic conditions (NRC 2003). Various studies reported the impact of diet formulation on the nutrient composition of stored manure (*e.g.*, Boadi et al. 2004b; Rejis 2007; Huang et al. 2010). The amount of partially degraded cell wall that escaped microbial digestion is lower for manure from animals fed concentrate diets relative to those fed high roughage diets (Boadi et al. 2004b). The subsequent release of C from the partially degraded cell wall affects the C content of stored manure and thus the associated CH_4 emissions. Relatively higher C:N ratio and low inorganic N:total N ratio has been reported for slurry from dairy cows fed diets with high fiber and lower protein content (Rejis 2007).

The proportional production of CH_4 from liquid storage systems (primarily anaerobic) is generally higher compared to solid systems (Amon et al. 2006). The amount of CH_4 reaching the environment from the liquid system depends on i) the rate of production ii) rate of diffusion and iii) the availability of oxygen through the diffusion pathway that affect the rate of CH_4 oxidation (Conrad 1989; Kebreab et al. 2006b). The amount of CH_4 reaching the atmosphere could be higher if the release of CH_4 is episodic which form bubbles on the surface. This is because the oxidation rate of CH_4 is reduced for episodic release relative to the release of CH_4 through diffusion (Whalen 2005). Surface crust formation may slow down the diffusion process and allow oxidation to occur (Petersen et al. 2005).

Due to the cost and difficulty of measuring the actual emissions from on-farm manure management facilities, reported CH_4 emissions from manure management are estimated using simple equations that consider animal type and number, manure temperature and feed characteristics (IPCC 2006; Wagner-Riddle et al. 2006; U.S. EPA 2012). However, as numerous factors influencing GHG emissions from manure storage, a need for dynamic mechanistic

models have been emphasized. Sommer et al. (2004) proposed algorithms that link C and N turnover to predict CH_4 and N_2O emissions which is currently used in the IFSM. More recently, Li et al. (2012) developed the manure-DNDC model by incorporating animal and manure management component to the existing DNDC model.

2.5.1.3. Methane production/consumption in soils

Methane production in the soil occurs following complete mineralization of organic compounds by methanogenic archaea under anaerobic conditions (Ball et al. 1997; Le Mer and Roger 2001). Methanogenic archaea are responsible for nearly all biogenically produced CH_4 from different habitats including rice paddies, landfills and wetlands. The lower concentration or lack of other electron acceptors, such as sulphate, NO_3^- or ferric ion in the soil is the major driver for soil CH_4 production (Le Mer and Roger, 2001). In the presence of O_2, methanotrophic archaea use CH_4 as a source of C and energy and produce CO_2 (Whitman et al. 2006).

Generally, well-aerated soil is considered as a sink for atmospheric CH_4 via methanotrophic oxidation (Le Mer and Roger 2001; Borken and Beese 2006). However, soil CH_4 consumption rate can be reduced by N fertilization (N fertilizer or manure applied as N source or deposited by animals). A field study by Tenuta et al. (2010) indicated an overall increase in CH_4 emissions from grassland soil following application of liquid hog manure. The authors also observed that soil CH_4 emissions and/or consumption are influenced by the level of water table during wet seasons with higher emissions occurring whengrasslands havehigh water tables. Manure is a source of readily available C which could increase the availability of soil C and enhance CH_4 emissions from a saturated soil environment. Furthermore, application of manure could reduce the redox potential of the soil and increase O_2 consumption by plant roots and heterotrophic organisms that may increase soil CH_4 emissions (Le Mer and Roger 2001).

Therefore, management systems that involve addition of N to the soil may have a significant effect on soil production/consumption of CH_4.

2.5.2. Nitrous oxide

Steinfeld et al. (2006) reported that livestock agriculture accounts for 65% of the global N_2O emissions. Production of N_2O is due to nitrification of ammonium (NH_4^+) to NO_3^-, or incomplete denitrification of NO_3^- to N_2 (Norton 2008).

Anthropogenic emissions of N_2O include direct emissions from manure management and crop and pasture land and/or indirect emissions from sources, such as N lost by volatilization from land-applied manure and/or N-based fertilizer and N loss via leaching and runoff from agricultural soils (IPCC 2006).

2.5.2.1. Nitrous oxide emission from manure management

Average estimates of N_2O emission from cattle and pig manure range from 0.1 and 0.9% of total N (Webb et al. 2012). The amount of N_2O emitted from manure storage depends on composition and characteristics of manure (*i.e.*, N and C content, pH, temperature, solid content) and the type of manure management system used. Composition is a function of the type and the associated digestibility of diet that the animals consumed (Mathot et al. 2012). Brown et al. (2000) explained that emission of N_2O tends to have direct positive relationship with manure solid content. In another study Wood et al. (2012) indicated that N_2O emissions from stored manure are influenced by manure total solid content. Furthermore, Harper et al. (2000) reported a correlation between manure N_2O emissions and the concentration of NH_4^+ and NO_3^- in manure.

Manure N_2O emissions vary according to season of manure storage. Mathot et al. (2012) reported a significant variation between manure stored in cold or warm season and recommended that manure storage in the warmer season should be avoided.

2.5.2.2. Nitrous oxide emissions from cropland and pastureland

The processes of nitrification and denitrification in cropland and pastureland are influenced by many factors including climate (*e.g.*, rainfall, temperature, freezing and thawing regimes), soil characteristics (*e.g.*, pH, OM, redox potential, moisture), management practices (tillage, legume cropping, crop residue management and N fertilization) and the interaction between them (Gregorich et al. 2005). Measured N_2O emission data collected from farming systems in various ecoregions across Canada (Helgason et al. 2005) indicated that N_2O emissions are correlated (r^2 = 0.27 to 0.47) to annual precipitation and soil and crop management practices (*i.e.*, inclusion of legume in the rotation, manure addition, N fertilizer addition). The authors calculated a simple emission coefficient for non-manured soils as 1.18% of N applied, which is higher than the IPCC default estimates of 1% of N (IPCC 2006). Rochette et al. (2008a) developed a Tier 2 methodology for the inventory of N_2O emissions from the agricultural soils in Canada by incorporating some of the influencing factors, such as climate, soil texture, tillage and topography. Rochette et al. (2008b) conducted a N_2O inventory from Canadian prairie for the year 1990 to 2005 using this methodology and concluded that 68% of the estimated mean direct N_2O emissions came from agricultural soils. The application of synthetic N fertilizer was the major contributor for the direct N_2O emissions (35%), followed by crop residue (24%), grazing animals (17%) and manure applied to soil (10%).

Wagner-Riddle and Thurtell (1998) found that N_2O emissions increased following alfalfa incorporation in the crop rotation. Similarly, Kagan (2000) reported almost a 10 fold increase in N_2O emissions during spring thaw from alfalfa site relative to native grass and wheat sites. Furthermore, relative to pasture or perennially cropped land, annually cropped land tends to release more N_2O (Gregorich et al. 2005). This could be due to: i) the difference in residue

(above-ground residue and roots) decomposition rate which is slower for perennial crops and ii) the difference in active growing period which is longer for perennials (Gregorich et al. 2005; Asgedom and Kebreab 2011; Lemke et al. 2011).

In general, application of N to the soil (N fertilizer, manure applied as N source or deposited by animals) increases the rate of N_2O emissions. Application of liquid manure results in a higher N_2O emission relative to solid manure or N fertilizer application (Gregorich et al. 2005). Gregorich et al. (2005) reviewed studies conducted in Eastern Canada (Quebec and Ontario) and reported a lower N_2O emissions from soil receiving solid manure (0.99 kg N_2O-N ha^{-1} yr^{-1}) compared to liquid manure (2.83 kg N_2O-N ha^{-1} yr^{-1}). Furthermore, several studies have reported higher N_2O emissions from soils that receive liquid manure relative to mineral N, however, emission may be influenced by several factors including soil texture and soil C content (Jarecki et al. 2008; Pelster et al. 2012).This could be due to the increased soil moisture, lower oxygen availability and increased quantity of labile C in the soil following liquid manure application which promotes denitrification process. In addition, solid manure has higher total N with slower mineralization and reduced availability in the short term for denitrification. Furthermore, N_2O emissions are also affected by the amount, time and method of manure application (Tenuta et al. 2010; VanderZaag et al. 2011). Tenuta et al. (2010) reported lower N_2O emissions (2.2 g N ha^{-1} d^{-1}) for grassland soil receiving the required liquid manure in a split application system where half of the manure was applied in spring and the rest in fall relative to application of all the liquid manure once during spring season (4.9 g N ha^{-1} d^{-1}).

2.5.2.3. Indirect nitrous oxide emissions

In addition to the direct sources, N_2O emission also contributed from indirect sources, such as emission arising from N leaching and runoff from agricultural soils and the volatilization and

subsequent deposition of NH_3. Indirect emissions occur through degassing of N_2O from aquifers and surface waters, steaming from N_2O dissolved in water draining through soils, or from denitrification of NO_3^- leached from soil (Well and Butterbach-Bahl 2010). The default values used by IPCC for fraction of N lost through volatilization are 0.1 kg NH_3-N (kg N applied)$^{-1}$ for synthetic fertilizer and 0.2 kg NH_3-N (kg N applied)$^{-1}$ for organic N as well as dung and urine N deposited by grazing animals (IPCC 2006). Furthermore, the default value used to estimate fraction of N lost through leaching/runoff is 0.3 kg N (kg N additions or deposition by grazing animals)$^{-1}$. To estimate indirect N_2O emissions from volatilized and re-deposited N as well as N lost through leaching/runoff, IPCC proposed default emission factors (EFs) of 0.01 kg N_2O-N (kg N volatilised)$^{-1}$ and 0.0075 kg N_2O-N (kg N lost through leaching/runoff)$^{-1}$ (IPCC 2006). Furthermore, studies indicate that 10 to 30% of the applied fertilizer or N excreted by animals may be volatilized as NH_3 (Bouwman et al. 2002). However, not all volatilized NH_3 is lost from the production system. Ammonia is rapidly absorbed by vegetation, soils and surface water and thus the large portion of volatilized NH_3 is re-deposit close to its sources. Asman (1998) estimated that within 2 km of the source, up to 60% of the volatilized NH_3 could be re-deposited.

However, due to the higher uncertainty associated with current estimates (IPCC, 2006), the magnitude of the emissions is still under debate (Weymann et al. 2008). Generally, estimating indirect agricultural N_2O is complicated because it is often difficult to differentiate between fluxes originating from agricultural and other N sources (Crosson et al. 2011).

2.5.3. Carbon dioxide

Multiple processes assimilate and emit CO_2 in livestock production systems. Carbon dioxide enters an agricultural production system from the atmosphere through photosynthetic fixation by plants during growth. Sources of CO_2 emissions include respiration of plants, animals and

microorganisms decomposing OM from manure and crop residue. On-farm energy use also serves as a source of CO_2 emissions from the production system. Chianese et al. (2009) indicated that typically over the course of one year, croplands are a net sink of CO_2, which means that plants assimilate more CO_2 in plant biomass than they emit. The authors calculated CO_2 flux as net ecosystem production (NEP) where NEP = photosynthesis - plant respiration – soil respiration for different croplands. They also indicated that balancing the CO_2 flux for a specific production system required consideration of all C inputs into the system (*e.g.*, manure C) and C outputs from the system (*e.g*, harvest). Various factors including climate and management practices (current and previous) may affect the flux of CO_2 from the production system.

The use of perennial crops (grass, grass-legume mix) or reduced tillage practices tend to increase carbon sequestration (the net removal of CO_2 from the atmosphere into long-lived pool of C (*i.e.*, terrestrial and geologic, Lal 2004) and reduce CO_2 emissions (West and Post 2002; Gregorich et al. 2005; Johnson et al. 2007). Using a global database of long term agricultural experiments, West and Post (2002) reported that change from conventional tillage to no-till could sequester from 0.43 to 0.71 Mg C ha^{-1} yr^{-1}. In addition to C sequestration, reduced tillage minimizes CO_2 emissions from the production system through reduced use of fuel for tillage, which contributes the greatest amount of CO_2 from farming activities (Johnson et al. 2007). Carbon also lost from the production system upon conversion of pastureland to cropland through the decomposition of vegetation and mineralization of soil C (*e.g.*, Lal 2004). Janzen et al. (1998) reported that upon conversion to cultivated cropland, ≥30% of soil C may be lost.

Other sources of CO_2 emissions in animal production system include the use of fossil fuel for the various operations in the farm (*e.g.*, harvesting feeds, feed delivery, manure removal and application), on-farm energy use and energy use during production of machinery and agricultural

inputs (*e.g.,* seed, herbicide and fertilizer). In a whole-farm GHG analysis CO_2 emissions related to production and transport of inputs into the system are considered as *pre-chain* emissions. However, in a sectoral approaches used by IPCC, emissions generated by farm activity through the use of farm inputs (*e.g.,* fertilizer, feed, pesticide) do not belong to agricultural sector but are covered by other sectors such as industry (*e.g.,* for the synthesis and packaging of inorganic N fertilizer and of pesticide) and transport (*e.g.,* transport of feed and fertilizer). Emissions from electricity and fuel use are covered in the buildings and transport sector, respectively (IPCC 2006).

SUMMARY

Rumen fermentation is a complex process involving microbial community and degradable substrates that result in production of VFA, CH_4, CO_2 and NH_3. Representation of this process using mathematical model is also complex. Various attempts have been made to represent the VFA dynamics and CH_4 production in the rumen using mathematical models which range from simple empirical models to complex mechanistic models. However, various studies observed weakness in the current models in representing rumen VFA dynamics and enteric CH_4 production.

Research and mathematical modelling in animal agriculture have been focused on identification and estimation of GHG emissions from individual components (*e.g.*, animal, soil, manure) in the farm without considering their interaction. However, assessment of total GHG emissions from animal production systems needs to incorporate all the sources and sinks of emissions that require consideration of the interactions among the farm components. This could only be achieved by implementing whole-farm models that integrate all the farm components. As such, in recent years, there has been an increasing interest in developing whole-farm models in animal agriculture.

3. RESEARCH HYPOTHESIS AND OBJECTIVES

HYPOTHESIS

Mathematical models have been implemented in animal agriculture at various levels, ranging from a micro-level of tissue and, organ to a macro-level of animal, farm and region, for different purposes including quantification of greenhouse gas (GHG) emissions. At the animal level, for example, empirical and mechanistic models have been used to simulate the digestion and metabolism processes including the complex rumen fermentation process that involve microbial activity and degradable dietary compounds. As such, the products of rumen fermentation process, such as volatile fatty acid (VFA) and methane (CH_4) have been quantified using different models. However, estimates by the models appeared to show considerable variation under different conditions. Therefore, it is anticipated that evaluating the currently available rumen VFA stoichiometric and CH_4 models using independent data will reveal the prediction accuracy and precision of the models. Furthermore, various models have also been developed for other components within animal production systems, such as agricultural soils and crop production to simulate the different biochemical process and estimate the associated GHG emissions. However, given the complexity of animal production systems, prior approaches of developing mathematical models for individual components in the system (*i.e.*, animal, soil, crop) separately does not consider the interaction among the components and with the surrounding environment. As such, it is anticipated that development of mechanistic, process-based models at farm level could provide opportunities to integrate the various components in the production systems and consider their interactions. Furthermore, the type of models used for estimating GHG emissions at the animal and/or farm level depends on the objective and the

available data. However, since the accuracy of estimates depend on the type of models used, caution needs to be taken when selecting models for GHG inventory particularly, if the estimates are used for management and/or policy-related decision making processes. Therefore, it is anticipated that estimating enteric CH_4 production using different models that are developed to estimate enteric CH_4 production will indicate the differences among the models and facilitate model selection processes for enteric CH_4 inventory.

OBJECTIVES

The overall objectives of the studies were: 1) to assess the accuracy of mathematical models that are used to estimate VFA and CH_4 production in the rumen and their impact on regional CH_4 inventory and 2) to develop and apply process-based, whole-farm model in order to analyze total farm GHG emissions.

The specific objectives include:

i) To evaluate the prediction accuracy of current VFA stoichiometric models using independent data sources.

ii) To demonstrate the challenges of representing rumen VFA and CH_4 production using mathematical models and highlight the existing research gaps.

iii) To estimate enteric CH_4 emissions from Manitoba beef cattle and assess emissions trends using empirical and dynamic mechanistic rumen models.

iv) To estimate total farm GHG emissions intensity from a cow-calf production system by integrating the existing process-based farm component models.

v) To compare total farm GHG emissions estimated using the integrated whole-farm model with estimates from other extant whole-farm models.

vi) To compare estimates of total farm GHG emissions and relative contribution of farm components associated with changes in management strategy *i.e.*, amount and time of liquid hog manure application.

4. MANUSCRIPT I

Rumen stoichiometric models and their contribution and challenges in predicting enteric methane production

Aklilu. W. Alemu[1], J. Dijkstra[2], A. Bannink[3], J. France[4], E. Kebreab[5]

[1]Department of Animal Science, University of Manitoba, Winnipeg, MB R3T 2N2, Manitoba, Canada

[2]Animal Nutrition Group, Wageningen University, Marijkeweg 40, 6709 PG Wageningen, the Netherlands

[3]Wageningen UR Livestock Research, PO Box 65, 8200 AB Lelystad, the Netherlands

[4]Centre for Nutrition Modelling, Department of Animal and Poultry Science, University of Guelph, Guelph, Ontario, N1G 2W1, Canada

[5]Department of Animal Science, University of California, Davis, CA, 95616, USA

Published in Animal Feed Science and Technology 166 – 167 (2011) 761 – 778.

4.1. ABSTRACT

Fermentation in the rumen is a complex process involving microbial activities and degradable dietary components. Therefore, representation of this process using mathematical models is also complex. Besides degradation of dietary components and microbial growth, fermentation stoichiometry needs to be known in order to evaluate specific dietary components for type of volatile fatty acid (VFA), H_2 and CH_4 produced during rumen fermentation. The objectives were to evaluate extant VFA stoichiometric models for their capacity to predict VFA molar proportion and CH_4 using independent data sources. Two data sets were organized from published literature. The first contained 141 treatments of rumen digestion studies with lactating dairy cows collected from 43 published experiments. The second data set contained 18 treatments from 8 studies. The experiments reported information on diet composition, true rumen substrate digestibility, molar proportion of VFA and enteric CH_4 production (the latter only for data set 2). Model comparison was based on mean square prediction error (MSPE), concordance correlation coefficient and regression analysis. In general, models showed different prediction performance with respect to the type of VFA in rumen fluid with root MSPE (RMSPE, % observed mean) values from 5.2 to 43.2. Among the 4 models evaluated, that of Murphy et al. (1982, MUR) had the highest RMSPE value for propionate (25.7%) with 19.6% MSPE being random error. The model of Bannink et al. (2006, BAN) had the lowest RMSPE (10.7%) for butyrate with 97.8% MSPE being random error. Similarly, the model of Nozière et al. (2010, NOZ) had the lowest RMSPE (5.2%) for acetate with 83.0% MSPE being random error. Variations among stoichiometric models in predicting VFA molar proportions affected estimated CH_4 production. Comparison of predicted *versus* measured CH_4 production showed that BAN had the lowest RMSPE (9.8%) with only 18.1% of MSPE error due to deviation of the regression slope from unity (ER). The RMSPE was

11.2 and 12.2% for NOZ and MUR, respectively, with ER being 44.3 and 21.4%, respectively. Prediction of CH_4 production using SVE had the highest RMSPE (16.7%) with 44.3% MSPE being ER. Results indicate that there were unexplained variations in model predicted VFA molar proportions *versus* observed values. The variation among stoichiometric models in predicting VFA production will have a major influence on the accuracy of estimated enteric CH_4 production. Currently, CH_4 inventory is usually based on IPCC Tier 2 approach, which compared to BAN, NOZ and MUR showed a higher prediction error in estimating CH_4 emissions. The IPCC Tier 2 approach had an RMSPE of 16.4% of observed mean with 56.9% of the error due to deviation of the regression slope from unity, indicating proportional bias due to inadequate representation of relationships in this approach. There may be a need for more mechanistic approaches that consider nutritional and microbial factors rather than empirical models that relate VFA molar proportions to nutritional factors. Based on our analysis, mechanistic models, particularly BAN, may be preferred for CH_4 inventory or mitigation purposes. Although current mechanistic models have a higher prediction accuracy and a demonstrably more adequate representation of relationships compared with the widely used IPCC Tier 2 approach, the prediction accuracy of current models requires further improvement and they still should be used with care for regulatory purposes either to create enteric CH_4 mitigation strategies or document impacts of mitigation strategies.

Key words: methane, rumen, stoichiometric model, volatile fatty acid.

4.2. INTRODUCTION

Fermentation in the rumen is a complex process involving microbial activities and degradable dietary components. End products of rumen fermentation include volatile fatty acids (VFA), H_2, CH_4, NH_3 and CO_2. Volatile fatty acids are important sources of metabolizable energy for ruminants (Bergman 1990), representing between 40 and 70% of digestible energy intake (France and Dijkstra 2005). The major VFA produced in the rumen include acetate (Ac), propionate (Pr) and butyrate (Bu), which generally account for more than 95% of the total VFA production (Bannink et al. 2006). Excess reducing power generated during conversion of hexose to Ac or Bu is utilized in part by Pr, but mainly by conversion to CH_4 (Moss et al. 2000). Therefore, proportions of Ac, Bu and Pr determine the amount of H_2 available in the rumen for utilization by CH_4 producing microbes (*i.e.*, methanogens). Thus, the accuracy of predictions of CH_4 production strongly depends on accuracy of quantifying the VFA produced in the rumen. Modelling approaches to this end were recently described by Ellis et al. (2008). Indeed, empirical models of enteric CH_4 formation that do not represent important aspects of diet composition have low prediction accuracy (Ellis et al. 2010).

Considerable efforts have been made to measure ruminal VFA production under various nutritional conditions, initially using a variety of non-tracer methods and, subsequently, tracer methods utilizing compartmental analysis to interpret isotope dilution data (France and Dijkstra, 2005). However, only a few studies have measured VFA production in dairy cows using isotope dilution (*e.g.,* Annison et al. 1974; Sutton et al. 2003) due to difficulties with the technique and cost. Therefore, VFA production is usually measured in animals by assuming that molar proportions of VFA in the rumen fluid represent the proportions of VFA produced. Using this assumption, VFA molar proportions in the rumen can be derived stoichiometrically (*e.g.*,

Murphy et al. 1982). Stoichiometric models estimate the proportion of individual VFA in rumen fluid by partitioning C between each VFA for each fermented substrate. Stoichiometric coefficients developed for various substrates fermented in the rumen have been used in several mechanistic whole-rumen models (*e.g.*, Dijkstra et al. 1992; Baldwin, 1995; Danfær et al. 2006) to estimate VFA proportions. However, it has been shown that the prediction of molar proportion of VFA for lactating dairy cows using those coefficients is inaccurate (Bannink et al. 1997a). In attempts to improve predictability of molar proportion of VFA by the models, Bannink et al. (2006) and Sveinbjörnsson et al. (2006) developed new sets of stoichiometric coefficients. Also, empirical equations to calculate VFA production in the rumen can be found in the literature (Lescoat and Sauvant, 1995; Nozière et al. 2010). Empirical approaches relate the molar proportion of a specific type of VFA in the total amount of VFA formed to general nutritional factors such as the dietary content of fiber, starch and protein, and/or the forage to concentrate ratio. Nozière et al. (2010) developed equations of VFA proportions in the rumen using a nonlinear regression approach.

The objectives were to evaluate the commonly used extant VFA stoichiometric models developed by Murphy et al. (1982, MUR), Bannink et al. (2006, BAN), Sveinbjörnsson et al. (2006, SVE), and Nozière et al. (2010, NOZ) for their ability to predict VFA and CH_4 using independent data sources.

4.3. MATERIALS AND METHODS

4.3.1. Model Descriptions

All models were developed by regression analysis of observed rumen fermentation data. MUR, BAN and SVE were based on the relationship between observed proportions of VFA in rumen fluid and the observed amount of substrate digested in the rumen with a mechanistic

representation of conversion of substrate into microbial mass and VFA. In contrast, NOZ related the proportion of VFA to measured digestible neutral detergent fiber (dNDF) and digestible organic matter (dOM), ruminal starch digestibility (RStD) and DM intake. Detailed descriptions of the individual models follow.

The VFA model of Murphy et al. (1982) was adapted from the stoichiometric model of Koong et al. (1975), which was developed using data generated mainly from beef cattle and sheep by dividing the data set into a forage based group and a concentrate group. 'Forage' refers to fresh grass or whole crops or stovers harvested from a field which are included in the diets of dairy cows and preserved as silage or hay, whereas 'concentrate' refers to everything else. A forage group include diets containing >500 g forage kg^{-1} dietary DM and a concentrate group include diets containing ≥500 g concentrates kg^{-1} dietary DM. Substrate compositions of the diet were divided into soluble carbohydrate (*i.e.*, sugars and soluble carbohydrate, pectin, organic acids), starch, cellulose, hemicellulose and protein. Each carbohydrate utilized in the rumen was considered to be partly fermented and partly incorporated into microbial biomass. A single model parameter was assigned to the portion of substrate incorporated into microbes from all substrate types. The model estimated stoichiometric coefficients describe the partitioning of fermented C between each VFA for the type of substrates fermented in the rumen. The coefficients were used in MOLLY, a mechanistic whole cow model (Baldwin et al. 1987) to estimate VFA molar proportions.

Bannink et al. (2006) gathered data from literature that conducted experiments on lactating dairy cattle and report the calculated forestomach true digestibility of dietary substrate and VFA molar proportions in order to develop a stoichiometric VFA model. Bannink et al. (2006) adopted a similar approach to that used by Murphy et al. (1982) with some modifications in the

model description. Diets were organized into forage based diets (*i.e.,* >500 g forage kg^{-1} dietary DM) and a concentrate group (*i.e.,* ≥500 g concentrates kg^{-1} dietary DM). Dry matter was divided into components such as starch, cellulose, hemicellulose, crude protein (CP) and fraction of DM not accounted for by component analysis (calculated as DM – ash – crude fat – starch – neutral detergent fiber (NDF) – CP). The dietary fraction not accounted for by component analysis was considered as rapidly fermentable or soluble (non-starch) carbohydrate. This calculation of soluble carbohydrate captures analytical error. The model assumed a fixed proportion of each substrate to be converted into microbial biomass. The model has been used in a mechanistic whole rumen model (Dijkstra et al. 1992), which was extended latter by Mills et al. (2001) to include calculation of CH_4 production.

Sveinbjörnsson et al. (2006) developed a VFA stoichiometric model for incorporation into the Nordic dairy cow model Karoline (Danfær et al. 2006), a model which is similar to BAN but also includes some dietary factors as additional explanatory variables. Coefficients for substrate fermentation in the rumen were developed based on particular data sets with diets from a Nordic database of dairy cow digestion studies. Input variables for the model are rumen degraded forage NDF (fNDF), concentrate NDF (cNDF), starch, CP, lactate and the remaining (or rest) fraction of organic matter (OM) (Re, calculated as DM – ash – starch – CP – fNDF – cNDF – lactate – VFA). Furthermore, concentrate ether extract (cEE) and feeding level (FL) were shown to have an effect on the variation in rumen VFA concentration patterns, and were incorporated as variables in the model to improve the explanation of observed data.

Recently, Nozière et al. (2010) proposed an empirical approach to estimate production of total VFA and the proportion of individual VFA in the rumen by a meta-analysis of literature

data. The ratio between measured dNDF and dOM explained change in VFA molar proportions in the rumen and a curvilinear response to change in the ratio of dNDF and dOM occurred. Also, RStD (g kg^{-1} starch intake) and DM intake (kg d^{-1} (100 kg body weight, BW)$^{-1}$) were included as covariates. The following equations were derived that can be used to estimate proportions of Ac, Pr and Bu (mol individual VFA (100 mol total VFA)$^{-1}$) as:

$$\text{Ac (mM mol}^{-1}) = 54.2 + 12.0 \log (100 \text{ dNDF/dOM}) - 0.052\text{RStD} - 1.99 \text{ DM intake} \quad [4.1]$$

$$\text{Pr (mM mol}^{-1}) = 19.7 - 6.63 \log (100 \text{ dNDF/dOM}) + 0.07 \text{ RStD} + 2.62 \text{ DM intake} \quad [4.2]$$

$$\text{Bu (mM mol}^{-1}) = 19.0 - 3.99 \log (100 \text{ dNDF/dOM}) - 0.026 \text{ RStD} \quad [4.3]$$

where: dNDF/dOM = digestible NDF/digestible OM; RStD = ruminal starch digestibility (g kg^{-1} starch intake), and DM intake= dry matter intake (kg d^{-1} (100 kg BW)$^{-1}$).

4.3.2. Data and Estimates

True rumen substrate digestibility values are required as an input for MUR, BAN and SVE to estimate production rate of individual VFA. However, studies that report true rumen digestibility, molar proportions of VFA and CH_4 production from a particular digestion trial are scarce. Therefore, two data sets were organized: the first (data set 1) was constructed using studies that mainly reported the calculated forestomach true digestibility of dietary substrate and proportion of VFA whereas the second (data set 2) was constructed using studies that reported forestomach or whole tract digestibility of dietary substrate and measured CH_4 production. Detailed descriptions of the two data sets are provided below.

4.3.3. Volatile fatty acids

The VFA prediction potential of the models was evaluated with data obtained from 43 published rumen digestion studies (141 treatments, data set 1) on dairy cows in several stages of lactation (114 ± 63 days in lactation), and BW (624 kg ± 47). Data were collected from Poore et al.

(1993), McAllan et al. (1994), Joy et al. (1997), Stensig and Robinson (1997), Yang et al. (1997; 2007), Zhu et al. (1997), Crocker et al. (1998), De Visser et al. (1998a, 1998b), Knowlton et al. (1998), Sutton et al. (1998; 2000), Abdalla et al. (1999), Beauchemin et al. (1999), Van Vuuren et al. (1999), Callison et al. (2001), Greenfield et al. (2001), Khorasani et al. (2001a, 2001b), Phipps et al. (2001), Ipharraguerre et al. (2002, 2005a, 2005b), Johnson et al. (2002a, 2002b, 2002c), Hristov and Ropp (2003), Oba and Allen (2003), Qiu et al. (2003), Ueda et al. (2003), Voelker and Allen (2003a, 2003b), Fernandez et al. (2004), Hristov et al. (2004), Taylor and Allen (2005a, 2005b, 2005c), Foley et al. (2006), Moorby et al. (2006), Martineau et al. (2007), Silveira et al. (2007), and Broderick et al. (2008). The data set included nutritional contents of diet, presence or absence of additives, description of the experimental animals (*i.e.*, BW, breed, stage of lactation), forestomach digestibility of dietary substrate, total VFA production and proportion of individual VFA (*i.e.*, Ac, Pr, Bu, other VFA). Treatments with supplements that have a potential effect on rumen fermentation were excluded from the dataset. Missing diet and ingredient composition values were taken from NRC (2001) or other digestion studies with similar diets. Average diet composition values of database 1 are in Table 4.1.

Input values for the stoichiometric models are in Table 4.2. For experiments that did not report forestomach digestibility of substrate, rumen digestibility was estimated according to Archimède et al. (1997), as well as using *in situ* and *in vitro* estimates of rumen digestibility coefficients and estimated rates of passage of dietary substrates (Robinson et al. 1987; Tothi et al. 2003; Huhtanen and Sveinbjörnsson 2006). The ability of this approach to create values that mimic rates of passage and digestion of dietary substrates from and in the rumen is not known.

Table 4.1. Mean, median, standard deviation (SD), maximum (Max) and minimum (Min) values of diet composition for data set 1[z] (*n* = 141).

	Mean	Median	SD	Max	Min
Dry matter intake (kg d^{-1}, *n* = 141)	20.8	20.9	3.1	26.8	11.3
Body weight (kg, *n* = 141)	624.2	630.0	47.1	757.0	504.0
Concentrate diets[y] (g kg^{-1} DM, *n* = 91)					
Soluble and rapidly fermentable carbohydrate	122.5	114.3	43.7	238.9	13.7
Starch (*n* = 71)	266.0	270.0	61.4	377.0	137.0
NDF (*n* = 91)	329.8	323.0	50.3	454.0	231.0
N (*n* = 91)	27.6	27.8	2.3	32.6	22.2
Cellulose (*n* = 48)	170.7	169.6	30.7	256.7	117.2
Hemicellulose (*n* = 80)	123.7	119.0	28.6	207.0	66.0
Forage (g kg^{-1} dietary DM, *n* = 91)	427.1	406.0	39.9	500.0	350.0
Forage diets[y] (g kg^{-1} DM, *n* = 50)					
Soluble and rapidly fermentable carbohydrate	145.9	152.8	49.7	289.2	27.3
Starch (*n* = 45)	218.3	236.4	80.2	336.0	4.0
NDF (*n* = 50)	361.4	365.5	72.7	544.8	249.0
N (*n* = 50)	26.9	26.9	3.9	38.7	19.2
Cellulose (*n* = 25)	183.1	175.5	38.1	262.8	104.4
Hemicellulose (*n* = 50)	145.3	150.0	41.5	224.8	61.1
Forage (g kg^{-1} dietary DM, *n* = 50)	621.9	600.0	81.2	800.0	510.0
Lactic acid (g kg^{-1} DM, *n* = 78)	23.5	23.2	18.0	93.6	0.0
cEE[x] (g kg^{-1} DM, *n* = 4)	19.0	12.1	14.1	58.6	0.02
fNDF[w] (g kg^{-1} DM, *n* = 50)	229.3	203.0	68.8	495.2	154.0
cNDF[v] (g kg^{-1} DM, *n* = 14)	111.6	110.0	54.2	279.9	7.2
Re[u] (g kg^{-1} DM)	135.9	134.5	49.3	265.6	13.7
Feeding level (kg DMI kg^{-1} BW)	3.3	3.4	0.5	4.3	1.8

[z]Data from: Poore et al. (1993); McAllan et al. (1994), Joy et al. (1997), Stensig andRobinson (1997), Zhu et al. (1997), Yang et al. (1997; 2007), Crocker et al. (1998), De Visser et al. (1998a, 1998b), Knowlton et al. (1998), Sutton et al. (1998; 2000), Abdalla et al. (1999), Beauchemin et al. (1999), Van Vuuren et al. (1999), Callison et al. (2001), Greenfield et al. (2001), Khorasani et al. (2001a, 2001b), Phipps et al. (2001), Ipharraguerre et al. (2002, 2005a, 2005b), Johnson et al. (2002a, 2002b, 2002c), Hristov and Ropp, (2003), Oba and Allen (2003), Qiu et al. (2003), Ueda et al. (2003), Voelker and Allen (2003a, 2003b), Fernandez et al. (2004), Hristov et al. (2004), Taylor and Allen (2005a, 2005b, 2005c), Foley et al. (2006), Moorby et al. (2006), Martineau et al. (2007), Silveira et al. (2007) and Broderick et al. (2008).

[y]Concentrate diets: diets with more than 500 g concentrates kg^{-1} dietary DM; soluble and rapidly fermentable carbohydrate estimated as: DM - ash - crude fat - starch - NDF - CP; NDF: neutral detergent fiber; forage diets: diets with more than 500 g forage kg^{-1} dietary DM.

[x]cEE: concentrate ether extract.

[w]fNDF: forage NDF.

[v]cNDF: concentrate NDF.

[u]Re: Rest fraction of OM (total DM – (ash + fNDF + cNDF + starch + CP + lactic acid + VFA)).

True digestion of OM in the rumen was calculated by subtracting duodenal feed OM flow (total OM flow − microbial OM flow) from OM intake. Microbial OM was calculated as microbial N/0.0996 (Clark et al. 1992). Reported apparent digestibility values for starch were corrected for microbial starch to calculate true rumen digestibility by subtracting duodenal feed starch flow from starch intake. Duodenal microbial starch flow was estimated to be 5 g microbial starch/65 g microbial protein (Nocek and Tamminga 1991). Missing values for microbial N and flow of N fractions to the duodenum were calculated according to Clark et al. (1992) and Archimède et al. (1997). The model of Sveinbjörnsson et al. (2006) required separate inputs of rumen digested fNDF and cNDF. Forage NDF concentration was reported separately in 50 experiments. For the rest of the experiments, fNDF was calculated from diet NDF content based on forage NDF concentration and proportion of forages in the diet. In order to estimate rumen digestibility of fNDF in the diet, the indigestible forage NDF fraction was calculated as 2.4 × lignin concentrations (Traxler et al. 1998; Van Soest et al. 2005) and a rumen NDF digestibility coefficient of 0.49 kg kg^{-1} NDF (Offner and Sauvant 2004) was applied for the potentially digested NDF fraction in the rumen. However, the relationship between lignin and the indigestible NDF fraction of forage varies widely among forages (Robinson et al. 1986; Huhtanen et al. 2006), and is well known to not be predictive in concentrate ingredients. For experimental diets containing silage as a forage source, lactic acid concentration was calculated based on the concentration of silage in the diet and its lactic acid concentration. Soluble and rapidly fermentable carbohydrate, lactate and Re were considered to have digestibility coefficients of 1 in the rumen. It was assumed that for every type of substrate truly digested in the rumen, partition between microbial growth and VFA production was identical.

In order to calculate their monomer equivalents (*i.e.*, mole of carbohydrate, protein or

lactate d^{-1}), molar weight of 162 g for carbohydrate, 110 g for CP and 90 g for lactate were assumed. According to the biochemical pathway of microbial fermentation (Baldwin 1995), it is assumed that 1.1, 2.0 and 1.0 mol of pyruvate was formed per mol of amino acid, hexose and lactate, respectively; and 1.0, 1.0, 0.5 and 0.5 mol of Ac, Pr, Bu and branched chain (Bc) VFA (including valerate) were produced per mol of pyruvate used when converted into each of these VFA, respectively. Production of Bc for SVE was estimated by assuming one-third of CP converted to Bc (Sveinbjörnsson et al. 2006) whereas, proportion of Bc for NOZ was estimated as the difference of the sum of the proportion of Ac, Pr and Bu from unity (1 – Ac – Pr – Bu).

4.3.4. Methane

The data set including measured CH_4 production was collected from the literature to evaluate CH_4 prediction potential of the VFA models (data set 2, Table 4.3). Treatments, 18, from 8 published studies on dry and lactating dairy cows, average BW = 609 kg ± 29.4, were collected from Wilkerson et al. (1997), Sutton et al. (1998, 2000), Abdalla et al. (1999), Phipps et al. (2000), Hindrichsen et al. (2006), Martin et al. (2008), and Beauchemin et al. (2009b) to form data set 2. The experiments reported type and composition of diets, rumen or whole tract digestibility of substrate, description of experimental animals (*i.e.*, breed, BW, stages of lactation), measurement of enteric CH_4 production and molar proportion of VFA in the rumen (5 studies). Only control treatments were used from experiments containing additives that might influence rumen fermentation and CH_4 production. For experiments that did not report forestomach digestibility, apparent forestomach digestibility of substrates was calculated using Archimède et al. (1997). A meta-analysis was conducted by Archimède et al. (1997) using 157 digestion studies that report forestomach and total tract dietary nutrient digestibility.

Table 4.2. Summary of rate of forestomach truly digested substrate for model inputs and observed molar VFA proportions in the rumen fluid for data set 1[z] (*n* = 141).

	Mean	Median	SD	Max	Min
Concentrate diets[y] (kg d^{-1}, *n*= 91)					
Soluble and rapidly fermentable carbohydrate	1.8	1.7	0.9	4.0	0.1
Starch (*n* = 69)	3.7	3.8	1.5	7.9	0.4
Crude protein (*n* = 89)	2.3	2.2	0.5	3.7	1.6
Cellulose (*n* = 18)	1.7	1.7	0.9	3.8	0.02
Hemicellulose (*n* = 10)	1.3	1.3	0.4	2.7	0.7
Forage diets[y] (kg d^{-1}, *n* = 50)					
Soluble and rapidly fermentable carbohydrate	2.2	2.3	1.1	4.8	0.04
Starch (*n*= 41)	3.1	3.0	1.5	6.4	0.1
Crude protein[x] (*n* = 33)	2.1	1.8	1.0	4.3	0.9
Cellulose (*n* = 10)	2.1	1.9	1.0	4.6	0.1
Hemicellulose (*n* = 6)	1.3	1.4	0.3	2.0	0.6
fNDF[w] (kg d^{-1})	1.4	1.4	0.5	3.1	0.4
cNDF[v] (kg d^{-1})	1.8	1.6	0.9	4.5	0.1
Observed molar VFA proportions in rumen fluid[u] (mol (100 mol total VFA)$^{-1}$)					
Ac (*n* = 141)	62.1	62.1	3.2	69.4	51.2
Pr (*n* = 141)	22.1	22.7	2.9	30.2	15.3
Bu (*n* = 141)	11.7	11.6	1.3	14.9	8.7
Bc (*n*= 88)	3.7	3.7	1.0	6.9	0.9
pH (*n* = 113)	6.1	6.1	0.2	6.5	5.7

[z]Data from: Poore et al. (1993); McAllan et al. (1994), Joy et al. (1997), Stensig and Robinson (1997), Zhu et al. (1997), Yang et al. (1997; 2007), Crocker et al. (1998), De Visser et al. (1998a, 1998b), Knowlton et al. (1998), Sutton et al. (1998; 2000), Abdalla et al. (1999), Beauchemin et al. (1999), Van Vuuren et al. (1999), Callison et al. (2001), Greenfield et al. (2001), Khorasani et al. (2001a, 2001b), Phipps et al. (2001), Ipharraguerre et al. (2002, 2005a, 2005b), Johnson et al. (2002a, 2002b, 2002c), Hristov and Ropp, (2003), Oba and Allen (2003), Qiu et al. (2003), Ueda et al. (2003), Voelker and Allen (2003a, 2003b), Fernandez et al. (2004), Hristov et al. (2004), Taylor and Allen (2005a, 2005b, 2005c), Foley et al. (2006), Moorby et al. (2006), Martineau et al. (2007), Silveira et al. (2007) and Broderick et al. (2008).

[y]Concentrate diets: diets with more than 500 g concentrates kg^{-1} dietary DM; soluble and rapidly fermentable carbohydrate estimated as: DM - ash - crude fat - starch - NDF - CP values calculated as N x 6.25; forage diets: diets with more than 500 g forage kg^{-1} dietary DM.

[x]Crude protein values calculated as N × 6.25.

[w]fNDF: forage NDF.

[v]cNDF: concentrate NDF.

[u]Ac: acetate, Pr: propionate, Bu: butyrate, Bc: branched chain VFA including valerate.

The equations (linear and non-linear) were used to estimate apparent forestomach digestibility of dietary substrates. Methane production was measured using the SF_6 tracer technique (1 treatment) and open circuit respiration chambers (17 treatments).

The theoretical fermentation balance equation of Demeyer (1991) was used to calculate CH_4 production (moles d^{-1}) from model predicted VFA production (moles d^{-1}). From the stoichiometry of the main anaerobic pathways, H_2 transfer reactions can be summarized as H_2 producing and H_2 using reactions as H_2 producing reactions:

$$C_6H_{12}O_6 \rightarrow 2CH_3COCOOH + 4H \quad [4.4]$$

$$CH_3COCOOH + H_2O \rightarrow Acetate + CO_2 + 2H \quad [4.5]$$

$$2CH_3COCOOH \rightarrow Butyrate + 2CO_2 \quad [4.6]$$

and H_2 using reactions,

$$CH_3COCOOH + 4H \rightarrow Propionate + H_2O \quad [4.7]$$

$$CO_2 + 8H \rightarrow CH_4 + 2H_2O \quad [4.8]$$

The theoretical fermentation balance equation of Demeyer (1991) assumes the amount of H_2 produced is equal to H_2 used in molar basis (*i.e.,* 100% H_2 recovery in VFA and CH_4). Methane production (moles d^{-1}) from predicted VFA in data set 2 was calculated as:

$$CH_4 \text{ (moles } d^{-1}) = (2Ac - Pr + 2Bu - Bc)/4 \quad [4.9]$$

$$CH_4 \text{ (MJ } d^{-1}) = CH_4 \text{ (moles } d^{-1}) \times 0.882, \quad [4.10]$$

where: 0.882 is heat combustion of CH_4 in MJ mol^{-1}.

4.3.5. Statistical Analysis

Observed molar proportions of individual VFA (data set 1) were used to evaluate model VFA predictions. Measured CH_4 values (data set 2) were used to evaluate estimated CH_4 production from model predicted VFA (equation 9). Assessment of error of prediction was conducted by

calculation of mean square prediction error (MSPE):

$$\text{MSPE} = \sum_{i=1}^{n} (O_i - P_i)^2 / n, \quad [4.11]$$

where: n is the number of observations and O_i and P_i are the observed and predicted individual VFA concentrations or CH_4 production, respectively. The square root of MSPE (RMSPE), expressed as a percentage of the observed mean, was used as a measure of accuracy of prediction. Mean square prediction error was decomposed into error due to overall bias of prediction, error due to deviation of the prediction line from unity, and error due to disturbance (random variation, Bibby and Toutenburg 1977).

Concordance correlation coefficient (CCC, Lin, 1989) was also calculated for evaluation of the precision and accuracy of the predicted values. Calculated CCC is the product of two components being: (i) the correlation coefficient estimate which is a measure of precision (r, deviation of observations from the best fit line) and (ii) a bias correlation factor (C_b) that measures how far the regression line deviates from the line of unity (accuracy).

$$\text{CCC} = \text{r} \times \text{C}_\text{b} \quad [4.12]$$

Another estimates (v) taht measures scale shift and (μ) that measures location shift relative to the product of two standard deviations is also calculated. A negative value for μ indicates model overestimation whereas a positive value indicates underestimation, of observed values.

$$C_b = \frac{2}{[v + 1/v + \mu^2]} \quad [4.13]$$

Table 4.3. Mean, median, standard deviation (SD), maximum (Max) and minimum (Min) values for dry matter intake, the rate of forestomach truly digested substrate and measured methane production (data set 2, *n* = 18)[z]

	Mean	Median	SD	Max	Min
Body weight (kg, *n* = 18)	608.6	604.0	29.4	672.0	572.0
Methane production (MJ d^{-1}, *n* = 18)	22.6	23.0	2.4	26.0	16.3
GE intake (MJ d^{-1}, *n* = 18)	375.1	369.5	70.7	490.8	250.0
Dry matter intake (kg d^{-1}, *n*= 18)	19.6	19.7	3.3	24.9	13.5
Forestomach true digestion rate (kg d^{-1})					
Soluble and rapidly fermentable carbohydrate[y]	2.3	2.3	0.6	3.1	1.1
Starch (*n*= 12)	3.4	3.5	1.7	5.7	0.2
Crude protein[x] (*n*= 12)	1.8	1.4	0.8	3.3	1.1
Cellulose (*n*= 4)	1.6	1.7	0.4	2.3	1.0
Hemicellulose (*n*= 4)	1.4	1.4	0.5	2.4	0.8
Lactic acid	0.6	0.7	0.2	1.0	0.2
fNDF[w]	2.0	1.9	0.7	3.7	1.1
cNDF[v]	1.1	1.1	0.6	2.1	0.0
Re[u]	1.7	1.7	0.5	2.7	0.7

[z]Data from: Wilkerson et al. (1997), Sutton et al. (1998, 2000), Abdalla et al. (1999), Phipps et al. (2000), Hindrichsen et al. (2006), Martin et al. (2008), and Beauchemin et al. (2009b).
[y]Soluble and rapidly fermentable carbohydrate estimated as: DM - ash - crude fat - starch - NDF - CP.
[x]Crude protein values calculated as N × 6.25.
[w]fNDF: forage NDF.
[v]cNDF: concentrate NDF.
[u]Re: Rest fraction of OM (total DM – (ash + fNDF + cNDF + starch + CP + lactic acid + VFA)).

Predicted values were regressed against observed values for assessment of model prediction bias (St-Pierre, 2003). The independent variable, predicted VFA or CH_4, was centred on its mean value to estimate the intercept at the mean value which measures the overall prediction bias (mean prediction bias). The slope of the regression is the estimate of the linear prediction bias. Mean centred bias and bias at the minimum and maximum values were calculated as described by St-Pierre (2003).

4.4. RESULTS

The combined data set has a wide range of DM intake (*i.e.,* 11.3 to 26.8 kg d^{-1}), diet composition, forage content of the diet and amount of calculated truly digested substrate in forestomach (Table 4.1 to 4.3) and was hence suitable for evaluation of the different models. Pearson's correlation test (Table 4.4) showed that Ac had a positive relationship with truly rumen digested cellulose and cNDF, but negative relationship with starch, soluble and rapidly fermentable carbohydrate, hemicellulose, CP, Re and feeding level. Butyrate was positively correlated with truly rumen digested NDF, soluble and rapidly fermentable carbohydrate, hemicellulose cellulose and Re. In contrast, Pr was negatively correlated with cellulose and NDF and positively with starch, soluble and rapidly fermentable carbohydrate, CP, Re and feeding level. Acetate had a negative correlation with Pr and Bu (Table 4.4).

4.4.1. Volatile Fatty Acids

Table 4.5 provides summary statistics of model performance in predicting VFA proportions (as mol individual VFA (100 mol total VFA)$^{-1}$). In general, prediction potential of the stoichiometric models varied based on the type of VFA with root MSPE (% of observed mean) values ranging from 5.2 to 13.1, 13.3 to 25.7 and CCC values ranging from 0.06 to 0.36, 0.20 to 0.36, respectively, for Ac and Pr.

Table 4.4. Pearson's correlation between measured VFA (mol individual VFA mol^{-1} total VFA)[z], forestomach truly digested dietary substrate (kg d^{-1})[y] and feeding level (FL, kg DM intake kg^{-1} live weight) (n = 141)

	Ac	Pr	Bu	Bc	Sc	St	He	Ce	CP	fNDF	cNDF	Re	FL
Ac	1.00												
Pr	-0.85*	1.00											
Bu	-0.35*	-0.08	1.00										
Bc	0.08	-0.32*	0.04	1.00									
Sc	-0.31*	0.18*	0.27*	-0.02	1.00								
St	-0.43*	0.44*	-0.12	0.16	0.04	1.00							
He	-0.19*	0.13	0.29*	-0.20*	0.12	-0.07	1.00						
Ce	0.37*	-0.45*	0.18*	0.08	-0.15	-0.29*	0.08	1.00					
CP	-0.22*	0.20*	-0.07	-0.06	0.29*	0.30*	0.001	0.14	1.00				
fNDF	0.10	-0.19*	0.19*	0.10	0.001	-0.04	0.45*	0.41*	-0.29*	1.00			
cNDF	0.26*	-0.32*	0.20*	-0.05	-0.21*	-0.33*	0.26*	0.89*	-0.04	0.08	1.00		
Re	-0.36*	0.22*	0.30*	-0.06	0.94*	0.04	-0.12	-0.17*	0.25*	-0.06	-0.20*	1.00	
FL	-0.34*	0.42*	-0.11	-0.16	0.23*	0.39*	-0.01	-0.30*	0.42*	-0.29*	-0.16	0.22*	1.00

[z]Ac: acetate, Pr: propionate, Bu: butyrate, Bc: branched chain.

[y]Sc: Soluble and rapidly fermentable carbohydrate, St: starch, He: hemicelluloses, Ce: cellulose, CP: crude protein, fNDF: forage NDF, cNDF: concentrate NDF, Re: total DM – (ash + fNDF + cNDF + starch + CP + lactic acid + VFA).

* Significant level of correlation, $P < 0.05$.

Prediction using BAN showed lower RMSPE for Pr (13.3%) and Bu (10.7%) with 80.7 and 97.8% of MSPE due to random error, respectively. Prediction of VFA using MUR had higher RMSPE for Pr (25.7%) and intermediate RMSPE for Ac and Bu whereas, prediction using SVE had higher RMSPE for Ac, Bu and Bc (13.1, 43.2 and 38.8%, respectively). However, when NOZ was used for prediction of VFA proportions, Ac and Bc had a lower RMSPE value (5.2 and 25.9%, respectively). Similarly, decomposition of CCC showed that the accuracy (C_b) and precision (r) of the models varied among VFA. Acetate was predicted more accurately (C_b = 0.94) and more precisely (r = 0.38) using NOZ compared to the other models, but Pr was predicted more accurately (C_b = 0.88) using BAN. Results of regressions between observed and predicted proportion of Ac and Pr are in Figures 4.1 and 4.2, respectively. Proportions of Ac were overestimated by SVE and underestimated by MUR (Figure 4.1). Propionate predictions by the models, except MUR, were distributed around the line of unity (Figure 4.2). When residuals were plotted against predicted Ac and Pr values (Figure 4.3A, 4.3B), there was mean and linear bias for all models. The magnitude of mean bias for BAN was less than 0.02 mol mol^{-1} at the minimum (*i.e.*, 0.17 mol mol^{-1}) and less than 0.02 mol mol^{-1} for the maximum (*i.e.*, 0.28 mol mol^{-1}) predicted Pr value.

Table 4.6 provides summary statistics for evaluation of MUR and BAN for the predominantly forage (*i.e.*, >500 g forage kg^{-1} dietary DM) and concentrate diets. Prediction using BAN had lower RMSPE for Ac, Bu and Bc on concentrate and forage diets as compared to MUR. In generally, the value of RMSPE ranged from 4.8 to 33.1% and 6.4 to 27.4% for BAN on predominantly concentrate and forage diets, respectively.

Table 4.5. Comparison of model performance in predicting VFA proportions (mol individual VFA (100 mol total VFA)$^{-1}$) using data from literature (n = 141)

	Models[z]															
	MUR				BAN				SVE				NOZ			
VFA[y]	Ac	Pr	Bu	Bc	Ac	Pr	Bu	Bc	Ac	Pr	Bu	Bc	Ac	Pr	Bu	Bc
Mean measured	62.1	22.1	11.7	3.7	62.1	22.1	11.7	3.7	62.1	22.1	11.7	3.7	62.1	22.1	11.7	3.7
Mean predicted	60.1	25.7	10.5	3.8	61.2	23.0	11.7	4.1	69.5	20.4	6.8	3.3	61.5	23.4	11.6	3.5
Mean squared prediction error (MSPE)[x]																
MSPE	40.5	32.1	4.4	2.3	11.5	8.6	1.6	1.3	65.8	10.8	25.4	2.0	10.4	9.1	2.9	0.9
RMSPE (mol (100 mol)$^{-1}$)	6.4	5.7	2.1	1.5	3.4	2.9	1.3	1.1	8.1	3.3	5.0	1.4	3.2	3.0	1.7	1.0
RMSPE (%)	10.2	25.7	18.0	41.7	5.5	13.3	10.7	31.1	13.1	14.8	43.2	38.8	5.2	13.6	14.7	25.9
ECT (%)	10.4	39.7	32.3	0.5	7.1	9.8	0.0	13.4	82.6	27.0	92.0	8.1	3.7	18.0	0.2	2.2
ER (%)	67.2	40.7	28.3	60.0	10.0	9.5	2.2	15.8	3.1	4.5	1.0	47.2	13.3	8.7	40.7	1.4
ED (%)	22.4	19.6	39.5	39.4	82.9	80.7	97.8	70.8	14.3	68.5	7.0	44.7	83.0	73.3	59.2	96.4
R^2	0.12	0.23	0.02	0.00	0.06	0.14	0.11	0.00	0.07	0.10	0.00	0.02	0.15	0.19	0.02	0.06
CCC[w]	0.24	0.30	-0.09	0.00	0.21	0.33	0.26	0.00	0.06	0.20	-0.00	-0.12	0.36	0.36	-0.14	0.15
r	0.32	0.48	-0.15	0.00	0.25	0.38	0.34	0.00	0.27	0.31	-0.00	-0.13	0.38	0.43	-0.15	0.24
C_b	0.75	0.62	0.61	0.97	0.82	0.88	0.78	0.67	0.21	0.66	0.05	0.90	0.94	0.85	0.92	0.63
μ	0.46	-0.95	1.08	-0.10	0.37	-0.39	0.00	-0.63	-2.73	0.80	5.92	0.45	0.23	-0.52	0.07	0.24

[z]MUR: Murphy et al. (1982) model, BAN: Bannink et al. (2006) model, SVE: Sveinbjörnsson et al. (2006) model, NOZ: Nozière et al. (2010) model.

[y]Ac: acetate, Pr: propionate, Bu: butyrate, Bc: branched chain VFA including valerate.(for SVE one-third of CP converted to Bc whereas for NOZ, Bc = 1 – Ac – Pr – Bu).

[x]RMSPE: root mean square prediction error (mol (100 mol)$^{-1}$) and % of measured mean value), ECT: error due to overall bias of prediction, ER: error due to deviation of the regression slope from unity, ED: error due to the disturbance or random variation.

[w]CCC: concordance correlation coefficient, r: correlation coefficient estimate, C_b: bias correlation factor, μ: location shift relative to the scale (squared difference of the means relative to the product of two standard deviations).

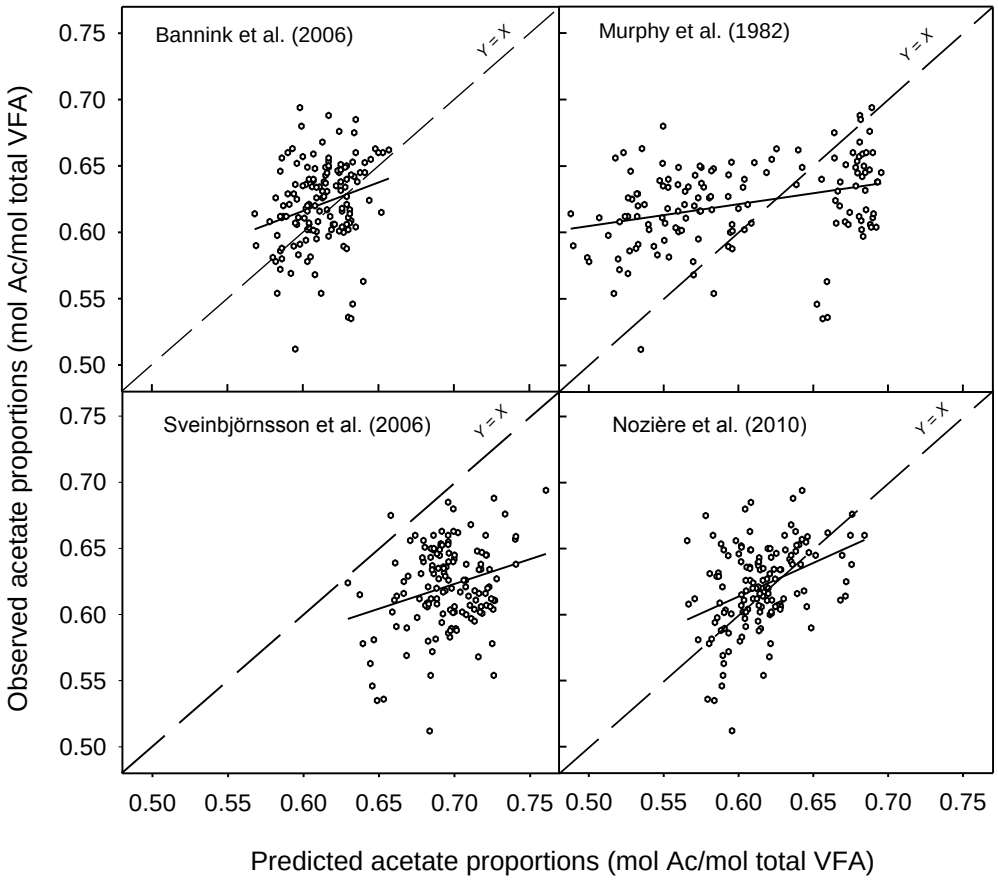

Figure 4.1. Observed *versus* predicted acetate molar proportions (mol Ac mol^{-1} total VFA, data set 1, n = 141) for the stoichiometric models of Bannink et al. (2006), Murphy et al. (1982), Sveinbjörnsson et al. (2006), and Nozière et al. (2010).

MUR had lower RMSPE for Pr (11.9%) on forage diets where 75.4% of MSPE were due to random error. Decomposition of CCC supported the above findings showing that Ac was predicted more accurately (C_b = 0.83) and precisely (r = 0.36) using BAN on predominantly concentrate diets. In contrast, Pr was predicted more precisely (r = 0.62) using MUR for predominantly forage diets. The bias correlation factor for the two diet categories varied between 0.18 and 0.95 with the lowest value for Bc from forage diet using MUR and the highest value for Pr from concentrate diet using BAN. For MUR, predicted VFA values were better correlated with observed values for, predominantly, the forage diet compared to concentrate diet category.

4.4.2. Methane

Summary of statistics for model evaluation using regression analysis of measured *versus* predicted CH_4 production estimated from model predicted VFA proportion values (data set 2) are in Table 4.7. The prediction performance of CH_4 production for all models evaluated was unsatisfactory with RMSPE values ranging from 9.8 to 16.7%. Root MSPE was lowest for BAN (9.8%), highest for SVE (16.7%) and intermediate for MUR (12.2%) and NOZ (11.2%). Similarly, CCC was highest for BAN (0.64). Decomposition of CCC analysis indicated that CH_4 was predicted more precisely (r = 0.71) using BAN and more accurately (C_b = 0.96) using NOZ. The plot of observed *versus* predicted CH_4 production (Figure 4.4A) showed that predictions by the models were distributed around the line of unity. Results of residuals plotted against predicted values (Figure 4.4B) indicated mean and linear bias for all models except NOZ, for which there was only a linear bias. The magnitude of linear bias for NOZ was less than 2.8 MJ d^{-1} at the minimum (*i.e.*, 17.7 MJ d^{-1}) and less than 2.7 MJ d^{-1} at the maximum (*i.e.*, 27.6 MJ d^{-1}) predicted CH_4 values.

4.5. DISCUSSION

4.5.1. Volatile Fatty Acids

Enteric CH_4 production would be better predicted from VFA production rates, than from their concentrations, in the rumen. However, there is a paucity of VFA production rate data in the literature. Most experiments measure VFA concentrations and mathematical models have been used to estimate the total amount of VFA produced and the relative proportions of Ac, Pr, Bu and other VFA. Therefore, evaluation of model predicted VFA was performed using reported VFA proportion values. The range of measured VFA proportions used in our study was similar to previously reported values (*e.g.*, Bannink et al. 2006).

Poor agreement between observed and predicted VFA proportions observed for all models (Table 4.5) are consistent with previous studies which reported that the proportions of VFA from rumen fermentation are poorly predicted using linear stoichiometric models (Neal et al. 1992; Pitt et al. 1996; Friggens et al. 1998; Bannink et al. 2000; Offner and Sauvant 2004). Less than 23% of the variation in individual VFA proportions was accounted for by the models.

However, correlation values for Ac and Pr for MUR and BAN were higher than values reported by Sveinbjörnsson et al. (2006) for those same models using diets from the Nordic database.

As part of evaluating mechanistic whole rumen models (Dijkstra et al. 1992; Baldwin 1995), unsatisfactory prediction performance of stoichiometric models has been reported. Bannink et al. (1997a, 1997b) suggested that poor prediction of VFA molar proportions in mechanistic whole rumen models is mainly due to inaccuracy of the stoichiometric models.

Table 4.6. Comparison of MUR[a] and BAN[z] for their prediction ability of VFA proportions (mol individual VFA (100 mol total VFA)$^{-1}$) on predominantly forage and concentrate diets

	Diets[y]							
	Concentrate (n = 91)				Forage (n = 50)			
VFA[x]	Ac	Pr	Bu	Bc	Ac	Pr	Bu	Bc
MUR								
Mean measured	61.6	22.8	11.5	3.6	63.1	20.8	12.0	3.7
Mean predicted	55.9	29.0	10.5	4.6	67.7	19.6	10.5	2.3
Mean squared prediction error (MSPE)[w]								
MSPE	45.1	46.4	3.9	1.8	32.2	6.1	5.4	3.2
RMSPE (mol (100 mol)$^{-1}$)	6.7	6.8	2.0	1.4	5.7	2.5	2.3	1.8
RMSPE (%)	10.9	29.8	17.1	37.3	9.0	11.9	19.3	48.4
ECT (%)	72.3	82.7	26.5	51.3	65.6	23.8	42.9	64.8
ER (%)	11.5	4.3	35.3	4.8	0.1	0.8	19.8	2.3
ED (%)	16.2	13.0	38.2	44.0	34.3	75.4	37.3	32.9
R^2	0.11	0.14	0.01	0.04	0.10	0.39	0.06	0.00
CCC[f]	0.12	0.09	-0.08	0.08	0.08	0.46	-0.10	0.00
r	0.34	0.38	-0.11	0.20	0.32	0.62	-0.26	0.02
C_b	0.36	0.25	0.70	0.43	0.26	0.73	0.40	0.18
μ	1.87	-2.45	0.90	-1.48	-2.15	0.60	1.55	2.65
BAN								
Mean measured	61.6	22.8	11.5	3.6	63.1	20.8	12.0	3.7
Mean predicted	60.6	23.4	11.7	4.3	62.4	22.2	11.7	3.7
Mean squared prediction error (MSPE)[w]								
MSPE	8.8	6.3	1.4	1.4	16.3	12.8	1.9	1.0
RMSPE (mol (100 mol)$^{-1}$)	3.0	2.5	1.2	1.2	4.0	3.6	1.4	1.0
RMSPE (%)	4.8	11.0	10.4	33.1	6.4	17.2	11.6	27.4
ECT (%)	11.7	6.4	2.2	29.0	2.9	16.2	5.4	0.0
ER (%)	7.0	10.5	4.7	13.1	24.5	27.0	0.1	0.1
ED (%)	81.3	83.2	93.1	58.0	72.6	56.8	94.6	99.9
R^2	0.13	0.25	0.11	0.00	0.04	0.03	0.15	0.03
CCC[v]	0.30	0.47	0.32	0.00	-0.12	-0.12	0.26	0.06
r	0.36	0.50	0.82	0.01	-0.19	-0.18	0.38	0.17
C_b	0.83	0.95	0.82	0.56	0.65	0.65	0.67	0.37
μ	0.44	-0.27	-0.20	-1.02	0.32	-0.75	0.35	0.00

[z]MUR: Murphy et al. (1982) model, BAN: Bannink et al. (2006) model.
[y]Concentrate diets: diets with more than 500 g concentrates kg^{-1} dietary DM, Forage diets: diets with more than 500 g forage kg^{-1} dietary DM.
[x]Ac: acetate, Pr: propionate, Bu: butyrate, Bc: branched chain volatile fatty acids including valerate.
[w]MSPE: mean square prediction error, RMSPE: root MSPE (mol (100 mol)$^{-1}$ and %), ECT: error due to overall bias of prediction, ER: error due to deviation of the regression slope from unity, ED: error due to the disturbance or random variation.
[v]CCC: concordance correlation coefficient, r: correlation coefficient estimate, C_b: bias correlation factor, μ: location shift relative to the scale (squared difference of the means relative to the product of two standard deviations).

Table 4.7. Comparison of stoichiometric VFA models in estimating methane production (n = 18)

	Models[z]			
	MUR	BAN	SVE	NOZ
Mean measured (MJ d^{-1})	22.6	22.6	22.6	22.6
Mean predicted (MJ d^{-1})	21.4	21.5	24.8	22.5
Mean squared prediction error (MSPE)[y]				
MSPE	7.6	4.9	14.3	6.4
RMSPE (MJ d^{-1})	2.8	2.2	3.8	2.5
RMSPE (%)	12.2	9.8	16.7	11.2
ECT (%)	20.0	26.5	33.7	0.3
ER (%)	21.4	18.1	41.2	44.3
ED (%)	58.6	55.4	25.1	55.4
R^2	0.19	0.51	0.35	0.36
CCC[x]	0.38	0.64	0.43	0.57
r	0.44	0.71	0.59	0.60
C_b	0.88	0.90	0.72	0.96
μ	0.53	0.46	-0.73	0.05

[z]MUR: Murphy et al. (1982) model, BAN: Bannink et al. (2006) model, SVE: Sveinbjörnsson et al. (2006) model, NOZ: Nozière et al. (2010) model.

[y]MSPE: mean square prediction error, RMSPE: root MSPE (mol (100 mol)$^{-1}$ and %), ECT: error due to overall bias of prediction, ER: error due to deviation of the regression slope from unity, ED: error due to the disturbance or random variation.

[x]CCC: concordance correlation coefficient, r: correlation coefficient estimate, C_b: bias correlation factor, μ: location shift relative to the scale (squared difference of the means relative to the product of two standard deviations).

Neal et al. (1992) evaluated the whole rumen model of Dijkstra et al. (1992) that employed MUR for its stoichiometric coefficients and found that VFA proportions were poorly predicted even though digestion of feed components was predicted satisfactorily. In that study, evaluation of the stoichiometric model was performed using measured VFA production rates by isotope dilution. Similarly, Offner and Sauvant (2004) reported unsatisfactory prediction of VFA concentrations while evaluating the whole rumen model of Baldwin et al. (1987), which uses MUR, and Lescoat and Sauvant (1995) which uses simple linear models to estimate VFA molar proportions.

Evaluating the stoichiometric coefficients of MUR and BAN in our study for predominantly forage and concentrate diets indicated that correlation between observed and predicted VFA proportions was higher for MUR on forage diets (R^2 = 0.00 to 0.39) and for BAN on concentrate diets (R^2 = 0.00 to 0.25; Table 4.6). Production of VFA and their relative concentration in rumen fluid appears to have a closer relationship when a forage diet was fed compared to a concentrate diet (Sutton 1980, 1985; Sharp et al. 1982). Two factors were suggested by Sharp et al. (1982) for the higher correlation between VFA production and ruminal concentration on provision of forage diets, namely slower synthesis rate of VFA and faster fractional rate of ruminal fluid dilution with forage than concentrate. These factors reduce differential fractional VFA absorption from the rumen.

Given the complexity of rumen fermentation and the low prediction ability of extant stoichiometric models, development of such models based on simple relationships between fermented substrate and VFA proportion has been questioned. Several attempts have been made to improve the models for better VFA representations, which range from incorporation of factors that affect rumen fermentation into the models (Argyle and Baldwin 1988; Pitt et al. 1996; Nagorcka et al. 2000; Bannink et al. 2008) to developing new stoichiometric coefficients

(Bannink et al. 2006; 2008; Sveinbjörnsson et al. 2006). A number of possible factors were suggested that could be incorporated into the extant stoichiometric models to improve their prediction ability, including rumen pH, variation in rumen microbes, interconversion between VFA, variation in absorption rate of individual VFA, fractional rumen fluid outflow rate and fluid volume (Bannink et al. 2006). By considering some of these factors as part of the model, improvements in VFA representation have been reported (*e.g.*, incorporation of rumen pH (Argyle and Baldwin 1988; Pitt et al. 1996; Bannink et al. 2008) and variation in rumen microbes (Nagorcka et al. 2000)). Incorporating factors that affect rumen fermentation in the regression models might allow more accurate VFA stoichiometry from rumen observations. However, the major challenge is that the complexity of the models also increases, probably requiring inputs not readily available from the majority of rumen digestion studies and inflating the number of model parameters that will need to be estimated. Even though they may not be directly applicable in current whole animal models, a few mechanistic approaches based on application of thermodynamic laws have also been attempted (*e.g.*, Kohn and Boston 2000).

Broader applicability of the stoichiometric coefficients, both for a range of conditions and types of animal, should be questioned during evaluation of different stoichiometric models (Friggens et al. 1998; Dijkstra et al. 2007), because the model coefficients were developed using different data sets. Coefficients of SVE, for example, were developed from a data set for Nordic countries characterized by high levels of grass silage and concentrates largely based on barley grain and rapeseed meal. In contrast, the coefficients of BAN were developed from an extensive data set based exclusively on true forestomach digestion from mainly lactating Holstein Friesian dairy cows.

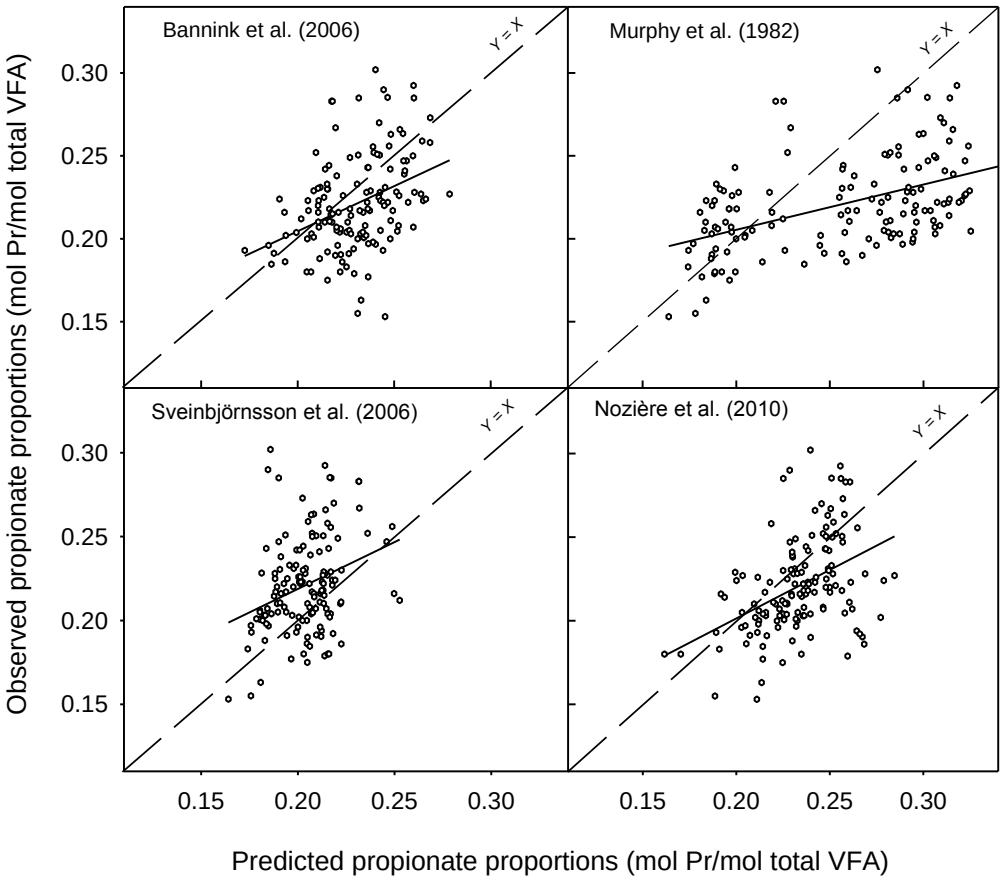

Figure 4.2. Observed *versus* predicted propionate molar proportions (mol Pr mol^{-1} total VFA, data set 1, n = 141) for the stoichiometric models of Bannink et al. (2006), Murphy et al. (1982), Sveinbjörnsson et al. (2006), and Nozière et al. (2010).

The model of Murphy et al. (1982) was developed from a data set mainly for beef cattle and sheep with hay as the forage source. Sveinbjörnsson et al. (2006) acknowledged that the coefficients of SVE may not necessarily have universal application to dairy cow diets. Application of the coefficients of MUR to silage based diets fed to sheep overpredict Ac and Bu and underpredict Pr (Friggens et al. 1998). Similarly, experiments that compared feeds have found substantial differences in proportions of VFA produced (Bosch et al. 1992; Martin et al. 1994). Thus stoichiometric coefficients probably need to be derived solely from data collected from the target ruminant.

The assumption of an identical fractional rate of absorption for all types of VFA may contribute to the lower prediction of the models. It has been reported that fractional rate of absorption varies with rumen pH (higher C chain VFA absorption increases at lower pH) and it is proportional to VFA concentration (Dijkstra et al. 1993; Lopez et al. 2003; Bannink et al. 2006). The *in vivo* VFA absorption study of Dijkstra et al. (1993), which used temporarily emptied and washed rumens, showed that at lower rumen pH, fractional absorption rate of Ac was lower than of Pr and Bu. For lower rumen pH, the assumption of equal fractional rate of absorption in extant stoichiometric models might overpredict the estimate of Ac production and underpredict those of Pr and Bu. The assumption is justified for high forage diets (*i.e.*, 850 to 900 g kg^{-1} forage on a DM basis), but is less likely to apply for diets, often with high concentrate levels, which result in a lower rumen pH (Sutton 1985; Dijkstra et al. 1993). The variation in the prediction performance of MUR and BAN on concentrate and forage diets (Table 4.6) in our study may be partly explained by the difference in fractional absorption rate among VFAs.

The importance of incorporating more than a single type of rumen microorganism relative to fermented substrate was emphasised by Nagorcka et al. (2000). Representing transformation of

substrates in extant stoichiometric models might indirectly represent the different effects of amylolytic and fibrolytic bacteria on substrate digestion. However, rumen protozoa, which use all types of substrate and produce relatively more butyrate than bacteria (Williams and Coleman 1997) are poorly represented. Protozoa, either directly or indirectly, influence types and numbers of bacteria, overall concentration and proportion of VFA, N recycling and rumen DM digestibility (Williams and Coleman 1997). Therefore, changes in protozoa populations in the rumen influence concentrations and proportions of VFA in rumen fluid. In their study, Nagorcka et al. (2000) set different stoichiometric coefficients of VFA yield for amylolytic bacteria, fibrolytic bacteria and protozoa using data from microbial incubation studies and reported an improvement in correlation between measured and model predicted VFA proportions. However, to represent the role of protozoa in rumen fermentation processes, the more detailed model of Dijkstra (1994) needs to be considered.

Incorporation of the additional parameters into the models could improve the accuracy of the models to properly represent the complex rumen fermentation process. Similarly, improving the quality of the data on which the models are developed and evaluated influence the estimation accuracy. For instance, estimation depends on accurate representation of dietary substrate degradation since the amount of degraded OM that is not incorporated into the microbial mass is used by the models to estimate the amount of VFA produced. Thus, instead of considering the data as a single data set, evaluation of data which include variation in dietary substrate digestibility might provide further improvement.

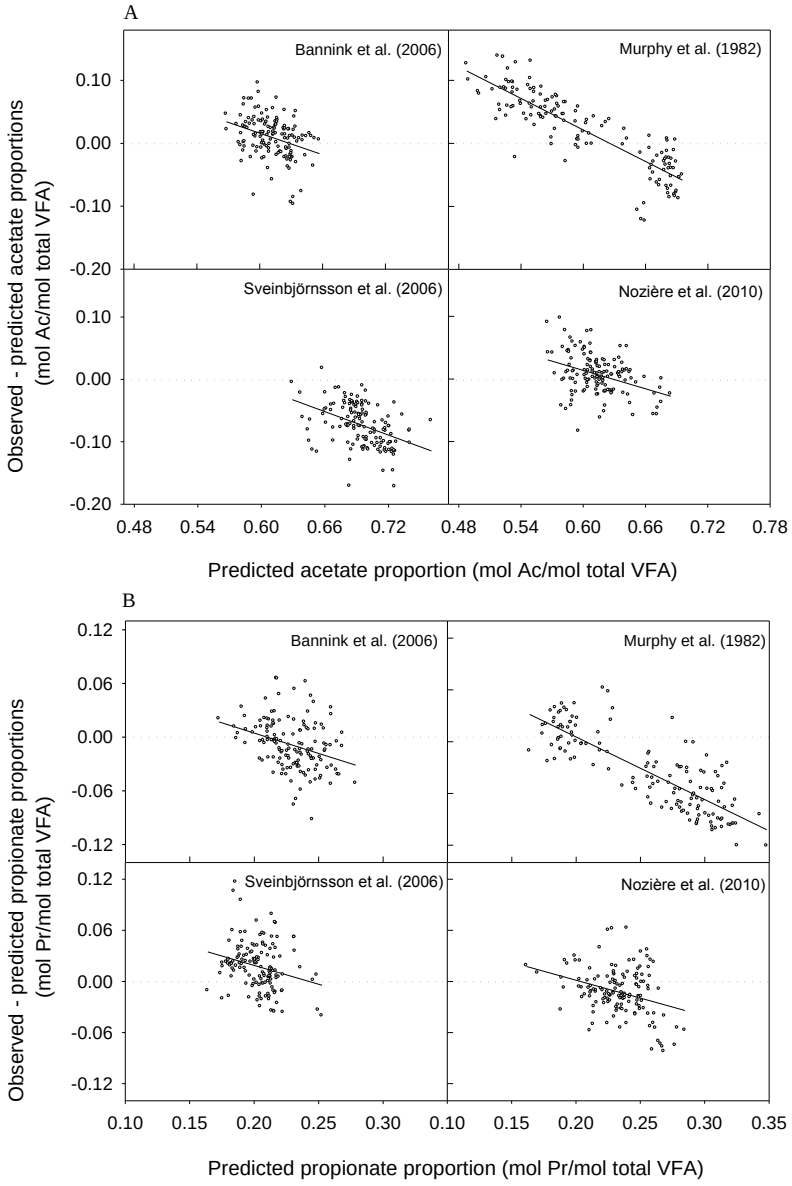

Figure 4.3. Residuals *versus* predicted acetate (A) or propionate (B) concentration (mol individual VFA mol^{-1} total VFA, data set 1, n = 141). The independent variables (predicted acetate and propionate proportions) were centered around the mean predicted value before the residuals were regressed on the predicted values. For acetate, the equations were: Bannink et al. (2006), $Y = 0.01(\pm0.003) - 0.57(\pm0.57)(X - 0.61)$, $R^2 = 0.11$, $P < 0.001$; Murphy et al. (1982), $Y = 0.02(\pm0.003) - 0.84(\pm0.04)(X - 0.60)$, $R^2 = 0.75$, $P < 0.001$; Sveinbjörnsson et al. (2006), $Y = -0.07(\pm0.003) - 0.63(\pm0.11)(X - 0.70)$, $R^2 = 0.18$, $P < 0.001$; Nozière et al. (2010), $Y = 0.01(\pm0.003) - 0.49(\pm0.10)(X - 0.62)$, $R^2 = 0.14$, $P < 0.001$. For propionate, the equations were: Bannink et al. (2006), $Y = -0.01\ (\pm0.002) - 0.46(\pm0.11)(X - 0.23)$, $R^2 = 0.11$, $P < 0.001$; Murphy et al. (1982), $Y = -0.04(\pm0.002) - 0.73(\pm0.04)(X - 0.26)$, $R^2 = 0.68$, $P < 0.001$; Sveinbjörnsson et al. (2006), $Y = 0.02(\pm0.002) - 0.44(\pm0.15)(X - 0.20)$, $R^2 = 0.06$, $P = 0.003$; Nozière et al. (2010), $Y = -0.01(\pm0.002) - 0.42(\pm0.10)(X - 0.23)$, $R^2 = 0.11$, $P < 0.001$.

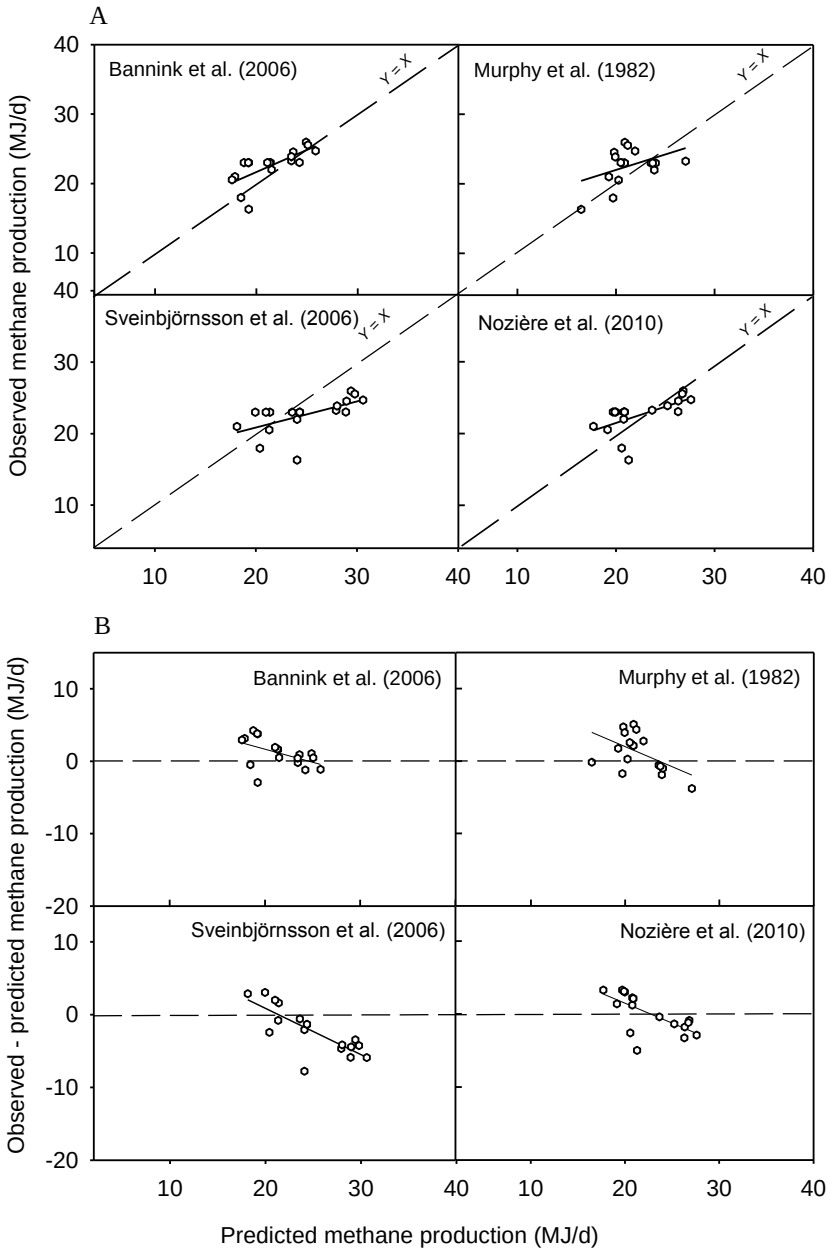

Figure 4.4. Observed (A) and residual methane production (B) *versus* predicted methane production estimated from VFA proportions calculated using stoichiometric models (MJ d^{-1}, data set 2, n = 18). Observed methane production values were the reported measured values. The independent variable (predicted methane production) was centered around the mean before the residuals were regressed on the predicted CH_4 production values. The equations were: Bannink et al. (2006), Y = 1.14(±0.41) – 0.36(±0.163)(X – 21.5), R^2 = 0.25, P = 0.04; Murphy et al. (1982), Y = 1.23(±0.53) – 0.56(±0.23)(X – 21.4), R^2 = 0.27, P = 0.03; Sveinbjörnsson et al. (2006), Y = –2.20(±0.47) – 0.64 (±0.12)(X – 24.8), R^2 = 0.62, P < 0.001; Nozière et al. (2010), Y = 0.14(±0.47) – 0.55(±0.15)(X- 22.5), R^2 = 0.44, P = 0.003.

4.5.2. Methane

Enteric CH_4 emission is the direct consequence of the net amount of H_2 produced through fermentation of feed in the rumen. Stoichiometrical relationships developed *in vitro* typically explain more than 95% of observed H_2 in VFA and CH_4 (Demeyer and Van Nevel 1975; Demeyer 1991). Accordingly, CH_4 production was calculated from VFA production using fermentation balance equations (Demeyer 1991) considering full recovery of H_2 in VFA and CH_4. However other pathways exist (*e.g.*, H_2 uptake in biohydrogenation of unsaturated fatty acids) and, although methanogenesis is usually a favoured route of disposal of H_2, competition for substrate between methanogens and other microorganisms occurs (Ellis et al. 2008).

Methane was predicted more precisely (r = 0.71) using BAN than the other models with lower RMSPE (9.8%, Table 4.7). SVE performed least well in predicting CH_4 production. The poor CH_4 prediction performance of the models was not surprising given the poor model prediction ability of VFA molar proportions (Table 4.5; Figure 4.1 to 4.3) and strong relationship between VFA and CH_4 production (Moss et al. 2000). The factors discussed earlier that affect rumen fermentation, but not incorporated into the stoichiometric VFA models (*e.g.*, rumen pH, variation in rumen microbes, inter-conversion between VFA), could also have a direct or indirect effect on enteric CH_4 production. For instance, error due to lack of incorporating variation in rumen microbes has been mentioned earlier, and the symbiotic relationship between protozoa and methanogens has been reported and these methanogens may be responsible for 25 to 37% of rumen CH_4 production (Finlay et al. 1994).

The rumen fermentation pathway depends on the level and type of substrate available for fermentation and its rate of depolymerization (France and Dijkstra, 2005). Thus, application of constant model estimated stoichiometric coefficients to partition the fermented C between each

VFA for all dietary composition or intake levels is not appropriate (Bannink et al. 2006), which may be the case in our evaluation given the wide ranges of DM intake and diet composition. With more substrate available for fermentation, either due to a higher rate of substrate fermentation or an increased level of feed intake, a shift in rumen fermentation pathway occurs from production of Ac to Pr, and to a small extent to Bu. Production of Ac and Bu results in production of excess reducing power whereas production of Pr is used as a sink for excess reducing power. Moss et al. (2000) reported that production of CH_4 has a negative correlation (R^2 = 0.77) with Pr production. Hence, poor prediction of CH_4 by the models could partly be due to omission of change in rumen fermentation rate. As such, besides type and amount of substrate, additional influencing factors, such as details on the composition of the rumen microbial population, physical aspects of rumen substrate degradation, outflow of substrate and microbial mass and conditions of the intra-ruminal environment, may need to be included in models to improve their prediction potential. Mills et al. (2001) evaluated CH_4 production of lactating dairy cattle using a modified version of the Dijkstra et al. (1992) model with VFA coefficients subsequently published by Bannink et al. (2006) against independent data. When using the MUR coefficients rather than the BAN coefficients, the prediction accuracy decreased (both mean bias and RMSPE increased).

Methane was also predicted using the IPCC Tier 2 approach, in which CH_4 energy output is 6.5% of GE intake (IPCC 2006), and is used widely for national inventory purposes under the Kyoto protocol. The IPCC Tier 2 approach overestimated CH_4 production (24.4 MJ d^{-1}), and RMSPE was higher (*i.e.*, 16.4% of observed mean) than that of BAN (9.8%), NOZ (11.2%) and MUR (12.2%). Moreover the error in the IPCC Tier 2 approach was dominated by the error due to deviation of the regression slope from unity (ER; 56.9%), whereas ER had a smaller

contribution especially in BAN (18.1%) and MUR (21.4%). This high ER contribution is a clear indication of proportional bias due to inadequate representation of relationships in the IPCC Tier 2 model. Ellis et al. (2010) evaluated the IPCC Tier 2 model against a larger set of independent data and concluded that the IPCC Tier 2 model does not have the capacity to fully describe changes in dietary composition and is limited in usefulness when estimating impacts of varying nutritional strategies on CH_4 emissions. Although the prediction accuracy of mechanistic models that we evaluated requires improvement, the accuracy of BAN in particular was better than that of the IPCC Tier 2 model. Moreover, these mechanistic models have the capacity or potential to describe several of the fermentative and digestive processes which are not included in simple empirical approaches, and to allow predictions of CH_4 emissions in response to dietary changes that are more credible than empirical approaches (*e.g.,* Benchaar et al. 1998; Mills 2001; Kebreab et al. 2006a).

4.6. CONCLUSIONS

Representation of ruminal fermentation using simple linear stoichiometric models to estimate the proportion of VFA showed poor correlation between measured and predicted VFA molar proportions (R^2<0.23), and there was variation among the stoichiometric models in predicting individual VFA. Failure to predict VFA proportions adequately influences accuracy of predicting enteric CH_4 production. Since the initial work of Murphy et al. (1982), various approaches have been undertaken to improve prediction of the VFA formed, with variable success. Of the major acetate, propionate and butyrate, the RMSPE using MUR varied between 10.2 and 25.7% of observed mean. In comparison to MUR, the SVE approach did not give a better prediction (RMSPE between 13.1 and 43.2%), but both BAN and NOZ substantially improve the prediction of type of VFA formed (RMSPE between 5.5 and 13.3, or between 5.2 and 14.7% of the

observed mean, respectively). Furthermore, stoichiometric models are typically developed based on a limited range of diets.

Developing models based on more mechanistic principles that incorporate both nutritional and microbial factors may improve the capacity of the models to predict both VFA and CH_4 more accurately. Of the mechanistic models, in particular BAN, predicted CH_4 production more accurately, and with far less error as proportional bias indicating inadequate representation of relationships, than the widely used IPCC Tier 2 approach and is to be preferred for inventory or mitigation purposes. Although the prediction accuracy of current VFA models require further improvement, the models show potential to predict effects of dietary interventions as a means to reduce CH_4 emissions of dairy cattle. Mechanistic models, in particular BAN, have a higher prediction accuracy and a demonstrably improved representation of relationships compared with the widely used IPCC Tier 2 approach, but at this time these models should still be used with care as quantitative predictors of enteric CH_4 production in ruminants for regulatory or assessment roles, either to create enteric CH_4 mitigation strategies or document impacts of mitigation strategies.

5. MANUSCRIPT II

Estimation of enteric methane emissions trends (1990 – 2008) from Manitoba beef cattle using empirical and mechanistic models

Aklilu W. Alemu[1*], K. H. Ominski[1] and E. Kebreab[2]

[1]*Department of Animal Science, University of Manitoba, Winnipeg MB, Canada; R3T 2N2,*

[2]*Department of Animal Science, University of California, Davis, CA, 95616, USA*

Published in Canadian Journal of Animal Science, 91 (2011):305-321.

5.1. ABSTRACT

The objective of this study was to estimate and assess trends in enteric methane (CH_4) emissions from the Manitoba beef cattle population from the base year of 1990 to 2008 using mathematical models. Two empirical (statistical) models: Intergovernmental Panel on Climate Change (IPCC) Tier-2 and a non-linear equation (Ellis), and two dynamic mechanistic models: MOLLY (Version 3) and COWPOLL were used. Beef cattle in Manitoba were categorized into 29 distinct subcategories based on management practice, physiological status, gender, age and production environment. Data on animal performance, feeding and management practices and feed composition were collected from the literature as well as provincial and national sources. Estimates of total enteric CH_4 production from the Manitoba beef cattle population varied between 0.9 and 2.4 Mt CO_2 eq from 1990 to 2008. Regardless of the type of models used, average CH_4 emissions for 2008 was estimated to be 45.2% higher than 1990 levels. More specifically, CH_4 emissions tended to increase between 1990 and 1996. Emissions were relatively stable between 1996 and 2002, increased between 2003 and 2005, but declined by 13.2% between 2005 and 2008, following the trend in the beef cattle population. Models varied in their estimates of CH_4 conversion rate (Y_m, % gross energy intake), emission factor (kg CH_4 $head^{-1}$ yr^{-1}) and CH_4 production. Total CH_4 production estimates ranged from 1.2 – 2.0 Mt CO_2 eq for IPCC Tier-2, 0.9 – 1.5 Mt CO_2 eq for Ellis, 1.3 – 2.1 Mt CO_2 eq for COWPOLL and 1.5 – 2.4 Mt CO_2 eq for MOLLY. The results indicated that enteric CH_4 estimates and emission trends in Manitoba were influenced by the type of model and beef cattle population. As such, it is necessary to use appropriate models for reliable estimates for enteric CH_4 inventory. A more robust approach may be to integrate different models by using mechanistic models to estimate regional Y_m values which are then used as input for IPCC Tier-2 model.

Key words: cattle (beef), model, enteric methane emissions, Manitoba

5.2. INTRODUCTION

Federal and provincial governments in Canada have committed to reduce greenhouse gas (GHG) emissions below the 1990 levels between 2008 and 2012 to meet the Kyoto target*. The province of Manitoba has also committed to reduce GHG emissions by about 4 Mt CO_2 eq during the same period (Government of Manitoba 2008). In 2008 the national GHG emissions were 32% higher than the 1990 Kyoto target of 557 Mt CO_2 eq (Environment Canada 2010). Manitoba contributed approximately 3.0% of the total national emissions in 2008. An examination of GHG emissions between 1990 and 2008 indicate that Manitoba's emissions increased by 18% during this time. The largest increase occurred in the agricultural sector (43.4%) followed by the road transportation sector (30.9%). Overall, in 2008 Manitoba's agricultural sector contributed 2.3 Mt CO_2 eq more than the 7.6 Mt CO_2 eq produced in 1990 (Environment Canada 2010).

Greenhouse gas emissions from the livestock sector are mainly from enteric fermentation and manure management. Methane (CH_4) is the major GHG produced from enteric fermentation during the normal digestive process of ruminants (Moss et al. 2000). Methane production arises principally from microbial fermentation of hydrolysed carbohydrates and is considered an energy loss for the host (Johnson and Johnson 1995). Enteric CH_4 production from ruminant livestock accounts for 17-37% of global anthropogenic CH_4 (Lassey 2008). In 2008, CH_4 emissions from enteric fermentation contributed an estimated 22.2 and 52.4% of the anthropogenic CH_4 in Canada and Manitoba, respectively. In Manitoba, a 56.0% increase in CH_4 emissions from enteric fermentation was reported between 1990 and 2008 (Environment Canada 2010).

*After this study was published, Canada announced its withdrawal from Kyoto protocol on climate change on Dec. 11, 2011 (http://www.cbc.ca/news/politics/story/2011/12/12/pol-kent-kyoto-pullout.html).

Various efforts have been made to identify animal husbandry practices that lower enteric CH_4 emission to reduce GHG emissions from agriculture (Boadi et al. 2004c; Beauchemin et al. 2008).

However, as a consequence of the expense associated with measuring emissions *in vivo*, mathematical models have been used extensively to calculate CH_4 emission inventories (Kebreab et al. 2009). These models can be classified into two main groups: empirical (statistical) models, which directly relate the nutrient intake of the animal to CH_4 production; and dynamic mechanistic models, which simulate CH_4 emissions based on a mathematical description of ruminal fermentation biochemistry (Kebreab et al. 2006b; 2009). Currently, national inventory of CH_4 emissions from enteric fermentation in most countries is calculated using the empirical Intergovernmental Panel on Climate Change (IPCC) model, which employs Tier-1, 2 or 3 methodologies in the derivation of estimates. The uncertainty of prediction using IPCC Tier-2 models is estimated to be ± 20% (IPCC 2006), and for Canada's national inventory, the agricultural sector has the highest uncertainty (37%) as compared to other sectors (Environment Canada 2010). In order to minimize the uncertainty associated with enteric CH_4 inventory using the IPCC Tier-2 model, countries such as the United States and the Netherlands have supported the use of IPCC Tier-3 by incorporating mechanistic models that consider more detailed diet information (United States Environmental Protection Agency (USEPA) 2010; Netherland Environmental Assessment Agency (NEAA) 2010). Several studies have been conducted to evaluate the prediction potential of dynamic mechanistic and empirical models for enteric CH_4 production from beef as well as dairy cattle using independent data sources (Benchaar et al. 1998; Kebreab et al. 2006b; 2008).

The objective of this study was to estimate and assess trends in enteric CH_4 production

from Manitoba beef cattle at distinct physiological stages of production over a 19 yr time frame (1990 - 2008) using empirical and dynamic mechanistic models.

5.3. MATERIALS AND METHODS

5.3.1. Model Description

Several models have been developed to predict enteric CH_4 production from beef cattle. In the current study, four models were selected based on their ease of application, widespread use to predict CH_4 emissions from enteric fermentation and their relevance to North American cattle production system. Two empirical (statistical) models [IPCC Tier-2 recommended by IPCC (IPCC 2006) and a non-linear equation developed by Ellis et al. (2009, Ellis)], and two mechanistic models [MOLLY (V3, Baldwin 1995) and COWPOLL (Dijkstra et al. 1992)] were used in this study.

IPCC Tier-2 (IPCC 2006)

There are three approaches that can be used to estimate enteric CH_4 emissions from livestock using IPCC methodology (IPCC 2006). The first and simplest methodology is the IPCC Tier-1, which uses a default emission factor (EF) to estimate enteric CH_4 emission. However, IPCC recommends either Tier-2 or Tier-3 methodology based on available country specific information on animal category, animal inventory and dietary information (IPCC 2006). The IPCC Tier-2 methodology, which is used in the current study, calculates CH_4 production from ruminants based on their gross energy intake (GEI, MJ d^{-1}) and the default CH_4 conversion rate (Y_m, % GEI). Methane conversion rate is the extent to which feed energy is converted to CH_4. The default Y_m values proposed by IPCC (2006) are 6.5 ± 1% for dairy cows and grazing beef cattle and 3 ± 1% for feedlot cattle. Tier-2 is used to quantify provincial and national enteric CH_4 production in Canada (Environment Canada 2010) and was examined in the current study. IPCC

Tier-3 methodology also requires country specific information, however, the level of information required is more detailed than Tier-2 approach. Tier-3 could employ the use of sophisticated models that consider detailed dietary information, seasonal variability in diet quality and possible mitigation strategies. Currently Tier-3 methodology is being used in the Netherlands and is under consideration in the U.S.

Ellis model

Several regression equations have been developed to estimate CH_4 emissions from ruminants (*e.g*. Blaxter and Clapperton 1965; Moe and Tyrrell 1979). Ellis et al. (2009) developed linear and non-linear equations and evaluated existing CH_4 prediction equations (Ellis et al. 2007) based on database containing nine published and three unpublished studies of enteric CH_4 emissions on Canadian beef cattle. The authors used individual animal data for their evaluation instead of treatment averages. In this study, all four equations recommended by Ellis et al. (2009) were considered in the preliminary analysis with three of the equations showing lower than expected values. Thus, the following non-linear equation was selected for this study:

$$CH_4 \text{ (MJ d}^{-1}) = 10.8 \text{ x } [1- e^{\{-[- 0.034 \text{ x (NFC/NDF)} + 0.228] \text{ x DMI, kg d-1}\}}], \quad [5.1]$$

where CH_4 = enteric methane production (MJ d^{-1}); NFC = non fiber carbohydrate (100 – (crude protein (CP)% + Fat% + NDF% + Ash%); NDF = neutral detergent fiber; DMI = dry matter intake (kg d^{-1}).

MOLLY Model

MOLLY is a dynamic and mechanistic model developed at the University of California, Davis based on rumen digestion and metabolism of dairy cow (Baldwin 1995). The model was constructed assuming continuous feeding, using Michaelis-Menten or mass action kinetics. The digestion element of the model is comprised of 15 state variables. Chemical composition of the

diet is presented as starch, cellulose, hemicellulose, lignin, soluble carbohydrate, acetate, propionate, butyrate, crude protein (soluble and insoluble), non-protein nitrogen, urea, ash (soluble and insoluble), lipid, organic acid, lactate, pectin and fat. After microbial attachment and substrate hydrolysis, the rumen model uses stoichiometric coefficients to convert starch, soluble carbohydrates and amino acids into volatile fatty acids (VFA). The VFA stoichiometry is based on the equation developed by Murphy et al. (1982), which relates the amount of VFA produced to the type of substrate fermented in the rumen. It was assumed that the H_2 produced in the rumen from fermentation of soluble carbohydrate and protein to VFA is used a) to support rumen microbial growth, b) for biohydrogenation of unsaturated fatty acids to saturated fatty acids, and c) for production of glucogenic VFA (*i.e.*, propionate and valerate). The remaining H_2 is used for reduction of CO_2 to CH_4 (Baldwin 1995).

COWPOLL Model

The model developed by Dijkstra et al. (1992) served as a basis for the development of the COWPOLL model. It is a dynamic and mechanistic model designed to simulate the digestion, absorption and outflow of nutrients in the rumen. The model contains 17 state variables representing N, carbohydrate (NDF, starch and sugar), lipid and VFA pools. Chemical composition of the diet is presented as starch (soluble and insoluble), NDF (degradable and undegradable), crude protein (soluble and undegradable), water soluble carbohydrate, ether extract, VFA (acetate, propionate, butyrate and valerate), ammonia, ethanol and lactate. Originally, the model developed by Dijkstra et al. (1992) did not predict rumen CH_4 production. Using the principles of Baldwin (1995), CH_4 production in the rumen and hindgut were added to the model by Mills et al. (2001). Kebreab et al. (2004) later integrated N processes and, as such, developed an extended model. As VFA molar proportions are important determinants of CH_4

formation, COWPOLL uses a VFA stoichiometry developed by Bannink et al. (2006) based on data collected from digestion trials with dairy cows. In addition to the stoichiometric differences described above, MOLLY and COWPOLL also differ in the number of microbial pools; MOLLY uses one microbial pool whereas COWPOLL uses three pools (amylolytic, cellulolytic and protozoa). As enteric CH_4 is produced in the rumen when excess H_2 is used by methanogens to reduce CO_2 to CH_4 (Moss et al. 2000), the two mechanistic models estimate CH_4 production in the rumen based on H_2 balance and sources (*i.e.*, acetate and butyrate formation) and ruminal H_2 sinks (*i.e,*. propionate formation, biohydrogenation; Mills et al. 2001).

Model simulation for COWPOLL was performed using acslX software (Version 2.4.2.1; The AEgis Technologies Group 2008). However, calculations for the two empirical models (IPCC Tier 2 and Ellis) were conducted using Microsoft Excel (Excel 2010).

5.3.2. Beef Cattle Population

Manitoba's beef cattle population data was obtained from CANSIM, Statistics Canada, Agricultural Division (Statistics Canada 2009), which publishes animal inventory data bi-annually in January and July. The inventory data was averaged to obtain the annual population of cattle (Table 5.1). Cross-border transport was not considered in the current study due to limitation of census data. According to the national census data, cattle were classified into the following categories: beef cows (female beef cattle that produces calves), replacement beef heifers (female beef cattle between the age of 12 months and first calving), bulls (male beef cattle above 12 months of age which are kept for reproduction purposes), steers (all castrated male beef cattle above the age of 12 months which are not kept for reproductive purposes), beef heifers (all female beef cattle which are not kept for reproductive purposes) and calves (dairy and beef cattle less than 12 months of age).

Table 5.1. Distribution of Manitoba's beef cattle population between the year 1990 and 2008 (`000 heads)[z]

Year	Beef cows	Replacement heifers	Heifers: Slaughter >1 yr	Steers, >1 yr	Breeding bulls, >1 yr[y]	Calves, <1 yr[x]	Total
1990	400.50	62.00	28.50	62.00	20.00	333.50	906.50
1991	409.00	71.50	34.50	59.00	21.00	368.00	963.00
1992	428.50	79.00	34.00	49.00	23.50	390.00	1004.00
1993	432.00	86.00	33.00	41.00	24.50	386.00	1002.50
1994	468.00	86.00	40.00	35.50	26.00	422.50	1078.00
1995	490.00	92.50	48.50	56.00	27.00	461.00	1175.00
1996	533.00	91.00	56.00	59.00	28.00	489.00	1256.00
1997	543.00	86.50	73.50	69.50	27.00	461.00	1260.50
1998	536.50	75.00	63.00	72.50	24.50	472.50	1244.00
1999	534.50	74.00	50.00	65.00	25.25	470.00	1218.75
2000	539.00	75.00	49.00	67.00	26.50	487.50	1244.00
2001	547.00	78.50	54.50	79.50	26.00	488.00	1273.50
2002	563.75	85.00	55.00	74.50	26.50	495.25	1300.00
2003	630.00	90.00	73.00	100.50	30.50	533.50	1457.50
2004	650.00	88.00	81.00	112.50	32.25	582.75	1546.50
2005	674.50	86.00	84.25	85.25	31.75	585.75	1547.50
2006	651.50	71.00	68.50	72.50	30.00	570.50	1464.00
2007	613.25	67.00	78.50	72.00	27.75	524.50	1383.00
2008	588.00	73.50	56.25	67.75	27.50	517.00	1330.00

[z]Population number is the average of July and January provincial inventory, Statistics Canada (2009).
[y]Include both beef and dairy bulls.
[x]Include both beef and dairy calves.

5.3.3. Dry Matter Intake and Diet Composition

Feed information was collected from previous surveys conducted in the province (Small and McCaughey 1999; Plaizier et al. 2004; Boadi et al. 2004a), which identified the major feeds used by Manitoba cattle producers. The most commonly used feeds in the province for beef and dairy cattle production include: hay, perennial silage, swath grazed crops (barley, oats), green feed (corn, oats, canola, barley), barley grain, barley silage, corn silage, pasture and straw. Hay is harvested from seeded forages (*i.e.*, grass and alfalfa-grass mix), and occasionally hay is used as a pasture supplementation for beef cows (Small and McCaughey 1999). A beef cattle management survey published by Small and McCaughey (1999) indicated that about 77% of the respondents use unimproved/native pasture. Subsequently, a Manitoba Agriculture Review (2002) supported this finding as 0.38 million ha were reported as improved pasture and 1.56 million ha as unimproved pasture. Feed quality information was collected from different sources including Manitoba Agriculture, Food and Rural Initiatives (MAFRI), Manitoba Forage Council (MFC), Central Testing Laboratory in Winnipeg, research reports as well as book values (National Research Council (NRC) 1982; 1996; 2001) and personal communication with provincial extension personnel. Pasture composition data was obtained from several pasture quality assessment studies conducted in the province. From 1996 to 1998, MAFRI conducted a provincial pasture quality assessment in the Eastern and Interlake regions of Manitoba (Manitoba Agriculture and Food 1999). Similarly, the Manitoba Benchmark Project (MFC 2006) was conducted by MFC from 2004 – 2008 to examine provincial pasture resources. The project collected pasture yield, quality and composition data from 72 sites located in four eco-regions of the province.

Both mechanistic models, MOLLY and COWPOLL, as well as Ellis require dry matter

intake (DMI) as an input in order to predict enteric CH_4 emissions. Due to lack of information on actual DMI for each category of the beef cattle population, DMI was estimated using Cowbytes©, beef cattle ration balancer (V.4.6.8, Alberta Agriculture, Food and Rural Development; Table 5.2). Feeds that closely resembled the province's feed composition were selected from the program and the default values were corrected for dry matter (DM), total digestible nutrient (TDN), CP, NDF and minerals based on Manitoba analyses. Representative beef cattle diets used under different management practices in the province were also identified for each beef cattle category (Table 5.2). For Ellis, diet gross energy values were calculated by considering a default feed energy density value of 18.45 MJ kg^{-1} of DM (IPCC 2006).

5.3.4. Management Practices and Beef Cattle Performance

Live Weight

Enteric CH_4 production is related to live weight of the animal (IPCC 2006). Carcass weight from meat production records (Agriculture and Agri-Food Canada (AAFC) 1990-2008) indicated that there was an increase in carcass weight between 1990 and 2008. The average live weight of beef cattle in the various categories for 2001 was reported by Boadi et al. (2004a). These live weights and their corresponding carcass weights were used to estimate live weights for all other years based on the following equation:

$$Lwt_{ij} = Lwt_{2001j} \text{ X } (Cwt_{ij}/Cwt_{2001j}), \quad [5.2]$$

where Lwt_{ij} is the live weight of j^{th} category of animal for the i^{th} census year, Lwt_{2001j} is live weight of the j^{th} category of animals for the census year 2001, Cwt_{ij} is a carcass weight for the j^{th} category of animals for ith census year, and Cwt_{2001j} is the carcass weight of the j^{th} animal category for the 2001 census year.

Table 5.2. Categories of Manitoba beef cattle population and average input values used to estimate enteric CH_4 emission from the different categories

No	Animal categories	Age (mo)	Management[z]	Time of year	Duration (mo)	Ave. weight (kg)	ADG[y]	DMI[x]	GEI[w]	DE[v]
1	Beef cows - pregnant[u]	Uknown[t]	Confined	Jan. - May/Nov. - Dec.	7.0	588	0.00	11.3	180.3	136.3
2	Beef cows - lactating[u]	Uknown[t]	Pasture	June - Oct.	5.0	588	0.11	12.2	229.8	170.2
3	Beef cows - pregnant[u]	Uknown[t]	Confined	Feb. - Mar.	2.0	588	0.00	11.3	180.3	135.5
4	Beef cows - lactating[u]	Uknown[t]	Pasture	Jan./Apr. - Dec.	10.0	588	0.11	11.9	224.9	151.0
5	Calves, birth to pasture	0 to 2.6	Confined	Mar. - May	2.6	82	1.00	0.0	0.0	0.0
6	Calves, pasture	2.6 to7.3	Pasture	June - Oct.	4.7	194	1.00	3.0	71.1	47.6
7	Calves, heifer replacement	7.3 to 12	Confined	Nov. - Dec./Jan. - Mar.	4.7	313	0.63	7.6	134.5	87.8
8	Calves, heifer backgrounded	7.3 to 12	Confined	Nov. - Dec./Jan. - Mar.	4.7	344	1.20	7.4	139.8	101.6
9	Calves, heifer feedlot	7.3 to 12	Feedlot	Nov. - Dec./Jan. - Mar.	4.7	356	1.30	8.3	174.6	134.9
10	Calves, steer backgrounded	7.3 to 12	Confined	Nov. - Dec./Jan. - Mar.	4.7	357	1.20	8.0	143.5	107.6
11	Calves, steer feedlot	7.3 to 12	Feedlot	Nov. - Dec./Jan. - Mar.	4.7	370	1.34	8.3	149.4	117.4
12	Heifers replacement	13 to 15	Confined	Mar. - May	3.0	359	0.63	9.1	157.1	99.9
13	Heifers replacement	16 to 20	Pasture	June - Oct.	5.0	430	0.63	10.2	180.5	130.3
14	Heifers replacement	21 to 24	Confined	Nov. - Dec./Jan. - Feb.	4.0	510	0.63	12.1	210.8	133.3
15	Finisher, heifers	13 to 15	Feedlot	Mar. - May	3.0	439	1.26	10.3	185.0	142.3
16	Finisher heifers-backgrounded[s]	13 to 17	Feedlot	Mar. - July	5.0	472	1.26	10.3	187.8	145.1
17	Backgrounded heifers	13 to 15	Confined	Mar. - May	3.0	434	1.20	10.0	179.3	131.2
18	Backgrounded heifers	16 to 18	Pasture	June - Aug.	3.0	524	1.20	12.9	242.9	160.1
19	Finisher, heifers	19	Feedlot	Sep.	1.0	555	1.26	11.5	213.5	163.4
20	Finisher, steers	13 to 15	Feedlot	Mar. - May	3.0	458	1.26	10.3	182.0	143.7
21	Finisher steers-backgrounded[s]	13 to 17	Feedlot	Mar.- July	5.0	491	1.26	10.3	184.6	146.6
22	Backrounded, steers	13 to 15	Confined	Mar. - May	3.0	455	1.20	10.0	177.3	132.2
23	Backgrounded, steers	16 to 18	Pasture	June - Aug.	3.0	546	1.20	13.2	243.0	164.6
24	Finisher, steers	19	Feedlot	Sep.	1.0	576	1.26	11.5	209.9	164.8
25	Breeding bulls, mature	Uknown[t]	Confined	Jan. - May/Dec.	6.0	937	0.00	15.5	276.2	185.5
26	Breeding bulls, mature	Uknown[t]	Pasture	June - Nov.	6.0	937	0.00	18.3	307.6	232.6
27	Breeding bulls, young[r]	13 to 16	Confined	Feb. - May	4.0	579	1.28	11.9	222.5	151.7
28	Breeding bulls, young[r]	17 to 22	Pasture	June - Nov.	6.0	758	1.28	13.3	259.9	178.3
29	Breeding bulls, young[r]	23 to 24	Confined	Dec./Jan.	2.0	902	1.28	15.8	298.2	202.2

[z]Confined: winter housing, which is drylot with barn (shade) (Small and McCaughey, 1999).
[y]ADG = Average daily gain (kg d^{-1}).
[x]DMI = Dry matter intake (kg d^{-1}), estimated using Cowbytes© beef cattle ration balancer (V. 4.6.8, Alberta Agriculture, Food and Rural Development) based on the following assumptions: for beef cows and herd bulls: British/Continental excluding Simmentals breed, no ionophore, body condition score = 3, expected birth weight = 44 kg, lactation number = 4, peak milk yield = 9.6 kg, wind speed = 5 km hr^{-1}, hair depth = 0.51 cm, dry and clean hair condition, average hide thickness, mud in lot = < 10 cm, no heat stress; for cows and calves on pasture: no ionophore, no melengesterol acetate (MGA), good quality pasture (750 to 1150 kg DM ha^{-1}), level terrain, average stocking rate = 2 ha cow-calf $pair^{-1}$ or 0.41 ha $feeders^{-1}$. For feeders and replacements: British/Continental excluding Simmentals breed, growth implant, ionophores, MGA for stocker heifers, average slaughter weight = 671 kg, slight marbling (Canadian AA marbling) and fed to 26.8% body fat. Current temperature (CT) = 10°C and previous month temperature (PT) = -15°C for Jan. – May, CT = 5°C and PT = 15°C for June – Oct., CT = -13°C and PT = -4°C for Nov. - Dec., CT = -12°C and PT = -15°C for Jan. - Feb., CT = 18°C and PT = -5°C for Mar. - July and CT = 10°C and PT = -12°C for Feb. - May (Environment Canada 2009).
[w]GEI = Gross energy intake (MJ d^{-1}).
[v]DE = Digestible energy intake (MJ d^{-1}).
[u]Sixty five percent of the beef cow population was assumed to be managed on extended grazing (overwintering) and the rest in confinement (J. Kopp, personal communication).
[t]Unknown age (Boadi et al. 2004a).
[s]From the backgrounded steers and heifers 73% go directly to feedlot and the rest to pasture (Boadi et al. 2004a; J. Kopp, personal communication).
[r]Twenty five percent of the provincial breeding bull population is estimated as young (1-2 years of age) (Boadi et al. 2004a).

This equation was used to estimate live weight for steers, heifers, cows and bulls. Average body weight (BW) for steer calves and heifer calves categories was calculated based on weaning weight. Weaning weight of calves was adjusted according to gender, as steers were 5% heavier than heifers at weaning (Basarab et al. 1984; 2005).

Categorization of Manitoba's Beef Cattle Population

Estimation of enteric CH_4 emissions from ruminants using models is influenced by changes in management practices (DeRamus et al. 2003; Garnsworthy 2004). In order to better characterize management practices, the beef cattle population was categorized into 29 subcategories based on physiological status (dry, lactating), management practices (confined, feedlot, grazing), gender, BW, production season, age and growth rate (Table 5.2). Boadi et al. (2004a) conducted a beef cattle management survey in 2001 in order to collect information from beef cattle specialists related to beef cattle production for the different provinces across Canada. The report provided information regarding average BW, average daily gain, duration and type of production environment (pasture vs. confinement) and diet information for the different production stages. Furthermore, current information regarding animal performance and management practices collected by provincial and regional organizations such as MAFRI, Canadian Cattlemen's Association and AAFC were utilized.

Following the detection of bovine spongiform encephalopathy (BSE) in May 2003, more than 40 countries including the U.S. closed their borders to imports of Canadian cattle, and other ruminants (Carlberg et al. 2009). This closure had a significant impact on the Canadian beef cattle industry in terms of beef cattle export, animal market price, beef cattle cycle and management practices. However, there is a lack of documented information on the impact of BSE identification on beef cattle production at the provincial level. Thus, in order to incorporate

the changes in management and marketing practices that resulted from the identification of BSE and the subsequent impact on CH_4 emissions, provincial beef cattle specialists from MAFRI (J. Kopp - Farm Production Extension Specialist, M. German - Director, Livestock Knowledge Center and M. Buchen -Business Development Specialist) were consulted.

The major calving period Manitoba is between January and May (Small and McCaughey 1999; MAFRI beef cattle experts, J. Kopp, M. German and M. Buchen, personal communication). Beef cows were assumed to be managed under two different management systems (confined and pasture; Table 5.2). At the beginning of the census year (January), beef cows were assumed to be in the later stages of pregnancy (3^{rd} trimester). During this period cows were fed high roughage diets (hay and straw) supplemented with (1 to 2.5 kg d^{-1}) barley or oat grain. Management of beef cows using extensive over wintering strategies such as swath grazing or bale grazing is also a common management practice in Manitoba. It is estimated that approximately 35% (MAFRI beef cattle experts, personal communication) of the beef cow population in Manitoba were managed under confinement (drylot) from January to May and November to December (category 1), while the remainder were managed using winter grazing strategies (category 4) except for a period 61 d from February to March, when most of the beef cows are assumed to calve (category 3). For five consecutive months (June – October) cows and their calves were managed on a good quality pasture (64% TDN) as described in category 2 and 4. Average milk production was 6.7 kg d^{-1} with an average fat and protein content of 3.1 and 3.8%, respectively (Kopp et al. 2004). At peak lactation, cows were estimated to produce about 9.6 kg of milk d^{-1}, which was calculated according to Jenkins and Ferrell (1992).

New-born calves with an average birth weight of 44 kg were managed in confinement until moved to pasture (MAFRI beef cattle experts, J. Kopp, M. German and M. Buchen,

personal communication) (category 5). During the summer months, cow-calf pairs grazed until calves were weaned in October or November (category 6). Calves were weaned at 198 d of age (MAFRI beef cattle experts, personal communication) and weighed approximately 275 kg. The census data does not differentiate between male and female calves, therefore, the total number of calves from the census data were split into female and male calves assuming a 0.51 to 0.49 male to female ratio (Boadi et al. 2004a). Enteric CH_4 emissions were considered to be zero for calves in category 4 (birth to pasture), as described by Le Du et al. (1976) who observed that at 90 d of age, calves on high milk consumed less than 1 kg d^{-1} of herbage. Similarly, the USEPA report on GHG does not include enteric CH_4 emission from calves less than 7 months of age (USEPA 2010).

The number of beef and dairy replacement heifers was calculated from the number of calves in the census data by multiplying the average replacement rate of beef heifers (ranged from 5.0 – 15.0%; MAFRI beef cattle experts, personal communication) and dairy heifers (31.0%; Boadi et al. 2004a) by the number of beef and dairy cows in the census data, respectively. Beef cow culling rate was assumed to have declined from 15% to 5% during the period that BSE caused the closure of exports (Canfax 2009). Once the number of replacement calf heifers (Category 7) was deducted from the total calf population, the remaining 27% of the calves were sent for backgrounding (category 8 for heifer calves; category 10 for steer calves) and 73% to feedlot (category 9 for heifer calves; category 11 for steer calves) (Boadi et al. 2004a; MAFRI beef cattle experts, J. Kopp, M. German and M. Buchen, personal communication).

Replacement heifers were assumed to give birth at 24 months of age. Beef replacement heifers were managed under confinement from March to May (category 12) and November to

February (category 14). They were fed a forage-based diet supplemented with barley grain to achieve an average daily gain of 0.63 kg d^{-1}. During the grazing season (June to October) replacement heifers were managed on good quality pasture (category 13; 64% TDN).

Heavier steers and heifers at weaning (73%) were transferred to a high-energy feedlot diet and those with lighter weight (23%) were kept on a forage-based backgrounding diet until they attained the required BW for finishing. Those fed high-energy diets after weaning were slaughtered at 15 months of age (category 15 for heifers; category 20 for steers). From those heifers and steers sent for backgrounding after weaning, about 73% were assumed to be transferred to feedlot and slaughtered at 17 months of age (category 16 for heifers; category 21 for steers). The remaining heifers and steers were backgrounded (category 17 for heifers; category 22 for steers) and fed forage-based diets in confinement. During the grazing season the backgrounded heifer and steers were managed on pasture for three months (Category 18 for heifers; category 23 for steers), and then transferred to high-energy feedlot diet for finishing within a one-month period (category 19 for heifers; category 24 for steers).

The number of breeding bulls over 1 yr of age was taken from the census data and categorized as mature (>2 yr) and young (1 to 2 yr old) assuming that 25% of the bull population in the province were young (Boadi et al. 2004a; Table 5.2). Mature bulls were assumed to be managed in confinement for six months during the winter season (category 25) and fed straw, hay, barley silage and (3 to 4.5 kg d^{-1}) barley grain at maintenance levels, and moved to pasture during the grazing season (category 26). Similarly, young bulls were managed in confinement during winter period (category 27 and category 29) and fed hay or barley silage supplemented with oat or barley grain, and moved to pasture during the grazing season (category 28).

5.3.5. Calculation of Methane Production

Provincial enteric CH_4 emissions from beef cattle (t yr^{-1}) were calculated by multiplying the animal population in each year for a given category and subcategory by its corresponding annual CH_4 EF. The EF is the amount of CH_4 produced annually $animal^{-1}$ and expressed as kg $head^{-1}$ yr^{-1}. For all models, representative EF over the study period (1990 – 2008) was calculated for each animal category using average values.

$$CH_{4j}\ (\text{kg yr}^{-1}) = \sum_{i=1}^{29} C_{ij} \times EF_i \quad , \qquad [5.3]$$

where: CH_{4j} = enteric methane production from all animal categories for the j^{th} yr (1990 – 2008); C_{ij} = animal population for the i^{th} animal category in the j^{th} yr (1990 – 2008); EF_i = annual CH_4 emission factor for the i^{th} animal category (kg $head^{-1}$ yr^{-1}).

The amount of enteric CH_4 production for beef cattle was calculated first for each subcategory by considering the duration of time that cattle are kept in a given production system. This is because most subcategories of cattle spent less than one year under a specified production system. As such, annual enteric CH_4 production was calculated from each category of cattle by multiplying the corresponding EF by the total number of animals in that category for a specific part of the year. Methane emissions calculated using the four methodologies was converted to CO_2 eq by multiplying the amount of CH_4 produced by 21 (IPCC 1995). A factor of 21 was used in the current study as opposed to the 25 factor recommended by (IPCC 2006) in order to compare model estimated values with that of provincial enteric CH_4 inventory values reported by Environment Canada, which were also calculated using a factor of 21 (Environment Canada 2010). Comparison among the four methodologies were calculated as a percentage difference, [(model 1 – model 2)/ model 2] x 100.

Table 5.3. Methane conversion rates (Y_m, % of GEI) for the different categories of Manitoba beef cattle calculated using IPCC Tier-2, Ellis, MOLLY and COWPOLL methodologies[z] and values reported by Canadian research studies (CRS)

No	Animal categories	Management[y]	IPCC Tier-2	Ellis	MOLLY	COWPOLL	CRS
1	Beef cows, pregnant[x]	Confined	6.5	4.7	8.0	6.2	8.3[u]
2	Beef cows, lactating[x]	Pasture	6.5	4.5	8.6	8.1	8.3[u]
3	Beef cows, pregnant[x]	Confined	6.5	4.7	7.6	6.2	8.3[u]
4	Beef cows, lactating[x]	Pasture	6.5	4.6	8.1	7.1	8.3[u]
5	Calves, birth to pasture	Confined	0.0	0.0	0.0	0.0	0.0
6	Calves, pasture	Pasture	6.5	6.7	8.2	7.7	8.2[m]
7	Calves, heifer replacement	Confined	6.5	6.2	6.9	6.1	6.7[o]
8	Calves, heifer backgrounded	Confined	6.5	6.1	6.4	6.4	5.6[n]
9	Calves, heifer feedlot	Feedlot	3.0	4.8	4.6	6.7	4.6[q]
10	Calves, steer backgrounded	Confined	6.5	5.9	6.3	6.4	5.6[n]
11	Calves, steer feedlot	Feedlot	3.0	4.8	4.6	6.7	2.5[p]
12	Heifers replacement	Confined	6.5	5.5	6.8	6.0	6.7[o]
13	Heifers replacement	Pasture	6.5	5.4	8.5	7.7	8.2[m]
14	Heifers replacement	Confined	6.5	4.5	5.2	5.9	6.7[o]
15	Finisher, heifers	Feedlot	3.0	4.3	4.6	6.7	4.6[q]
16	Finisher heifers-backgrounded[w]	Feedlot	3.0	4.3	4.7	6.7	4.6[q]
17	Backgrounded heifers	Confined	6.5	5.1	6.2	6.5	7.4[r]
18	Backgrounded heifers	Pasture	6.5	4.2	8.2	7.2	8.2[m]
19	Finisher, heifers	Feedlot	3.0	4.1	4.6	6.4	4.6[q]
20	Finisher, steers	Feedlot	3.0	4.3	4.6	6.7	2.5[p]
21	Finisher steers-backgrounded[w]	Feedlot	3.0	4.3	4.7	6.7	2.5[p]
22	Backrounded, steers	Confined	6.5	5.1	6.7	6.3	5.5[s]
23	Backgrounded, steers	Pasture	6.5	4.8	8.1	7.2	6.7[t]
24	Finisher, steers	Feedlot	3.0	4.1	4.6	6.4	2.5[p]
25	Breeding bulls, mature	Confined	6.5	3.7	8.2	5.2	5.7[l]
26	Breeding bulls, mature	Pasture	6.5	3.1	8.7	7.1	8.7[s]
27	Breeding bulls, young[v]	Confined	6.5	4.5	5.9	5.8	5.5[s]
28	Breeding bulls, young[v]	Pasture	6.5	4.1	8.7	7.4	8.7[s]
29	Breeding bulls, young[v]	Confined	6.5	3.6	6.0	5.6	5.5[s]

[z]IPCC Tier-2 = Intergovernmental Panel on Climate Change Tier-2 methodology (IPCC 2006), Ellis = nonlinear model developed by Ellis et al. (2009), MOLLY = dynamic mechanistic model (Baldwin 1995), COWPOLL = dynamic mechanistic model (Dijkstra et al. 1992).

[y]Confined: winter housing, which is drylot with barn (shade) (Small and McCaughey, 1999).

[x]Sixty five percent of the beef cow population was assumed to be managed on extended grazing (overwintering) and the rest in confinement (J. Kopp, personal communication).

[w]From the backgrounded steers and heifers 73% go directly to feedlot and the rest to pasture (Boadi et al. 2004a; J. Kopp, personal communication).

[v]Twenty five percent of the provincial breeding bull population is estimated as young (1 - 2 years of age) (Boadi et al. 2004a).

[u]Based on early lactating first calve heifers, 511 kg BW grazing a grass pasture, producing 411 L CH_4 d^{-1} (McCaughey et al. 1999).

[t]Yearling steers (13 mo), 356 kg BW, grazing a mixed pasture (60% alfalfa, 27% meadow bromegrass, 5% Russian wildrye) using two grazing systems (continuous and rotational) and two stocking rates (high and low), producing 257.3 L CH_4 d^{-1} (McCaughey et al. 1997); 343 kg yearling steers grazing on grass-based pasture, producing 197.5 L CH_4 d^{-1} (Ominski et al. 2006) and 343 kg yearling steers grazing on alfalfa-meadow bromegrass mixed pasture, producing 310.5 L CH_4 d^{-1} (Boadi et al. 2002).

[s]Based on 262 kg steers fed an alfalfa-grass silage diet (*ad-libitum*) ranging in quality, producing 193.3 L CH_4 d^{-1}, and 343 kg yearling steers grazing a grass-based pasture, producing 197.5 L CH_4 d^{-1} (Ominski et al. 2006).

[r]Based on 328 kg heifers (9 mo) fed a barley silage-based diet (70% barley silage and 25% corn grain) or a corn silage-based diet

(70% corn silage and 25% barley grain), producing 208.6 L CH_4 d^{-1}; monensin was added to the diet to provide 33 mg kg^{-1} of total dietary DM (Beauchemin and McGinn 2005).

[q]Based on 419 kg heifers fed a barley grain-based diet (81% ground barley grain 9% barley silage) or a corn grain-based diet (81% steam-rolled corn gain and 9% barley silage), producing 99.0 L CH_4 d^{-1} monensin was added to the diet to provide 33 mg kg^{-1} of total dietary DM (Beauchemin and McGinn 2005).

[p]Based on 300 kg steers (6 mo) fed an 83.5% barley grain and 11.5% barley silage, producing 127.9 L CH_4 d^{-1} (Boadi et al. 2004b).

[o]Based on 310 kg confined, yearling heifers fed grass legume hay (ad-libitum), producing 258.7 L CH_4 d^{-1} (Boadi and Wittenberg 2002).

[n]Based on 223 kg heifer and steers (6 - 8 mo) fed a barley silage based diet, producing 137.1 L CH_4 d^{-1} (Beauchemin et al. 2007).

[m]Based on 380 kg yearling heifers grazing a grass pasture, producing 228.9 L CH_4 d^{-1}(Chaves et al. 2006)

[l]Based on 531 kg steers fed at maintenance, 50% bromegrass and 50% alfalfa hay (DM basis), producing 188.8 L CH_4 d^{-1} (Okine et al. 1989).

Model estimated Y_m values for the different animal categories were compared with Y_m values reported by Canadian research studies (Okine et al. 1989, McCaughey et al. 1997; 1999, Boadi and Wittenberg 2002, Boadi et al. 2002, Boadi et al. 2004b, Beauchemin and McGinn 2005, Chaves et al. 2006, Ominski et al. 2006 and Beauchemin et al. 2007; Table 5.3, 5.4). The Y_m values reported by Canadian research studies were calculated based on measured CH_4 values. Methane emissions were measured using whole-animal respiration calorimetry (2 studies) and open-circuit indirect calorimetry: hood chamber (1 study) and the SF_6 tracer gas technique (7 studies). Comparison of model estimated Y_m values vs. values reported by Canadian research studies were calculated as a percentage difference, [(model 1 – model 2)/ model 2] x100 (Table 5.4).

5.4. RESULTS

5.4.1. Methane Conversion Rates and Emission Factors

Methane conversion rates (Y_m) estimated using IPCC Tier-2, Ellis, MOLLY, COWPOLL methodologies and from Canadian research studies are provided in Table 5.3. Relative differences among models in estimating the Y_m values, regardless of animal category, were 16.7% for Tier-2 and Ellis, 0.4% for COWPOLL and MOLLY, and 16.6% for Tier-2 and the mechanistic models. The average estimated Y_m value for mature beef cattle (beef cows and breeding bulls) were 4.2% of GEI using Ellis, 8.2% of GEI using MOLLY and 6.7% of GEI using COWPOLL relative to the 6.5% of GEI default value used by IPCC Tier-2. For beef cows grazing on pasture, the Y_m value estimated using MOLLY was higher (8.3% of GEI) followed by COWPOLL (7.6% of GEI) and IPCC Tier-2 (6.5% of GEI). Methane conversion rates estimated using MOLLY were the highest for all animal categories compared to Ellis and IPCC Tier-2. In contrast, Y_m values calculated using Ellis were the lowest for all animal categories compared to

the two mechanistic models. For young growing beef cattle, the average Y_m value estimated using MOLLY was the highest (7.0% of GEI) followed by COWPOLL (6.6% of GEI), IPCC Tier-2 (6.5% of GEI), and Ellis (5.1% of GEI). Specifically, Y_m values calculated using Ellis for replacement heifers and backgrounding animals were 5.1 and 5.2% of GEI, respectively. However, for the same animal categories MOLLY estimated 6.9 and 7.0% of GEI and COWPOLL estimated 6.6 and 6.7% of GEI, respectively. For the feedlot category, the calculated average Y_m value was the highest for COWPOLL (6.6% of GEI) followed by MOLLY (4.6% of GEI), Ellis (4.4% of GEI) and IPCC Tier-2 (3.0% of GEI).

A comparison of Y_m values estimated using models and those reported by Canadian research studies is provided in Table 5.4. Average Y_m values from all animal categories estimated using MOLLY and COWPOLL were similar (7.7% and 8.1% difference, respectively) to the Y_m values reported in Canadian research studies. The greatest difference was observed for IPCC Tier-2 (-10.1%) and Ellis (-22.9%). An examination of estimated average Y_m value for categories related to cow-calf production system (category 1 - 7, 12 - 14) indicated that MOLLY resulted in the smallest difference (2.4%) while Ellis resulted in the greatest difference (-30.0%) as compared to Canadian research data (Table 5.4). However, for categories related to feedlot production systems (category 9, 11, 15, 16, 19 - 21, 24), the average Y_m value calculated using IPCC Tier-2 had the smallest difference (-15.5%) and COWPOLL the greatest difference (86.5%) relative to the average Y_m values reported by Canadian research studies. Emission factors for the enteric fermentation of Manitoba's beef cattle calculated using the four methodologies are shown in Table 5.5.

Table 5.4. Comparison of methane conversion rate values calculated using IPCC Tier-2, Ellis, MOLLY and COWPOLL methodologies[z] with values from Canadian research studies (CRS)

No	Animal categories	Management[y]	CRS vs. Tier-2 (% difference)[x]	CRS vs. Ellis (% difference)[x]	CRS vs. MOLLY (% difference)[x]	CRS vs. COWPOLL (% difference)[x]
1	Beef cows, pregnant[w]	Confined	27.7	75.6	4.1	34.1
2	Beef cows, lactating[w]	Pasture	27.7	86.2	-2.9	2.5
3	Beef cows, pregnant[w]	Confined	27.7	75.5	9.6	34.1
4	Beef cows, lactating[w]	Pasture	27.7	81.6	2.7	16.3
5	Calves, birth to pasture	Confined	0.0	0.0	0.0	0.0
6	Calves, pasture	Pasture	26.2	21.7	-0.1	6.3
7	Calves, heifer replacement	Confined	3.1	8.5	-2.2	10.6
8	Calves, heifer backgrounded	Confined	-13.8	-7.6	-12.4	-12.6
9	Calves, heifer feedlot	Feedlot	53.3	-3.8	-1.0	-31.6
10	Calves, steer backgrounded	Confined	-13.8	-4.5	-10.4	-12.2
11	Calves, steer feedlot	Feedlot	-16.7	-47.7	-46.2	-62.8
12	Heifers replacement	Confined	3.1	21.9	-2.1	11.8
13	Heifers replacement	Pasture	26.2	51.8	-3.9	6.3
14	Heifers replacement	Confined	3.1	49.8	28.4	12.9
15	Finisher, heifers	Feedlot	53.3	5.8	-0.3	-30.9
16	Finisher heifers-backgrounded[v]	Feedlot	53.3	5.8	-2.7	-30.9
17	Backgrounded heifers	Confined	13.8	45.4	19.5	14.7
18	Backgrounded heifers	Pasture	26.2	94.4	-0.4	13.7
19	Finisher, heifers	Feedlot	53.3	12.8	0.0	-28.6
20	Finisher, steers	Feedlot	-16.7	-42.5	-45.8	-62.5
21	Finisher steers-backgrounded[v]	Feedlot	-16.7	-42.5	-47.1	-62.5
22	Backrounded, steers	Confined	-15.4	8.7	-18.1	-12.1
23	Backgrounded, steers	Pasture	3.1	40.7	-17.5	-7.0
24	Finisher, steers	Feedlot	-16.7	-38.7	-45.6	-61.2
25	Breeding bulls, mature	Confined	-12.3	55.5	-30.7	8.9
26	Breeding bulls, mature	Pasture	33.8	177.3	0.2	21.8
27	Breeding bulls, young[u]	Confined	-15.4	22.2	-7.0	-4.4
28	Breeding bulls, young[u]	Pasture	33.8	110.0	0.5	17.9
29	Breeding bulls, young[u]	Confined	-15.4	53.9	-7.7	-2.1

[z]IPCC Tier-2 = Intergovernmental Panel on Climate Change Tier-2 methodology (IPCC 2006), Ellis = nonlinear model developed by Ellis et al. (2009), MOLLY = dynamic mechanistic model (Baldwin 1995), COWPOLL = dynamic mechanistic model (Dijkstra et al. 1992).

[y]Confined: winter housing, which is drylot with barn (shade) (Small and McCaughey, 1999).

[x]Calculated as [(model 1 – model 2)/model 2] x 100.

[w]Sixty five percent of the beef cow population was assumed to be managed on extended grazing (overwintering) and the rest on confinement (J. Kopp, personal communication).

[v]From the backgrounded steers and heifers 73% go directly to feedlot and the rest to pasture (Boadi et al. 2004a; J. Kopp, personal communication).

[u]Twenty five percent of the provincial breeding bull population is estimated as young (1 - 2 years of age) (Boadi et al. 2004a).

In general, regardless of the types of models used, average EF values varied from 65.6 to 131.1 for beef cows on pasture, 57.3 to 113.7 for beef cows managed under confinement, 67.3 to 135.3 for breeding bulls, 56.2 to 104.5 for young growing beef cattle on pasture and 33.8 to 85.9 kg $head^{-1}$ $yr^{-1}$for feedlot cattle. More specifically, average EF for beef cows, breeding bulls and replacement heifers calculated using IPCC Tier-2 were 82.8, 112.9 and 74.1 kg $head^{-1}$ yr^{-1}, respectively, which were similar to EF calculated using COWPOLL being 95.1, 115.1 and 78.5 kg $head^{-1}$ yr^{-1}, respectively (Table 5.5). MOLLY had the highest EF estimates (122.4, 135.3 and 94.0 kg $head^{-1}$ yr^{-1}, respectively) and Ellis had the lowest EF estimates (65.2, 67.3 and 62.3 kg $head^{-1}$ yr^{-1}, respectively) for the same beef cattle categories. For feedlot category, IPCC Tier-2 had the lowest average EF estimates (33.8 kg $head^{-1}$ yr^{-1}) and COWPOLL had the highest average EF estimate (85.9 kg $head^{-1}$ yr^{-1}).

5.4.2. Enteric CH_4 Production Trends

Enteric CH_4 emission trends from Manitoba beef cattle are provided in Table 5.6 and Figure 5.1. In general, enteric CH_4 production from Manitoba beef cattle calculated using IPCC Tier-2 ranged between 1.2 and 2.0 Mt CO_2 eq, 0.9 and 1.5 Mt CO_2 eq using Ellis, 1.5 and 2.4 Mt CO_2 eq using MOLLY and 1.3 and 2.1 Mt CO_2 eq using COWPOLL. The maximum production estimates were in 2005 for all models. Enteric CH_4 production calculated using IPCC Tier-2 was 1.2 and 1.7 Mt CO_2 eq for 1990 and 2008, respectively, which was higher than when these annual emission values were calculated using Ellis (0.9 and 1.3 Mt CO_2 eq for 1990 and 2008, respectively). Emission estimates calculated using mechanistic models were higher than both IPCC Tier-2 and Ellis (Table 5.6) with the differences being greater for MOLLY than COWPOLL.

Table 5.5. Enteric CH_4 emission factor (EF) for the different categories of Manitoba beef cattle calculated using IPCC Tier-2, Ellis, MOLLY and COWPOLL Methodologies[z] (kg head^{-1} yr^{-1})

No	Animal categories	Age (month)	Management[y]	Time of yr	IPCC Tier-2	Ellis	MOLLY	COWPOLL
1	Beef cows, pregnant[x]	Uknown[w]	Confined	Jan. - May/Nov. - Dec.	57.3	64.8	113.6	79.6
2	Beef cows, lactating[x]	Uknown[w]	Pasture	June - Oct.	108.3	65.7	147.3	118.0
3	Beef cows, pregnant[x]	Uknown[w]	Confined	Feb. - Mar.	57.3	64.8	113.9	79.6
4	Beef cows, pregnant[x]	Uknown[w]	Pasture	Jan./Apr. - Dec..	108.4	65.4	114.9	103.4
5	Calves, birth to pasture	0 to 2.6	Confined	Mar.- May	0.0	0.0	0.0	0.0
6	Calves, pasture	2.6 to7.3	Pasture	June - Oct.	45.4	24.4	30.3	27.5
7	Calves, heifer replacement	7.3 to 12	Confined	Nov. - Dec./Jan. - Mar.	55.1	56.5	59.8	52.6
8	Calves, heifer backgrounded	7.3 to 12	Confined	Nov.- Dec./Jan. - Mar.	60.1	54.2	55.6	58.9
9	Calves, heifer feedlot	7.3 to 12	Feedlot	Nov. - Dec./Jan. - Mar.	41.3	47.8	49.4	70.5
10	Calves, steer backgrounded	7.3 to 12	Confined	Nov. - Dec./Jan. - Mar.	56.7	56.3	57.5	62.6
11	Calves, steer feedlot	7.3 to 12	Feedlot	Nov. - Dec./Jan. - Mar.	26.4	47.8	49.4	70.5
12	Heifers replacement	13 to 15	Confined	Mar. - May	61.2	60.5	74.4	61.6
13	Heifers replacement	16 to 20	Pasture	June - Oct.	76.4	61.1	100.5	93.2
14	Heifers replacement	21 to 24	Confined	Nov. - Dec./Jan. - Feb.	84.5	65.4	107.1	80.8
15	Finisher, heifers	13 to 15	Feedlot	Mar. - May	32.1	54.1	62.6	87.5
16	Finisher heifers-backgrounded[v]	13 to 17	Feedlot	Mar. - July	33.8	54.1	63.2	87.5
17	Backgrounded heifers	13 to 15	Confined	Mar. - May	71.8	61.3	79.5	78.6
18	Backgrounded heifers	16 to 18	Pasture	June - Aug.	114.0	65.8	127.7	111.7
19	Finisher, heifers	19	Feedlot	Sep.	38.3	56.8	74.9	98.1
20	Finisher, steers	13 to 15	Feedlot	Mar. - May	30.3	54.1	62.6	87.5
21	Finisher steers-backgrounded[v]	13 to 17	Feedlot	Mar. - July	31.9	54.1	63.2	87.5
22	Backrounded, steers	13 to 15	Confined	Mar. - May	68.2	61.4	86.7	76.8
23	Backgrounded, steers	16 to 18	Pasture	June - Aug.	108.2	62.9	130.8	114.0
24	Finisher, steers	19	Feedlot	Sep.	36.0	56.8	74.9	98.1
25	Breeding bulls, mature	Uknown[w]	Confined	Jan. - May/Dec.	103.6	68.2	154.4	103.1
26	Breeding bulls, mature	Uknown[w]	Pasture	June - Nov.	114.6	69.4	183.0	156.8
27	Breeding bulls, young[u]	13 to 16	Confined	Feb. - May	90.8	64.6	87.4	87.1
28	Breeding bulls, young[u]	17 to 22	Pasture	June - Nov.	129.2	66.6	133.3	118.3
29	Breeding bulls, young[u]	23 to 24	Confined	Dec./Jan.	126.5	68.0	118.5	110.5

[z]IPCC Tier-2 = Intergovernmental Panel on Climate Change Tier-2 methodology (IPCC 2006), Ellis = nonlinear model developed by Ellis et al. (2009), MOLLY = dynamic mechanistic model (Baldwin 1995), COWPOLL = dynamic mechanistic model (Dijkstra et al. 1992).

[y]Confined: winter housing, which is drylot with barn (shade) (Small and McCaughey, 1999).
[x]Sixty five percent of the beef cow population was assumed to be managed on extended grazing (overwintering) and the reminder in confinement (J. Kopp, personal communication).
[w]Uknown age (Boadi et al. 2004a).
[v]From the backgrounded steers and heifers 73% go directly to feedlot and the rest to pasture (Boadi et al. 2004a; J. Kopp, personal communication).
[u]Twenty five percent of the provincial breeding bull population is estimated as young (1 - 2 years of age) (Boadi et al. 2004a).

Comparing model estimated CH_4 production values with that of provincial enteric CH_4 production estimates reported by Environment Canada (Environment Canada 2010) for 1990 - 2008 indicated that values calculated using IPCC Tier-2 were 0.9 to 9.6% lower and those calculated using MOLLY were 11.5 to 22.2% higher (Figure 5.1).

Regardless of the variation among models in estimating enteric CH_4 production, the overall trend of enteric CH_4 production was similar for all models, and paralleled the emission trends calculated from Environment Canada (2010) for Manitoba (Figure 5.1). Methane production tended to increase between 1990 and 1996, remain relatively constant between 1996 and 2002, increase between 2003 and 2005 and decrease between 2006 and 2008.

5.5. DISCUSSION

The Government of Manitoba has set goals to reduce GHG from various sectors to meet the Kyoto target of a 17.5 Mt CO_2 eq (6% less than the 1990 level) by 2012. This may require a 34.5% reduction in GHG emissions from agricultural sector and a 40.8% reduction in enteric CH_4 production from beef cattle by 2012 relative to the 2008 emission level of 7.6 Mt CO_2 eq for agricultural sector and 1.9 Mt CO_2 eq for enteric CH_4 emissions from beef cattle (Environment Canada 2010). Considering the long-term trend of GHG emissions from the agricultural sector in Manitoba, a 2.3 Mt CO_2 eq increase between 1990 and 2008 (Environment Canada 2010), the likelihood of the target being met within the next two years is minimal. Therefore, in addition to implementing the various mitigation strategies, accuracy of the models that are used to estimate emissions needs to be investigated. Strategies to mitigate enteric CH_4 to meet these goals include manipulation of rumen fermentation, improving animal productivity and dietary management, all of which has been the subject of previous reviews (Boadi et al. 2004c; Kebreab et al. 2006b; Beauchemin et al. 2008). The current study suggested that variation exists among models in

estimating enteric CH_4 production and these should be considered in emission estimates.

5.5.1. Methane Conversion Rates and Emission Factors

Methane conversion rates for Ellis, MOLLY and COWPOLL were calculated from the GEI for a specific animal category. However, IPCC Tier-2 has standard Y_m values of 3 ± 1% of GEI for feedlot cattle and 6.5 ± 1% of GEI for other cattle categories (IPCC, 2006). Variation between IPCC Tier-2 and MOLLY on Y_m estimates ranged from -19.2% to -5.3% for beef cows, breeding bulls and replacement heifers. However, the variation was minimal between IPCC Tier-2 and COWPOLL (-5.8% to 4.4%) for the same animal category. Compared to the other models, the Y_m values generated by Ellis were lower for all animal categories except for feedlot animals, where the Y_m estimate was 46.3% higher than the IPCC Tier-2 estimate for the same animal category. The lower Y_m estimates for the Ellis model may be a consequence of using the model outside of the production system and animal category on which it was originally developed. The model was developed using data collected from growing and feedlot beef cattle (Ellis et al. 2009). It is known that numerous factors including management, animal performance and diet characteristics contribute to variability in enteric CH_4 emissions (Boadi et al. 2004c). Empirical (statistical) models do not account for these factors and thus they fail to account for variability arising from these aforementioned factors. Ellis et al. (2010) evaluated the existing empirical models used in whole-farm models to estimate CH_4 production using independent data and found that variation observed in equations were much higher than variation observed from measured values.

Table 5.6. Total methane production from Manitoba beef cattle calculated using IPCC Tier-2, Ellis, MOLLY and COWPOLL methodologies[z]

	IPCC Tier-2		Ellis		MOLLY		COWPOLL	
Year	t CH_4 yr^{-1}	Mt CO_2 eq yr^{-1}[y]	t CH_4 yr^{-1}	Mt CO_2 eq yr^{-1}[y]	t CH_4 yr^{-1}	Mt CO_2 eq yr^{-1}[y]	t CH_4 yr^{-1}	Mt CO_2 eq yr^{-1}[y]
1990	56122.7	1.2	42411.2	0.9	69215.8	1.5	60813.3	1.3
1991	58949.9	1.2	44689.6	0.9	72468.4	1.5	63901.1	1.3
1992	62188.3	1.3	47041.3	1.0	76355.6	1.6	67326.2	1.4
1993	62908.8	1.3	47505.7	1.0	77283.5	1.6	67963.7	1.4
1994	67664.4	1.4	51046.3	1.1	83005.1	1.7	73034.8	1.5
1995	71954.2	1.5	54574.2	1.1	88256.2	1.9	78045.8	1.6
1996	77013.5	1.6	58356.0	1.2	94517.1	2.0	83550.7	1.8
1997	76851.9	1.6	58287.6	1.2	94771.9	2.0	83701.3	1.8
1998	75421.7	1.6	57219.0	1.2	92751.8	1.9	82107.9	1.7
1999	74969.9	1.6	56716.9	1.2	92093.5	1.9	81361.4	1.7
2000	76227.3	1.6	57699.0	1.2	93522.1	2.0	82729.9	1.7
2001	77452.0	1.6	58760.6	1.2	95175.7	2.0	84130.4	1.8
2002	79485.3	1.7	60337.2	1.3	97892.5	2.1	86508.8	1.8
2003	87994.0	1.8	66907.1	1.4	108788.1	2.3	96778.3	2.0
2004	91748.5	1.9	69953.6	1.5	113214.5	2.4	101139.9	2.1
2005	93546.6	2.0	71129.7	1.5	115521.8	2.4	102359.3	2.1
2006	89285.8	1.9	67681.5	1.4	110032.7	2.3	97406.5	2.0
2007	83757.8	1.8	63578.2	1.3	103425.7	2.2	91582.3	1.9
2008	81276.8	1.7	61631.1	1.3	100193.5	2.1	88796.2	1.9

[z]IPCC Tier-2 = Intergovernmental Panel on Climate Change Tier-2 methodology (IPCC 2006), Ellis = nonlinear model developed by Ellis et al. (2009), MOLLY = dynamic mechanistic model (Baldwin 1995), COWPOLL = dynamic mechanistic model (Dijkstra et al. 1992).

[y]Mt CO_2 eq calculated as (enteric CH_4 emissions (t yr^{-1}) x 21)/10^6.

Therefore, the reliability of empirical models to predict emission estimates when they are applied beyond the production systems in which they were developed may be questionable (Mills 2008).

Comparing the two mechanistic models, the average Y_m values calculated using MOLLY were 9.2 and 19.4% higher than that of COWPOLL (7.6 and 7.3% GEI), for beef cows and breeding bulls managed on pasture, respectively (Table 5.3). Methane conversion rates calculated for beef cows on pasture using MOLLY and COWPOLL were comparable to the values reported by McCaughey et al. (1999) for first calf cows (511 kg BW) grazing pasture (7.1 to 9.5% GEI) with the highest Y_m observed for cows grazing grass pasture and lowest value for cows grazing alfalfa-grass pasture.

For young growing beef cattle on pasture, average Y_m value estimated using IPCC Tier-2 was 12.7 and 22.2% lower than the values estimated using COWPOLL and MOLLY, respectively. Methane conversion rate values calculated using mechanistic models for growing steers fed on pasture were comparable to values reported by Ominski et al. (2006), which ranged between 6.9 and 11.3% of GEI, for growing steers (343 kg BW) grazing low fertility pasture typical to Manitoba (Small and McCaughey 1999). Similarly, the Y_m value estimated by mechanistic models (7.2 – 8.1% of GEI) for growing heifers on pasture was in agreement with the Y_m values reported by Chaves et al. (2006) which ranged between 5.8 and 8.2% of GEI. An earlier study by McCaughey et al. (1997) reported lower Y_m values (4.1 to 5.2 % of GEI) for growing steers on pasture (356 kg BW), which is similar to the Y_m value estimated using Ellis model (4.8% of GEI). In the study of McCaughey et al. (1997), steers were grazed on pasture containing a higher proportion of alfalfa (60%), which is not typical in Manitoba beef cattle production systems. The lower Y_m values for McCaughey et al. (1997) could also be explained by the higher DMI reported. Legume-based pastures increase voluntary intake as compared to

grass-based pastures as digestibility and rate of passage are higher for legume compared to grass (Minson and Wilson 1994), thereby resulting in lower enteric CH_4 emissions (Okine et al. 1989; McCaughey et al. 1999).

The default Y_m value of IPCC Tier-2 (3% of GEI) for feedlot animals was 35.4% and 54.7% lower compared to MOLLY and COWPOLL, respectively (Table 5.3). The values for MOLLY (4.6% of GEI) and COWPOLL (6.6% of GEI) were higher as compared to the Y_m value reported by Boadi et al. (2004b), which was 2.5% GEI for feedlot steers (300 kg BW) consuming diets containing 84% barley grain and 11.5% barley silage. Beauchemin and McGinn (2005) also reported Y_m of 2.8 and 4.0%, respectively, for feedlot heifers (419 kg BW) fed diets containing 81% corn and 81% barley grain. The higher Y_m value calculated by MOLLY and COWPOLL for feedlot cattle compared to IPCC (2006) and those reported in the literature (Boadi et al. 2004b; Beauchemin and McGinn 2005) may be attributed to the VFA stoichiometry used in the models to estimate VFA production. For instance, the stochiometric model of COWPOLL was developed based on diets formulated to meet the nutritional requirement of lactating dairy cows that contain a lower proportion of concentrate as compared to feedlot diets (Bannink et al. 2006). The applicability of the COWPOLL model for a wide range of dietary conditions has been questioned (Sveinbjörnsson et al. 2006). Therefore, VFA stoichiometry may underestimate the amount of propionic acid produced in feedlot cattle fed high concentrate diets (Kebreab et al. 2008). It has been demonstrated that feeding high-grain diets will increase propionic acid production and decrease CH_4 production (Moss et al. 2000). Hence, using Y_m values calculated by MOLLY and COWPOLL for estimation of CH_4 emissions might overestimate CH_4 emissions from feedlot cattle. Additionally, Y_m values from research studies on feedlot cattle may be lower due to the use of different alternative feeding strategies such as

inclusion of ionophores which may suppress enteric CH_4 emissions (Guan et al. 2006). Evaluation of MOLLY, COWPOLL and IPCC Tier-2 by Kebreab et al. (2008) using data from U.S. dairy and feedlot cattle suggest that CH_4 emissions for feedlot cattle are more accurately predicted by MOLLY than by COWPOLL. In the current study, COWPOLL estimated higher Y_m values for feedlot animals compared to MOLLY.

Variation was observed between model estimated Y_m values and those reported by Canadian research studies (Table 5.4). Performance of a given model depends not only on the assumptions and hypothesis made within the model to represent the biological phenomenon, but also on the accuracy of information (input parameters) provided to the model (Benchaar et al. 1998). The paucity of information related to diet composition, DMI, animal performance in the current study could contribute to the observed variation. Given the strong relationship between DMI and CH_4 production (Grainger et al. 2007), the use of estimated DMI values for the individual animal categories as an input for Ellis, MOLLY and COWPOLL could influence the model estimates, contributing to the observed variation. Furthermore, the observed variation in Y_m estimates between the mechanistic models (Table 5.3) could be due to difference in the interpretation of fermentation stoichiometry (Bannink et al. 2006), rumen microbes (Nagorcka et al. 2000) and dietary chemical composition (Benchaar et al. 1998). Compared to COWPOLL, MOLLY requires, in addition to NDF, starch and soluble sugar content, feed information on organic acid and pectin. Benchaar et al. (1998) indicated that inclusion of dietary pectin concentration as a separate model input improved the CH_4 prediction potential of the model of Dijkstra et al. (1992). On the other hand, Nagorcka et al. (2000) indicated that inclusion of total rumen microbial population as three microbial groups (amylolytic bacteria, cellulolytic bacteria and protozoa) in the fermentation stoichiometry of the rumen models improved VFA prediction,

which also influenced enteric CH_4 estimates. In general, studies on evaluation of MOLLY, COWPOLL, IPCC Tier-2 and other empirical models using independent measured CH_4 data collected from dairy and beef cattle suggested that Y_m values are better predicted by mechanistic rumen models than empirical models (Kebreab et al. 2006b; 2008).

Emission factor values calculated using IPCC Tier-2 and COWPOLL for beef cows on pasture (108.4 kg head^{-1} yr^{-1}) were comparable to values reported by McCaughey et al. (1999) for first-calf heifers grazing grass pastures in Manitoba (108.2 kg head^{-1} yr^{-1}). Furthermore, for feedlot cattle the average EF value estimated using IPCC Tier-2 (33.8 kg head^{-1} yr^{-1}) was comparable to that of 33.6 kg head^{-1} yr^{-1} reported by Boadi et al. (2004b). However, comparing the two mechanistic models for feedlot cattle, the EF calculated using COWPOLL was higher (37.4%) than EF calculated using MOLLY. This difference could also be due to the difference in VFA stoichiometry used to predict VFA profile from dietary nutrients as previously discussed.

5.5.2. Enteric CH_4 Production Trend

Annual enteric CH_4 production is a function of the annual EF and animal population (equation 3). Variation in estimates of enteric CH_4 production calculated using different methodologies was due to the differences in calculated annual EF among the models (Table 5.5). The EF calculated for beef cows using IPCC Tier-2 was 82.8 kg head^{-1} yr^{-1} in 1990 which resulted in an annual CH_4 emissions of 1.2 Mt CO_2 eq. Whereas, the EF calculated using MOLLY for the same category in the same year was 122.4 kg head^{-1} yr^{-1} which resulted in an annual emission of 1.5 Mt CO_2 eq. Relative to the annual enteric CH_4 production calculated using IPCC Tier-2 in the current study, values calculated from Environment Canada (Environment Canada 2010), had 0.9 to 9.6% higher estimates (Figure 5.1). This difference could be attributed to the fact that national EF values were used to calculate CH_4 emissions for the major beef cattle category of Manitoba.

For example, EF used for 2001 by Environment Canada was higher for replacement heifers (73.3 kg $head^{-1}$ yr^{-1}) and lower for beef cows (86.8 kg $head^{-1}$ yr^{-1}) compared to the values reported by Boadi et al. (2004a) for Manitoba, 71.8 and 89.4 kg $head^{-1}$ yr^{-1}, respectively.

Despite the type of models used, enteric CH_4 production from Manitoba beef cattle increased by 45.3% between the year 1990 and 2008 (Figure 5.1, Table 5.6), an estimate which is lower than the 56.0% reported by Environment Canada (2010) for the same period. Specifically, CH_4 production increased by 37.1% and 6.1% between 1990 and 1996 and 2003 and 2005, respectively, but decreased by 13.2% between 2006 and 2008. The overall trend of enteric CH_4 production was similar for all models, and paralleled the provincial emission trend calculated from Environment Canada (2010). This trend was mainly due to change in beef cattle population associated with the cattle cycle (Canfax 2009; Table 5.1).

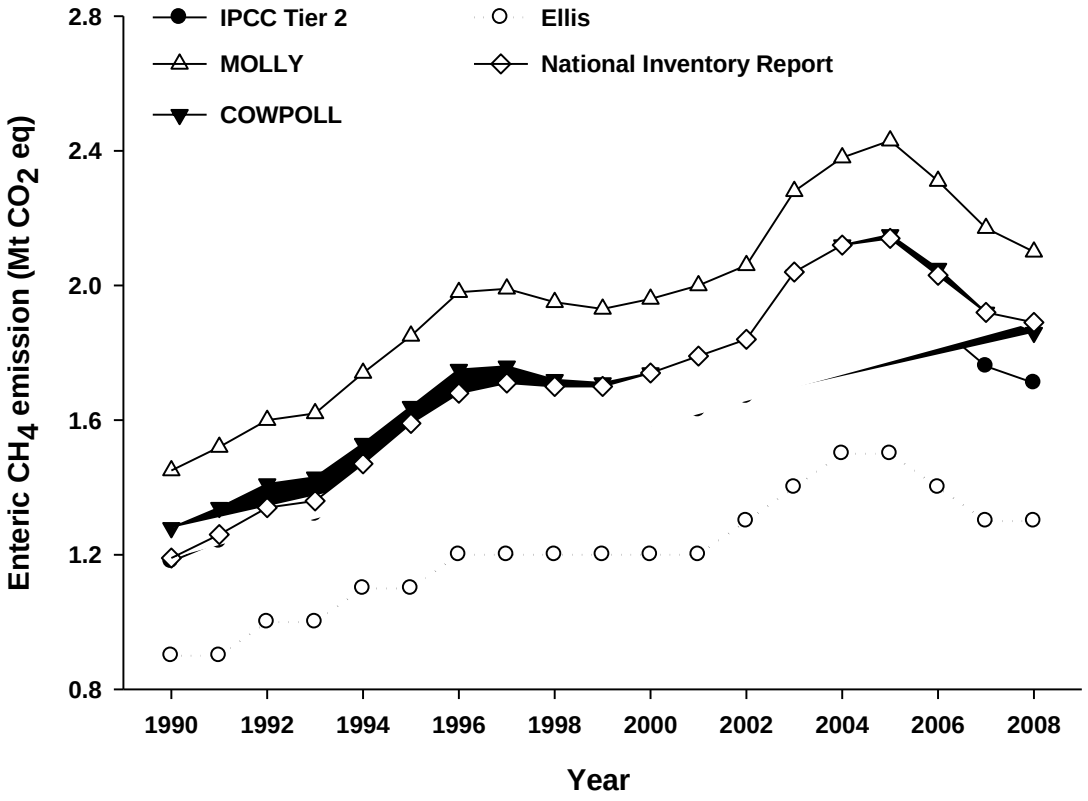

Figure 5.1. Pattern of enteric CH_4 production for Manitoba beef cattle calculated using IPCC Tier-2, Ellis, MOLLY, COWPOLL and values taken from National Inventory Report (NIR, Environment Canada 2010, Part 1, 2 and 3) for the year 1990 to 2008. For the NIR, enteric CH_4 production for Manitoba beef cattle was calculated based on national emission factor (kg CH_4 $head^{-1}$ yr^{-1}) for the major beef cattle categories.

Various factors contributed to fluctuation in the beef cattle population of Manitoba including market price, feed cost, live animal export, animal disease, U.S. exchange rate of the Canadian dollar and consumer demand. According to Canfax (2009), higher market price, lower feed cost, and increased live animal export were the major driving forces leading to the beef cattle population increase in Canada from 1986 to 1996. The beef cattle population in Manitoba increased by 38.6% in 1996 relative to the 1990 population, which resulted in a 37.1% increase in the province's enteric CH_4 production. Another increase in the beef cattle population occurred between 2003 and 2005, due to the occurrence of BSE which lead to a significant reduction in the demand for Canadian beef (Carlberg and Brewin 2005; Carlberg et al. 2009). As a result of reduced market demand for all classes of cattle, many beef cattle producers retained ownership for longer periods of time. Further, the price of cull beef cows in Manitoba dropped by more than 75% from an average price $65.7per hundred weight, pre-BSE (Jan. 2000 – Mar. 2003) to an average price of $15.9 per hundred weight (June 2003 – Sep. 2004; MAFRI 2004). As a consequence, culling rate of beef cows between 2003 and 2005 declined significantly from 15% to 5% (Canfax 2009), leading to a 7.1% increase in the Manitoba beef cow population (Table 5.1), and a 6.1% increase in the province's enteric CH_4 production. Following reopening of the U.S. border for younger live animals (< 30 months of age) in 2005 and for all live animals in 2007, the number of beef cattle declined as cattle began to move across the border for feeding and slaughter. As a result, CH_4 production declined by 13.2% between 2005 and 2008 (Figure 5.1).

It is apparent that there is a wide variation among models in estimating enteric CH_4 production (7.4 to 63.2%) reflecting the uncertainty of such estimates. Furthermore, the observed variation among models indicated that provincial and national enteric CH_4 inventories are

influenced by the type of model used and therefore may impact the accuracy of inventory estimates and contribute to uncertainty. In Canada, emission from the agricultural sector has the highest uncertainty where enteric CH_4 contributes approximately 35.5% of the total agricultural GHG emissions in 2008 (Environment Canada 2010). Given the complex nature of the models and their need for detailed dietary information and other inputs, application of mechanistic models in preparation of national inventory estimates may be challenging. However, they can be used to generate Y_m values that can be used in national inventory models such as IPCC Tier-2 (USEPA 2010; NEAA 2010).

5.5.3. Model Associated Uncertainties

In general, some of the uncertainties in the current study are related to limitation in the cattle population database, management strategies utilized, animal performance data, dietary information and estimation of DMI. Currently, reports from national agencies combine beef and dairy cattle into a single category for both calves and breeding bulls. In addition, the agricultural census record does not differentiate between different management strategies (*i.e.*, feedlot vs. pasture-based operations). The information on animal performance and management strategies used in this study were from personal communication and previous survey reports (Boadi et al. 2004a). There is paucity of published data on provincial production practices, dietary information, DMI and animal performance, which contributes to the uncertainty of model prediction.

Karimi-Zindashty et al. (2012) conducted a sensitivity analysis using data from Canadian beef cattle to analyze the uncertainties in implementing the IPCC Tier 2 methodology for enteric CH_4 inventory and reported that IPCC default parameters such as, Y_m, EF and the coefficients used to calculate net energy for maintenance are the greatest sources of uncertainty. Furthermore,

the use of fixed proportions for the conversion of GEI to CH_4, regardless of DMI, by models such as IPCC Tier-2 to estimate CH_4 production also creates additional uncertainty. It has been demonstrated that as intake increases the percentage of GE lost as CH_4 declines due to increased passage rate of particulate matter (Okine et al. 1989). Moreover, various types of carbohydrates that constitute the bulk of GE (Mills et al. 2001), as well as feed additives such as fat (Beauchemin and McGinn 2006) affect CH_4 production non-linearly. The performance of a given model depends on the accuracy of the input parameters provided to the models. For the mechanistic models, the detailed input of dietary information contributes to their more precise prediction of CH_4 emissions as compared to empirical models (Benchaar et al. 1998; Kebreab et al. 2006a; 2008), but the lack of detailed dietary information can also contribute to uncertainty in these models.

Enteric CH_4 emissions from rumen fermentation are affected by the temperature within which the animals are managed (Bernier et al. 2012). Several studies (Takahashi et al. 2002; Bernier et al. 2012) reviled that enteric CH_4 production is reduced for animals managed under cold temperature. However, the models used in the current study do not account for cold acclimatization experienced by Manitoba beef cattle. Thus, using these models for animals managed under cold environment could overestimate CH_4 production, contributing to the uncertainty of estimates.

5.6. CONCLUSIONS

The outcome from mechanistic and empirical models used in the current study showed a similar CH_4 production pattern over the period of study. All models showed an increasing trend in CH_4 production between 1990 and 1996 and 2003 to 2005 and a decreasing trend between 2005 and 2008. This study indicates that trends in enteric CH_4 production are heavily influenced by trends

in the beef cattle population. Large variation was observed among models in estimating Y_m, EF and total enteric CH_4 production from Manitoba beef cattle. The estimated absolute CH_4 production varied from 7.4% to 63.2% among models. The average Y_m value estimated using MOLLY for cow-calf production systems showed less deviation than the other three models as compared to average values reported by Canadian research studies. To date, provincial and national inventory values of enteric CH_4 emission are obtained using the IPCC Tier-2 methodology. The observed variation among models in the current study illustrates the uncertainty in estimating enteric CH_4 production. There is a need to select appropriate models for estimation of provincial and national enteric CH_4 inventories which minimize uncertainty. The complex nature of mechanistic models might hamper their application in national CH_4 inventory assessments. Thus, a more robust approach could be integration of different models by using mechanistic models to estimate regional Y_m values which in turn are used as inputs for IPCC models, a strategy currently implemented by U.S. and the Netherlands.

6. MANUSCRIPT III

Estimation of greenhouse gas emissions from beef production systems in the Canadian prairies using whole-farm models

Aklilu W. Alemu[1], K. H. Ominski[1] M. Tenuta[2] and E. Kebreab[3]

[1]Department of Animal Science, University of Manitoba, Winnipeg MB, Canada; R3T 2N2,

[2]Department of Soil Science, University of Manitoba, Winnipeg MB, Canada; R3T 2N2,

[3]Department of Animal Science, University of California, Davis, CA, 95616, USA

In preparation to be submitted to Agricultural Systems.

6.1. ABSTRACT

Measuring greenhouse gas (GHG) emissions from beef cattle production systems is challenging as these are complex systems, composed of multiple interacting components. Thus, a whole-farm simulation study was conducted to estimate total farm GHG emissions from a cow-calf beef production system in the Canadian prairies. The objectives of this study were to: (i) integrate process-based farm component models into a whole-farm GHG model to estimate farm GHG intensity (ii) compare the estimates from the integrated model with estimates of other extant whole-farm models (Integrated Whole Farm System Model (IFSM) and Holos model) and (iii) to compare farm GHG intensity associated with changes in management strategy (*i.e.*, amount and time of liquid hog manure application on grassland). An Integrated Components Model (ICM) that integrated process-based models, COWPOLL and manure-DNDC, as well as aspects of IPCC was used to analyze whole-farm GHG emissions intensity of a cow-calf production system in the Canadian prairie. The management scenarios examined were simulated based on available data from a 3-year field study and included the following: baseline scenario (no application of liquid hog manure on grassland), split scenario (application of half of the required liquid hog manure on grassland in fall and the rest in spring) and single scenario (application of all the required liquid hog manure on grassland in spring). Model estimates considered on-farm emissions of methane (CH_4), nitrous oxide (N_2O) and carbon dioxide (CO_2) as well as pre-chain emissions from manufacturing and transport of farm inputs and indirect emissions of N_2O associated with leaching and volatilization loss of nitrogen applied on the farm. Emissions related to storage of imported liquid hog manure were not included. It was assumed that the simulated beef operation was self-sufficient with regard to feed and bedding and the long term soil carbon balance for the cropland producing feeds was in steady state. The whole-farm GHG

analysis estimated emissions of 31, 27 and 28 kg CO_2 equivalent (eq) kg^{-1} carcass weight for the baseline scenario using Holos, ICM and IFSM, respectively. However, emission intensity estimates were higher for split scenario (46 and 42 kg CO_2 eq kg^{-1} carcass weight) and single scenario (46 and 44 kg CO_2 eq kg^{-1} carcass weight) using ICM and IFSM, respectively. Average model estimates indicated that enteric CH_4 was the major contributor in the baseline scenario (63-76%) whereas enteric CH_4 (43-51%) and soil N_2O (37- 45%) were the major contributors for the split and single scenarios. In general, the higher contribution of soil N_2O emissions in the split and single scenarios following liquid hog manure application increased the farm GHG intensity. Furthermore, variation was observed among models, not only in estimating total farm GHG emissions, but also in estimating the relative contribution of the different GHG sources in the production system.

Key words: greenhouse gas emission, whole-farm model, beef production system.

6.2. INTRODUCTION

The global demand for animal-sourced foods is expected to double by the first half of the century as a result of growing human population and improved standard of living (Thornton 2010; O'Mara 2011). To meet the increasing associated demand for food, the livestock sector will compete for resources, such as land, water and energy which may have a profound implication on the environment (Alexandratos 2009; Herrero et al. 2009). The agricultural sector worldwide is one of the major contributors to greenhouse gas (GHG) emissions (Smith et al. 2007). Animal agriculture contributes about 8-11% of global anthropogenic emissions (O'Mara 2011), which could be higher (18%) if emissions from land-use changes including deforestation caused by expansion of pastures and arable land for feed-crops are included (Steinfeld et al. 2006; Gill et al. 2010). In 2009, Canadian agriculture contributed about 8% of total national GHG emissions, an increase of 20% since 1990. The livestock sector contributed about 61% of the agricultural emission with 74% attributed to beef cattle (Environment Canada 2011). The major gasses responsible for emissions from the agricultural sector include methane (CH_4) from enteric fermentation and manure management, nitrous oxide (N_2O) from fertilizer application during cultivation and carbon dioxide (CO_2) from fossil-fuel combustion and deforestation (Steinfeld et al. 2006). The expansion of beef cattle and swine numbers and the increased use of synthetic N fertilizer in the Canadian prairies are the major driving forces associated with the increasing emission trend in Canadian agriculture. Over the last few years, the environmental impact of livestock production has received increasing attention and therefore, quantification and mitigation of GHG emissions from the sector has been a focal point in agricultural research.

Measuring GHG emissions from livestock production system is challenging as these are complex systems, composed of multiple interacting components (*e.g.*, animals, soil, and crops).

To date, research has been focused on identifying and quantifying emissions from the individual sources within the production system, and understanding the influencing factors and developing mitigation measures (*e.g.*, Boadi et al. 2004b; Amon et al. 2006; Berg et al. 2006; Ominski et al. 2006). Similarly, mathematical models have been developed, based on the information from the individual farm components, to estimate GHG emissions and simulate mitigation strategies (Dijkstra et al. 1992; Li et al. 1992a; Baldwin 1995; Kebreab et al. 2002, 2004) Given the interrelationships among the farm components, studying GHG sources from individual components separately fails to consider their interactions and impacts on the total farm GHG emissions (Janzen et al. 2006; Shills et al. 2007). Conversely, a whole-farm approach of analyzing GHG emissions from a given production systems, using whole-farm models, is able to fully capture the interrelationships among the different farm components (Shills et al. 2007; Rotz et al. 2010; Crosson et al. 2011), predict the effects of changes in management practices on net farm GHG emissions (Schils et al. 2005; Beauchemin et al. 2011), and identify cost-effective whole-farm GHG mitigation strategies (Rotz et al. 2011b). Effective mitigation strategies can only be developed using whole-farm approaches to analyze farm GHG emissions (Oenema et al. 2001).

It has been suggested that development of process-based, whole-farm models can be developed by integrating existing individual farm component models (Sherlock and Bright 1999; Wastney et al. 2002; Peters et al. 2010). White et al. (2010) integrated a feed formulation program, a nutrient budgeting/GHG emissions model and a whole-farm management models to investigate the environmental and economic implications of New Zealand`s beef cattle intensification. Process-based whole-farm models are more flexible in incorporating detailed information and different management practices compared to whole-farm models that are based

on the Intergovernmental Panel for Climate Change (IPCC) methodology (*e.g.,* Johnson et al. 2002, 2003; Casey and Holden 2006a, b; Ogino et al. 2007; Little et al. 2008; Stewart et al. 2009; Nguyen et al. 2010; Foley et al. 2011; Cederberg et al. 2011). The IPCC methodology uses constant emission factors (EFs) to estimate GHG emissions. However, using constant EFs has proven insufficient to estimate GHG emissions because emissions in a production system are governed by many factors which differ from one system to another. Therefore, the objectives of this study were: (i) to estimate whole-farm GHG emissions intensity, by integrating, the existing farm component models, from a cow-calf production system in the Canadian prairies (Manitoba); (ii) to compare estimates of GHG emissions from the integrated components model with estimates using other whole-farm models, and (iii) to compare farm GHG emissions intensity estimates associated with changes in management strategy (*i.e.*, amount and time of hog slurry application).

6.3. MATERIALS AND METHODS

6.3.1. Description of Models Used

An Integrated Components Model (ICM) and two extant whole-farm models, Integrated Farm System Model (IFSM, Rotz et al. 2011a) and Holos model (Little et al. 2008), were used to estimate total farm GHG emissions from a cow-calf production system in the Canadian prairies.

6.3.1.1. Integrated Components Model (ICM)

The ICM integrated components of COWPOLL (Dijkstra et al. 1992; Mills et al. 2001), manure-DNDC (Li et al. 2012) and some aspects of IPCC (IPCC 2006). The ICM serves to estimate CH_4, N_2O and CO_2 emissions from the production system as well as emissions related to production and transport of inputs into the system (pre-chain emissions, Table 6.1).

Enteric CH_4 in ICM was estimated using COWPOLL (Dijkstra et al. 1992; Mills et al.

2001) based on predictions from rumen methanogenesis and hind gut fermentation as described by Mills et al. (2001). COWPOLL is a dynamic and mechanistic model designed to simulate the digestion, absorption and outflow of nutrients in the rumen (Dijkstra et al. 1992). Enteric CH_4 production is estimated based on VFA stoichiometry developed by Bannink et al. (2006) which relates the VFA produced to the type of substrate fermented in the rumen. The assumption is that the H_2 produced in the rumen from fermentation of soluble carbohydrate and protein is used: i) to support rumen microbial growth, ii) for biohydrogenation of unsaturated fatty acids and iii) for production of glucogenic VFA (i.e. propionate and valerate). The remaining H_2 is used for reduction of CO_2 to CH_4 (Baldwin 1995). The model has been used to estimate enteric CH_4 emissions from dairy cattle (Kebreab et al. 2008; Bannink et al. 2011) and cow-calf production in western Canada (Legesse et al. 2011; Alemu et al. 2011). The model was selected based on its performance on estimating enteric CH_4 emissions from Manitoba beef cattle (Chapter 5). Model simulation was performed using acslX software (Version 2.4.2.1; The Aegis Technologies Group 2008). Diet for a representative animal for each category was formulated using Cowbytes[@], beef cattle ration balancer (Version.4.6.8, Alberta Agriculture, Food and Rural Development 2003).

To estimate faecal and urinary composition (organic matter (OM), carbon (C) and nitrogen (N)), the COWPOLL model was extended using static equations that described intestinal and hind gut digestion (Reijs 2007). Potentially degradable fiber, starch and protein not degraded in the rumen were assumed to be subjected to hindgut digestion. Volatile fatty acid production and microbial growth in the hindgut are dependent on the amount of OM fermented and were estimated using a fixed microbial efficiency (Reijs 2007) and VFA coefficients for roughage diets (Bannink et al. 2006). Faecal and urine OM, C and N were categorized into different

components (Reijs et al. 2007). Excreted faecal OM was categorized into four major components: endogenous (endogenous protein and lipid), feed fiber (undegradable and degradable NDF, undegradable protein), microbial (rumen and large intestine microbial biomass) and faecal other feed components (*e.g.*, lipids). Urinary N balance was calculated by deducting the amount of N in faeces, milk (for lactating cows) and retained in the body (for growing animals) from the total N intake of the animal. Nitrogen retention was assumed to be zero for mature (*i.e.*, non-growing) animals, a function of average daily gain (ADG) and retained energy for growing animals and a function of milk yield and milk protein content (350 g kg^{-1}) for lactating animals (NRC 2000). Urinary components were fractioned into urea-like component such as urea, uric acid and allantoin and non-urea like components such as amino acids, creatine, hippuric acid as well as xanthine and hypoxanthine. Specific values were used to estimate the C and N composition of the aforementioned components excreted in faeces and urine (Reijs et al. 2007).

The ICM estimated GHG emissions from cropland using a process-based model, manure-DNDC (Version 2.0; Li et al. 2012). The manure-DNDC model is a modification of the existing Denitrification-Decomposition (DNDC) model (Li et al. 1992a, b) which links the core of DNDC to a virtual animal farm (Li et al. 2012). The model is similar to the original DNDC model in that it shares the parameterized biogeochemical processes (*i.e.*, decomposition, urea hydrolysis, ammonium-ammonia equilibrium, NH_3 volatilization, nitrification, denitrification, nitrate leaching and fermentation). The biogeochemical processes that are developed in the DNDC model to describe the soil organic matter are fully adapted in manure-DNDC to describe the manure OM turnover, assuming OM in soil and manure is similar (Li et al. 2012). The model predicts transport and transformation of C and N in soil profiles driven by a series of

biological and geochemical processes (Li et al. 1992a, b; 1994; Li 2007).

Manure-DNDC contains four interacting submodels: (i) a thermal hydraulic submodel that uses soil physical property, air temperature, and precipitation data to calculate soil temperature and moisture profile and soil water fluxes through time; (ii) a denitrification submodel that calculates hourly denitrification rate and N_2O and dinitrogen production during periods when the soil has greater than 40% water-filled pore space; (iii) a decomposition submodel that calculates daily decomposition, nitrification, NH_3 volatilization process and CO_2 production (soil microbial respiration) and (iv) a plant growth submodel that calculates N uptake by plants, plant growth and daily root respiration (Li et al. 1992a, b; 1994). Crop growth is simulated using a crop growth curve (Li et al. 1994). Crop N uptake from the soil is the key process in the model that links the climate and soil status, and is affected by the crop potential maximum yield, the crop C:N ratio, the crop growth curve and the availability of dissolved inorganic N (NO_3^- and NH_4^+) (Li et al. 1994).

The input information required for manure-DNDC includes climate data (*i.e.*, site location, daily temperature and precipitation), soil characteristics (*i.e.*, texture, pH, bulk density, clay fraction, wilting point, soil organic C, initial NO_3^- and NH_4^+ concentration), crop parameters (*i.e.*, grain production, grain, leaf, stem and root fraction, and their C:N ratio), and management practices (*i.e.*, tillage, fertilization, manure amendment, irrigation, grazing). The model is simulated on a daily time steps and annual GHG fluxes of N and C are calculated as the sum of daily estimates. Manure-DNDC does not calculate indirect N_2O emissions from N loss due to volatilization and leaching, therefore, the IPCC default EF for volatilization and leaching were adopted to calculate indirect N_2O emission assuming that N lost as NH_3 is deposited on land or water (Asman, 1998; Table 6.1).

Emissions of CH_4 and N_2O from manure management in ICM were estimated using IPCC Tier 2 methodology (IPCC 2006), as a consequence of the limitations associated with manure-DNDC. Firstly, the virtual farm constructed in manure-DNDC model contains a limited number of beef cattle categories (beef cow and veal) and further, the model can only accommodate one animal category per simulation. The typical cow-calf operation in western Canada, however, is comprised of beef cows, bulls, replacement heifers, backgrounded animals and calves (Basarab et al. 2005; Alemu et al. 2011), all of which contribute to total on-farm manure production. Secondly, manure management methods incorporated in the model are limited to compost, lagoon and anaerobic digester which differ from the deep bedding manure management system utilized in the majority of western Canadian beef operations (Martin et al. 2004). Methane emissions from manure storage were a function of volatile solid and CH_4 conversion factor whereas CH_4 emissions from fresh faeces deposited on pasture during grazing were calculated using a constant emission factor (Yamulki et al. 1999; Table 6.1). The excreted urinary urea-like components estimated by COWPOLL model were assumed to be converted in to NH_4 during manure storage and used to estimate NH_4 concentration of the on-farm produced solid manure.

Calculation of CO_2 emissions from on-farm energy used in ICM includes fossil fuels and power use for seeding, harvesting, manure spreading and transport of supplemental feed to the farm. Emissions related to the manufacture of machinery and fuel were not considered. Emissions of CO_2 from direct on-farm fuel use were calculated as the product of the cropland size and a unique energy use value associated with each crop type (Table 6.1). The long term soil carbon balance for the cropland producing feeds is assumed to be zero (steady state).

Table 6.1 Models, equations and/or emission factors (EF) used in the Integrated Components Model (ICM), Integrated Farm System Model (IFSM) and Holos model to estimate greenhouse gas emissions from a cow-calf production system

	ICM		IFSM		Holos	
Greenhouse gases	Model/ Equations/ EF	Ref[z]	Model/ Equation/ EF	Ref[z]	Model/ Equations/ EF	Ref[z]
Methane (CH_4)						
Enteric fermentation	Based on rumen fermentation and rumen VFA stoichiometric models	21, 9	Nonlinear Mitscherlich (Mits 3) equation	1	Based on GE requirement, DE of feed and Y_m factor	4
Deep bedding manure[y]	Based on volatile solid and MCF	4	Based on volatile solid and MCF	2	Based on volatile solid and MCF	4
Field applied manure	na		Linear Equation based on manure VFA concentration[x]	2	na	
Fresh faeces on pasture	EF = 0.086 g CH_4 kg^{-1} faeces	2	EF = 0.086 g CH_4 kg^{-1} faeces	2	na	
Soil emission/uptake	Fermentation submodel of Manure-DNDC model	11	na		Assumed to be negligible	14
Direct N_2O-N						
Deep bedding manure	EF = 0.01 kg N kg^{-1} N	4	EF = 0.02 kg N kg^{-1} excreted N (floor), EF = 0.005 kg N kg^{-1} excreted N (stacked manure)	4	EF = 0.01 kg N kg^{-1} N	4
Soil/cropping nitrogen Field applied manure Pasture deposited dung & urine	Denitrification and nitrification submodel of Manure-DNDC	11	Simplified DAYCENT model	5	EF_{eco} = 0.022 P/PE –0.0048	18
Indirect N_2O-N						
Deep bedding manure						
Leaching	EF = 0.0075 kg N $Frac_{leach}$ = 0 kg N	4	na		EF = 0.0075 kg N $Frac_{leach}$ = 0 kg N	4
Volatilization	EF = 0.01 kg N Manure NH_4-N concentration was estimated using COWPOLL model[w]	4	Diffusion, dissociation, aqueous to gas partitioning and mass transport equations	2	EF = 0.01 kg N $Frac_{vol}$ = 0.30 kg N	4
Soil/cropping nitrogen						
Field-applied manure						
Pasture deposited dung & urine						
Leaching	Hydrological model	12	NLEAP model	6	EF = 0.0075 kg N	4, 18

					$Frac_{leach}$ = 0.3247 P/PE – 0.0247	
Volatilization	Decomposition submodel of Manure-DNDC	11, 4	NLEAP model	6, 4	EF = 0.01 kg N $Frac_{vol}$ = 0.1 kg N	4
Carbon dioxide (CO_2)						
On-farm energy use and cropping[v]	Unique energy use coefficients for different crops		EF = 2.637 kg CO_2 L^{-1} diesel fuel used; GREET model	7	F4E2 model to estimate energy coefficients	19
Manure application	EF_{liquid} = 0.42 kg CO_2 kg^{-1} N; EF_{solid} = 0.27 kg CO_2 kg^{-1} N	13	Fuel use factor was used to estimate fuel consumption	2	0.0248 GJ (1000 L manure)$^{-1}$, 75 Kg CO_2 GJ^{-1}	14
Barley production	EF = 107 kg CO_2 ha^{-1}	14	Fuel use factor was used to estimate fuel consumption	2	E_{fuel} x crop area (ha) x 75 kg CO_2 GJ^{-1}	14
Hay production	EF = 60.8 kg CO_2 ha^{-1}	14	Fuel use factor was used to estimate fuel consumption	3	E_{fuel} x Hay area (ha) x 75 kg CO_2 GJ^{-1}	14
Soybean meal transportation	EF = 0.0016 kg CO_2 kg^{-1} soybean meal	20	na		na	
Electricity	0.22 kg CO_2 kWh^{-1} , 47.1 kWh beef cow^{-1} yr^{-1}	15, 16	EF = 0.73 kg CO_2 kWh^{-1}, derived from GREET model	7, 8	65.7 kWh $Cattle^{-1}$ yr^{-1}, 0.22 kg CO_2 kWh^{-1}	14
Energy to dry barley grain	0.001 kg CO_2 kg^{-1} grain	17	Electric use factor is used to estimate electric consumption	2	na	
Herbicide production[u]	EF = 2.67 kg of CO_2 ha^{-1}	14	EF = 22 kg of CO_2 kg^{-1} of pesticide active ingredient	2	E_{herbic} x area (ha) x 5.8 kg CO_2 GJ^{-1}	14
Nitrogen fertilizer production	EF = 3.59 kg CO_2/kg N	22	3.307 kg CO_2/kg N	2	3.59 kg CO_2/kg N	14

DE = Digestible energy, E_{fuel} = Energy from fuel use (GJ ha^{-1}), E_{herbic} = Energy for herbicide production (GJ ha^{-1}), EF = Emission factor, EF_{eco} = Emission factor for ecodistrict, EF_{liquid} = Emission factor for liquid manure application, EF_{solid} = Emission factor for solid manure application, F4E2 = Farm Fieldwork and Fossil Fuel Energy and Emissions model, $Frac_{leach}$ = leaching fraction, $Frac_{vol}$ = volatilization fraction, MCF = Methane conversion factor (by handling system), na = not applicable, NLEAP = Nitrate Leaching and Economical Analysis Package model, P = Growing season (May to October) precipitation, PE = Growing season (May to October) evapotranspiration, VFA = Volatile fatty acid, Y_m = Methane conversion factor (by diet).

[z]1 = Mills et al. (2003), 2= Rotz et al. (2011a), 3 = Rotz et al. (2010), 4 = IPCC (2006), 5 = Chianese et al. (2009), 6 = Shaffer et al. (1991), 7 = Wang (2007), 8 = Ludington and Johnson (2003), 9 = Bannink et al. (2006), 10 = Yamulki et al. (1999), 11 = Li et al.(2012), 12 = Li et al. (2006), 13 = Wiens et al. (2008), 14 = Little et al. (2008), 15 = Dyer and Desjardins (2006), 16 = Environment Canada (2011), 17 = Vergé et al. (2007), 18 = Rochette et al. (2008a), 19 = Dyer and Desjardins (2003), 20 = Meough et al. (2012), 21 = Mills et al. (2001), 22 = Nagy (2000).

[y]Methane emission from deep bedding manure management was estimated using IPCC (2006). For IFSM, MCF = $7.11e^{0.0884(T)}$, where T = ambient barn temperature (^{0}C) and methane producing capacity (B_o) = 0.24 m^3 CH_4 (kg volatile solid)$^{-1}$. For ICM and Holos MCF = 0.17 kg CH_4 (kg CH_4)$^{-1}$ and B_o = 0.19 m^3 CH_4 (kg volatile soild)$^{-1}$

[x]CH_4 (kg d^{-1}) = (0.17 x VFA + 0.026) x land area (ha) x 0.032; where VFA is the daily concentration of VFAs in the manure (mmol (kg manure)$^{-1}$) which is a function of initial concentration and time.

[w]Urinary urea-like compounds including urea, uric acid and allantoin estimated using COWPOLL model was assumed to be converted to NH_4-N during manure storage.

[v]For IFSM, fuel consumption is estimated by using fuel use factor (average amount of fuel used to produce and deliver a unit of feed to the herd or remove a unit of manure) and total farm fuel use is calculated by summing the fuel use over all operations. Engine CO_2 emissions = fuel use (L h^{-1}) x 2.637. Fuel use is a function of fuel consumption rate (L kWh^{-1}), engine power (kW), fuel use efficiency, engine load and fuel use index (Rotz et al., 2011a).

[u]Application rate of 2 kg h^{-1} for grain crops (Rotz et al., 2011a).

6.3.1.2. Integrated Farm Systems Model (IFSM, Version 3.4)

The IFSM is a farm simulation model that predicts the long term performance, environmental impact and economic viability of dairy, beef and crop farms over multiple years of weather (Rotz et al. 2011a). Initial conditions are reset each year as the model does not consider inter-year dynamics. The IFSM contains nine major sub models that represent the following major processes: crop and soil, grazing, machinery, tillage and planting, crop harvest, crop storage, herd and feeding, manure management, and economic analysis. Nutrient flows through the farm are modeled to predict nutrient accumulation in the soil and loss to the environment. Whole-farm mass balance of N, C, potassium and phosphorus are determined as the sum of all imports in feed, fertilizer, deposition, and crop fixation minus the export in milk, excess feed, animals, manure, and losses leaving the farm. The model predicts the growth and development of grass, alfalfa, corn, soybean, and small grain crops based up on daily soil and water conditions. The IFSM has been used to quantify the C footprint of beef (Rotz et al. 2005) and dairy (Rotz et al. 2010; Sejian et al. 2011) production systems and to evaluate various management practices and their impact on net farm GHG emission (Rotz 2004; Rotz et al. 2011b; Belflower et al. 2012).

In a given year, IFSM performs a sequence of simulations in a daily time step that begin with manure handling, tillage, planting operation, growth and harvest operation, feed storage, feed utilization and herd production (Rotz et al. 2011a). The beef herd in the model is described using six categories: cows (mix of primiparous and multiparous females), nursing calves, young heifers (weaning to 1 yr), yearling replacement heifers, backgrounded/stocker cattle, and finishing cattle. At weaning, calves are categorized as backgrounder and/or young replacement heifers. Animal feed intake, performance, and manure production are modeled using the herd and feeding components of the model. The diet for each group is formulated using linear

programming to meet requirements for roughage, energy, minimum RDP and minimum RUP (Rotz et al. 2011a). Energy, protein and mineral requirements are determined using the Cornell Net Carbohydrate and Protein System (CNCPS; Rotz et al. 2005).

The IFSM estimates CH_4, N_2O and CO_2 emissions from different sources in the production system as well as emissions related to production and transport of resources used on the farm (pre-chain emissions, Table 6.1). Enteric CH_4 production is estimated using the nonlinear Mitscherlich 3 (Mits3) equation developed by Mills et al. (2003). The model calculates CH_4 emissions from manure storage using IPCC Tier 2 methodology where emissions are a function of the total volatile solid and the CH_4 conversion factors. For field applied manure, CH_4 emissions are a factor of manure VFA content which are assumed to decline exponentially after application (Sherlock et al. 2002). A constant EF is applied for CH_4 emissions from fresh faeces deposited on pasture (Rotz et al. 2011a).

Nitrous oxide emissions from crop land are estimated in the IFSM using a simplified, process-based DAYCENT model (Chianese et al. 2009). For N_2O emissions from manure storage, the model applies a constant EF value (Table 6.1). The IFSM estimates NH_3 volatilization from animal housing, manure storage, field applied manure, faecal and urine N deposited on pasture (Rotz et al. 2011a). Total farm NH_3 emissions are the sum of emissions from each source. Indirect N_2O emissions from N lost through volatilization and leaching are calculated using the default IPCC EF for volatilization and leaching (IPCC, 2006).

The IFSM estimates CO_2 emission from primary (*i.e.*, net feed production, on-farm energy use) and secondary sources (production of fuel, electricity, machinery, fertilizer, pesticide, plastic and manure handling) (Chianese et al. 2009; Rotz et al. 2011a). On-farm energy use is determined for individual operations based on the equipment used and the power required to

perform the operations.

6.3.1.3. Holos model (Version 1.1.2)

The Holos model is an empirical whole-farm model developed by Agriculture and Agri-Food Canada (Little et al. 2008). The model is based on IPCC (2006) methodology modified for Canadian conditions and cattle production practices. The model has been used for life cycle analysis (LCA) of GHG emissions from Canadian beef (Beauchemin et al. 2010) and dairy (McGeough et al. 2012) production and to simulate different enteric CH_4 mitigation strategies and their impacts on total farm GHG emissions (Beauchemin et al. 2011). The common Canadian farm management practices or '*scenarios*' are included in Holos model where each farm component or agricultural operation contains at least one scenario (Little et al. 2008). There are seven cow-calf scenarios that differ in season of calving (spring or fall), feeding strategy (year-round grazing or winter feeding), and the management of calves (sold or backgrounded on farm). The "cattle fed over winter, spring calving, calves backgrounded on farm after weaning" scenario was used in the current study. The Holos model was used to estimate emissions for only one of the management practices tested in the current study as the model does not have the capacity to import animal manure into the production system.

The Holos model considers the following emission sources: (i) on-farm emissions of CH_4, N_2O and CO_2, (ii) emissions associated with manufacturing of farm inputs (herbicide), and (iii) off-farm (indirect) emissions of N_2O derived from leaching and volatilization loss of N applied on the farm (Table 6.1). Methane emissions were calculated from enteric fermentation and manure management. However, considering the offset of CH_4 emissions from wet areas of the soil by amount of CH_4 oxidation in dry, aerobic areas of the soil, net CH_4 emissions from soils in the model were assumed to be negligible (Little et al. 2008). Enteric CH_4 emissions were

estimated using the IPCC Tier 2 approach whereby emissions are calculated using a percent of GE intake (GEI) with a diet specific CH_4 conversion factor (Y_m, % GEI). The model estimates manure CH_4 emissions based on total volatile solid produced (IPCC 2006) that accounts for the GE intake of the animal and digestibility of the diet. The volatile solid produced is multiplied by maximum CH_4 producing capacity of manure and CH_4 conversion factor specific to the manure management practice used (Table 6.1).

Direct N_2O estimated from soil is based on total N inputs (*i.e.,* from synthetic N fertilizer, land-applied manure, crop residue decomposition and net mineralization). Nitrogen mineralization is estimated from net change in soil C and assumed to be zero in the current study. Therefore, N mineralization was assumed be at steady-state and mineralized-N was not added to the sum of crop N-inputs. Total N input was multiplied by an EF, adjusted for growing season precipitation and potential evapotranspiration, soil texture, tillage and topography (Rochette et al. 2008a). Furthermore, direct N_2O emissions from manure storage (deep bedding) are calculated from manure N content and an EF for the manure handling system (Table 6.1). The model also estimates indirect N_2O emissions from N leaching, runoff, and volatilization based on the IPCC (2006) EF and the assumed fraction of N lost from N input.

The Holos model calculates CO_2 emissions using primary and secondary on-farm energy sources. Primary sources include the use of fossil fuel and power for seeding, harvesting, feeding animals, manure spreading. Secondary sources include the emissions related to manufacturing of herbicide (Little et al. 2008, Table 6.1). Emissions of CO_2 associated with manufacture of machinery and transport of goods are not part of the model.

6.3.2. Organization of Model Inputs

Model input values were obtained from a 3-year field experiment conducted on a 32-ha forage

field located near the town of La Broquerie, Manitoba (49°31'N, 96°30'W, Ecodistrict 849) as well as from the literature.

6.3.2.1. Description of experimental site and climate

A detailed description of the experimental site has been given by Tenuta et al. (2010) and Wilson et al. (2010, 2011). The study investigated the time of application of liquid hog manure on quantity and quality of forage, animal performance (Wilson et al. 2010), GHG emissions, including N_2O, CH_4 and CO_2 (Tenuta et al. 2010; Wilson et al. 2010; Tremorin et al. 2012), and nutrient cycling (Wilson et al. 2011) from hayed and grazed paddocks. A detailed description of botanical composition at the experimental site has been described by Wilson et al. (2010).

Climate data for the experimental site is provided in Table 6.2. The long term normal for precipitation obtained from the Steinbach Municipality Airport, located 12 km from the study site, is 541 mm annually and 408 mm in the growing season (May to October). On site growing season precipitation during 2004, 2005 and 2006 was 584, 553 and 271 mm, respectively.

The soil series present at the experimental site was Berlo loamy fine sand (70%) and Kergwenan loamy sand to gravel (30%) with the former being a Gleyed Dark Gray Chernozem (FAO Greyzem) according to the Canadian System of Soil Classification (Hopkins, 1985). Detailed soil properties were measured before the start of the experiment (fall 2003) and throughout the trial in the fall of 2004, 2005 and 2006 prior to manure application (Tenuta et al. 2010; Wilson et al. 2011, Table 6.3).

The quality of imported liquid hog manure applied to the pasture is summarized in Table 6.3. Liquid hog manure was sourced from the primary cell of earthen manure storage at an adjacent commercial hog finishing operation. Manure was applied at rate based on plant available N content in the manure assuming that 25% of manure NH_3 and ammonium (NH_4^+)

were lost by volatilization and 25% of organic N was available for plant use in the year of application (Prairie Provinces' Committee on Livestock Development and Manure Management, 2006). Nitrogen was applied at the rate of 72 kg available N ha^{-1} in the spring and fall for the split application and 144 kg available N ha^{-1} in the spring for single application. Average total N applied was 240 and 252 kg ha^{-1} for split and single application, respectively.

6.3.3. Simulation of a Beef Cattle Production System

6.3.3.1. System boundary and scope

Whole-farm GHG emissions were assessed using a simulated cow-calf operation with a spring calving herd consisting of 150 cows, bulls, backgrounding steers and heifers, and calves from birth to weaning (Figure 6.1; Table 6.4). The study used an International Organization for Standardization (ISO) LCA (ISO 2006a, b) to compare cradle to farm-gate cumulative GHG emissions (Figure 6.1). A LCA quantifies the environmental impacts of a given product or process by accounting for all resources used in the process.

Primary (emissions produced on farm during the production process) and secondary (emissions produced during the manufacture or production of resources used on the farm) sources of GHG emissions were considered in the analysis to estimate the carbon footprint of the cow-calf operation. Primary sources of GHG emissions include the on-farm emissions of CH_4 from enteric fermentation and manure management, on-farm N_2O emissions from manure and soil (direct N_2O emissions), off-farm N_2O emissions from N leaching, run-off and volatilization (indirect N_2O emissions), and CO_2 emissions from on-farm energy use. On-farm energy use included diesel fuel for farm operations (*e.g.*, seeding, harvesting and manure spreading) and electricity for housing and crop processing.

Table 6.2. Mean monthly air temperature and total precipitation at the experimental site from 2004 to 2006 and long term normal (1971 to 2000).

Months	Mean temperature, ^{0}C				Total precipitation, mm			
	2004	2005	2006	Long term normal[z]	2004	2005	2006	Long term normal[z]
January	-21.3	-18.3	-6.2	-17.4	47	36	44	22
February	-11.2	-12.5	-14.2	-13	15	12	15	14
March	-5.4	-6.2	-3.2	-5.5	72	19	39	19
April	3.8	7.4	8.9	4.1	27	21	12	29
May	7.8	10.4	11.9	11.9	137	110	23	59
June	14.4	17.6	17.4	16.6	90	232	50	95
July	18.2	19.9	21.3	19.1	85	68	42	80
August	14.2	17.1	18.5	18.1	137	42	26	69
September	15.1	14.4	12.8	12.1	89	19	94	60
October	6.1	7.1	3.9	5.4	46	82	36	45
November	-0.2	-2.2	-4.3	-5.0	12	40	8	27
December	-13.2	-8.5	-9.0	-14.1	36	25	36	21
Annual mean/total	2.2	3.9	4.7	2.7	793	705	425	541

[z]Long term normal (1971 to 2000) were obtained from Environment Canada (2012a) for the Steinbach Airport, [online] Avilable: http://www.climate.weatheroffice.ec.gc.ca/climate_normals/stnselect_e.html. [2011 September 20].

Table 6.3. Characteristics of the climate, soil and manure applied to the pasture at the experimental site

Climate[z]	Items
Latitude	$49^0$31`N
Longitude	$96^0$30`W
Annual solar radiation, MJ m^{-2}	13.53
Nitrogen in precipitation, mg L^{-1}	0.87
Atmospheric CO_2 concentrations, mg L^{-1}	390
Soil characteristics (0-30 cm)[y]	
Soil type	Gleyed Dark Gray Chernozem
Soil texture	Loamy sand
pH	7.90
Bulk density, g cm^{-3}	1.38
Field capacity (WFPS)	0.6
Soil total organic carbon, g C kg^{-1}	10.2
Soil NO_3^- concentration, mg N kg^{-1}	1.6
Soil NH_4^+ concentration, mg N kg^{-1}	1.9
Olsen-P, mg kg^{-1}	5.5
K^+, mg kg^{-1}	45.0
Land topography	Nearly level (<2%)
Imported liquid hog manure[x]	
Manure type	Slurry hog manure
Application method	Surface spread without incorporation
Application, `000 L ha^{-1}	46.48
DM, %	6.04
pH	7.10
Total N, % DM	10.63
Carbon to N ratio	7.82
Nitrate-N, mg N L^{-1}	3.34
Ammonium-N, g N L^{-1}	3.40
Organic N, % total N	31.63
Total P, g L^{-1}	1.08
Total K, g L^{-1}	2.17

[z]An average value (2005 to 2010) was used to estimate solar radiation at the experimental site; Nitrogen in precipitation was obtained from Environment Canada (2012b). [Online] Avilable: http://www.ec.gc.ca/natchem/default.asp?lang=En&n=B385159B-1. [2011 September 11].

[y]Determined in 2003 before the start of the study, textural class was according to USDA classification. Detail soil characteristics information was obtained from Wilson et al. (2011) and Tenuta et al. (2010).

[x]Average values for liquid hog manure applied in spring and fall (2003 to 2006). Detail information on imported liquid hog manure was obtained from Wilson et al. (2010, 2011) and Tenuta et al. (2010).

Secondary sources of GHG emissions include CO_2 emissions related to production and transportation of inputs to the farm (*e.g*, machinery, pesticide, supplemental feeds, Figure 6.1). A number of potential GHG emission sources and sinks were excluded from the analysis such as GHG emissions created after the cattle had left the farm, emission related to energy use for building farm infrastructures (*e.g.*, roads, buildings) and GHG sinks associated with land use.

Feedstuffs for the animals were comprised of home-grown grass (grass hay and grazed pasture), home-grown cereal (barley), imported soybean meal and minerals. Bedding material was obtained from the home-grown cereal.

All gasses were expressed as CO_2 equivalents (eq) to account for the global warming potential of each gas compared to CO_2, where: CO_2 = 1, CH_4 = 25 and N_2O = 298 (IPCC 2007). The GHG intensity of the cow-calf production system (environmental cost, carbon foot print) was calculated as sum of all GHG emissions converted to CO_2 eq. units divided by: (i) kg of beef produced (*i.e.*, live and carcass weight), and (ii) hectare land area, with the land area based on the amount of forage and crop land required to support the nutritional need of the animals (Table 6.4).

6.3.3.2. Description of the cow-calf production system

The simulated production system consisted of a cow-calf and backgrounding beef operation, a cropping operation (barley) and a forage production. The production cycle began in late October when the animals were managed in confinement (Table 6.4). Weaned calves were retained and backgrounded until the end of March. The annual production cycle consisted of three major production periods as outlined in Table 6.4.

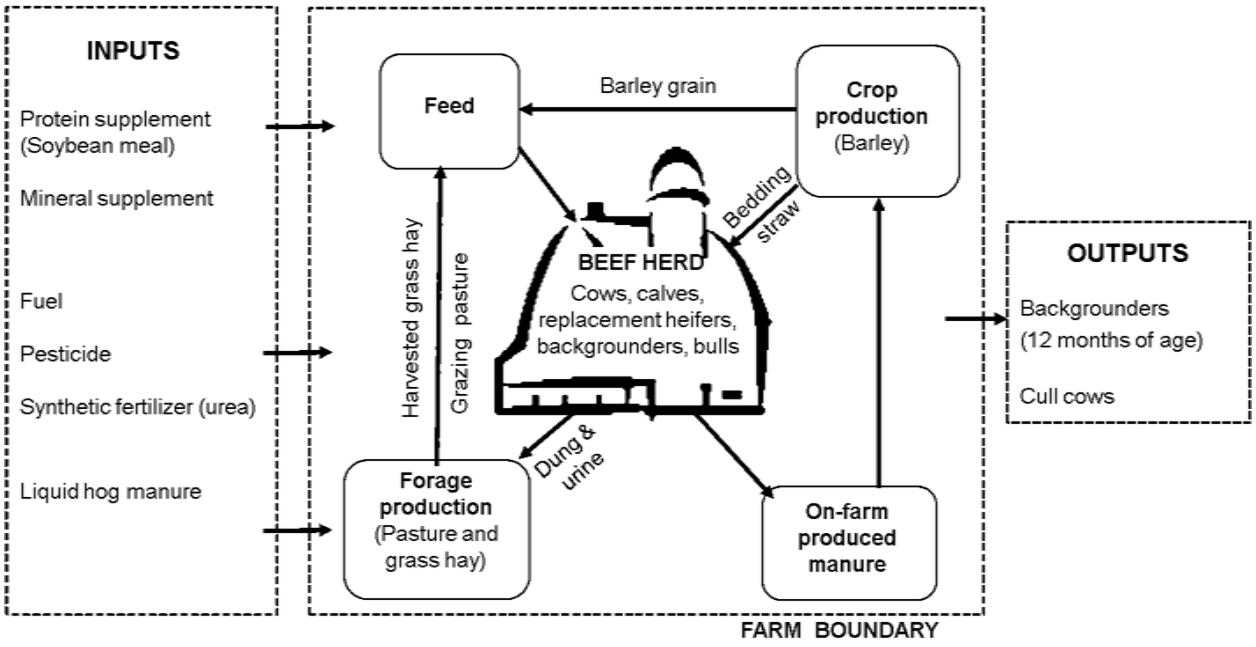

Figure 6.1. Schematic representation of the boundaries and processes of a cow-calf production system and the associated greenhouse gas emissions.

During the first period (1 November to 29 February) the operation consisted of 150 cows, 24 replacement heifers, 7 bulls and 104 backgrounded animals. Animals were confined (drylot) and feed rations formulated based on grass hay supplemented with barley grain and soybean meal. Beef cows were assumed to be in their late stage of pregnancy (3^{rd} trimester). At the beginning of period one, culled cows were replaced by replacement heifers from the previous year. Average cow culling and mortality rates were 15% and 1.25%, respectively (Waldner et al. 2009). Replacement heifers (ADG = 0.68 kg d^{-1}) were bred and calved at the age of 15 and 24 months, respectively (Alemu et al. 2011).

During the second period, 1 March to 30 April (61 days), animals were managed in confinement. Calves were born in late winter/early spring (March 15 was used as the average calving date) with an average body weight of 44 kg. The gender ratio of calves was assumed to be 1:1 (MacNeil et al., 1994). At the age of weaning (7 months, average weight = 209 kg), calves (128) were categorized as replacement heifers (24) and/or backgrounded animals (104). Backgrounded animals were fed a high forage diet (70 to 75% grass hay, ADG = 1.2 kg d^{-1}) and shipped to market at the end of March (average BW = 411 kg for steers and 394 kg for heifers). Average milk production for beef cows was 9.6 kg d^{-1} with an average fat and protein content of 31 g kg^{-1} and 40 g kg^{-1}, respectively (Kopp et al. 2004). During the third period, 1 May to 31 October (183 days.), beef animals (beef cows, suckling calves yearling replacement heifers and bulls) were grazed on a naturalized grass pasture.

Table 6.4 Animal categories, dry matter intake (DMI), average body weight and beef cattle population within the different periods of the simulated cow-calf production system.

	Period 1 (1 Novemebr–29 February) 121 days				Period 2 (1 March – 30 April) 61 days				Period 3 (1 May – 31 October) 183 days				
Animals	No	ADG, kg d^{-1}	Average DMI, kg	Average BW, kg	No	ADG, kg d^{-1}	Average DMI, kg	Average BW, kg	No	ADG, kg d^{-1}	Average DMI, kg	Average BW, kg	Diet[z]
Beef cow	150	0	11.6	588	150	0.10	13.3	617	150	0.11	13.7	651	Grass hay supplemented with barley grain and soybean meal (6 mo), pasture.
Suckling calves[y]	0	-	-	-	128	1.20	0.44	81	128	1.20	2.8	190	Grass hay and milk (1.5 mo), pasture (6 mo)
Replacement heifer calves	24	0.68	8.0	351	0	-	-	-	0	-	-	-	Grass hay supplemented with barley grain.
Yearling replacement heifers	0	-	-	-	24	0.68	9.8	372	24	0.68	10.9	434	Grass hay supplemented with barley grain (2 mo), pasture (6 mo).
Bulls	7	1.28	12.9	579	7	1.28	13.9	618	7	1.28	17.2	735	Grass hay supplemented with barley grain and soybean meal (6 mo), pasture (6 mo)
Backgrounding – Steers	64	1.20	10.4	383	64	1.20	10.5	411	0	-	-	-	High forage backgrounding diet[c].
Backgrounding – Heifers	40	1.20	9.2	366	40	1.20	10.2	394	0	-	-	-	High forage backgrounding diet[x].

ADG = Average daily gain, BW = Body weight, No = number of animals, mo = months.

[z]Cattle feed requirement and diets were formulated using CowBytes© software, a beef cattle ration balancer (Alberta Agriculture, Food and Rural Development (AAFRD, 2003).

[y]Average caving date = March 15.

[x]Approximately 75% grass hay, 15% barley grain and 10% soybean meal during period 1 and 2 as well as 90% pasture and 10% barley grain during period 3 for the control scenario; 80% grass hay and 20% barley grain during period 1 and 2 as well as 100% pasture during period 3 for the split and single scenario.

Total GHG emissions from the beef herd were calculated by summing the emissions for the cows, bulls, replacement heifers and backgrounding animals within the annual production cycle of the farm. Enteric CH_4 and manure CH_4 and N_2O were calculated as kg head^{-1} d^{-1} and multiplied by the number of animals for each cattle class and the number of days in each feeding period to provide an estimate of kg CH_4 and N_2O period^{-1}. Emissions were summed across feeding periods for each cattle class to give kg CH_4 and N_2O yr^{-1} or kg CH_4 and N_2O duration^{-1} for backgrounded steers and heifers where duration refers to the number of days from animal birth to departure from the farm. Total carcass weight was calculated from the final weight of all culled cows and backgrounders assuming 60% as a dressing percentage (Manitoba Agriculture, Food and Rural Initiative 2012).

Manure was managed in a deep bedding system. Barley straw was used as the bedding material during period 1 and 2 when animals where in confinement. The quantity of bedding required was estimated by calculating the number of animal units on the farm (Rotz et al. 2011a) considering the average mass of mature cow as an animal unit (1.81 kg DM (animal units)$^{-1}$ d^{-1}). The amount of excreted nutrient in faeces and urine was quantified using the extended COWPOLL model. Total manure DM included DM excreted by the animals plus that of bedding and feed loss (3% of forage DMI, Rotz et al. 2011a). Manure was applied as a N source to the barley field in spring and its composition was adjusted to account for the loss of N during manure management and field application. Manure was applied according to recommendations in Manitoba (Prairie Provinces` Committee on Livestock Development and Manure Management, 2006). During the third period (May to October), beef cattle were managed on pasture and manure was deposited and remained on pasture.

6.3.3.3. Description of cropping and land use

The required land size for production of feed to support the nutritional needs of the animals was calculated from the annual total feed required and the expected land productivity (Table 6.5). The daily DM intake of each group of animals in the farm, the number of days the feed was offered, and the proportion of feed in the diet DM was used to calculated the total DM required for each feed. Calculation of total DM required also included losses related to harvest, storage and feeding (Rotz et al. 2003) as well as trampling and wastage during grazing (Adams et al. 2009; Table 6.4). Land required for grass hay production was calculated based on the amount of DM required to feed the animals over 6-month period during confinement (period 1 and 2) and the DM losses during hay harvest and storage (15%, Rotz 2003). Similarly, for pasture land, a 25% allowance for trampling and wastage (Adams et al. 2009) was added to the quantity of DM required to feed the animals over the 6 month period (May to October). Average yield of pasture and grass hay was obtained from the 3-year field experiment (Table 6.5, Wilson et al. 2010). Barley was planted in mid-May and harvested in late August providing an average grain yield of 2.02 Mg DM ha^{-1} (Manitoba Management Plus Program 2012).

6.3.4. Simulated management scenarios

Beef cattle producers may apply cattle or hog manure to forage land to improve composition, quality and productivity (Blonski et al. 2004; Wilson et al. 2010; Bork and Blonski 2012). However, little is known about its environmental impact from a whole-farm perspective (Petersen et al. 2007).

Table 6.5. Total annual land required, expected yield, losses and nutrient composition of forage and barley grain used for the control, split and single management scenarios in the whole-farm analysis

Items[z]	Yield[y] (Mg DM ha^{-1})	DM[y] (%)	Crude protein[y] (g kg^{-1})	Herbicide used	Harvest and storage loss[x] (%)	Feed utilization loss[w] (%)	Land required[v] (ha)
Barley grain	2.02	88	127.3	Yes	3	0	44
Baseline							
Grass hay	1.12	92	74.6	No	15	15	476
Pasture	1.52	47	97.5	No	0	25	385
Split							
Grass hay	4.04	89	90.6	No	15	15	159
Pasture	3.69	40	162.5	No	0	25	174
Single							
Grass hay	4.21	90	100.4	No	15	15	158
Pasture	3.45	36	181.7	No	0	25	192

[z]Baseline = no application of liquid hog manure on forage land, Split = application of liquid hog manure on forage land in fall and spring (50% of the manure is applied in fall and 50% in spring), Single = application of liquid hog manure on forage land in spring (100% spring application).

[y]Yield and composition of grass hay and pasture were obtained from the field experiment conducted between 2003 and 2006 (Wilson et al., 2010), barley grain yield for La Broiqurie area was obtained from Manitoba Management Plus Program (MMPP, http://www.mmpp.com/mmpp.nsf/mmpp_index.html) and barley straw yield was obtained from Narasimhalu et al. (1998), DM = dry matter.

[x]Harvest and storage loss for grass hay was based on Rotz et al. (2003).

[w]Feeding loss was according to Adams et al. (2009).

[v]Land requirement was calculated based on the productivity of the land and total feed required to feed the animals described in Table 6.4. Land requirement value for barley grain production was an average value for control (55 ha), split (41 ha) and single (37 ha) management scenarios.

Therefore, to investigate the impact of time and amount of manure application on the total farm GHG emissions, three management scenarios, based on the management practices studied in the field, were simulated. The simulated management scenarios were (i) baseline, no application of liquid hog manure on grassland; (ii) application of liquid hog manure at two half application rate of 70 ± 6 kg available N ha^{-1} in fall and spring (split, 50% of the manure is applied in fall and 50% in spring); and (iii) application of liquid hog manure at a full rate of 142 ± 20 kg available N ha^{-1} in spring (single, 100% spring application) . The imported liquid hog manure was surface applied to forage (pasture and grass hay field) using splash plate without incorporation. Greenhouse gas emissions associated with liquid hog manure storage were not accounted. On-farm produced solid manure and imported synthetic N (urea) were applied on the barley field at a rate of 100 kg available N ha^{-1} in which 60% of the N was from solid manure and the rest from urea.

All management practices used on the farm can never fully be represented by a model. However, to make sure that the most important aspects of the farm were properly represented, an evaluation was used to match certain model outputs with known conditions on the farm. Durations of manure storage were adjusted in IFSM to simulate the management scenarios described. Manure storage time was assumed to be six and 12 months in duration to accommodate twice yearly (early April and late October) and annual (early April) manure application frequency for the split and single scenarios, respectively.

As the performance of IFSM is weather dependent (Rotz et al. 2011a), the model was simulated using six years of weather data (2005- 2010). The sward composition of the simulated grassland in IFSM consisted of 99% cool-season grasses (Kentucky bluegrass, orchard grass, ryegrass, tall fescue) and 1% legume (red clover) with an average annual production and quality

described in Table 6.5. The model assumed that approximately half of the assigned grassland is consumed through grazing by beef animals during grazing period (period 3) and the rest is harvested and stored for winter feeding during period 1 and 2.

For the ICM, manure-DNDC was simulated for individual years (2004 to 2006) to estimate soil-related emissions and average values were considered. The model was simulated using the measured soil characteristics and the forage yield and quality data collected from the experimental site (Tenuta et al. 2010; Wilson et al. 2010; Table 6.5). From the model default, pasture and non-legume hay were selected to simulate the field utilized for pasture and hay production in the system, respectively.

6.4. RESULTS AND DISCUSSION

6.4.1. Estimates of the Component Models within ICM

Estimated CH_4 emissions from enteric fermentation and manure management, N_2O emissions from manure management and excretion of N and volatile solid for beef animal categories managed in the production system are summarized in Table 6.6. Beef cows contributed about 69, 68 and 69% of the total farm enteric CH_4 production for baseline, split and single scenarios, respectively. Comparison of model estimated Y_m (% GEI) values ranged between 7.3% GEI, for backgrounding animals in the split scenario and 8.1% GEI, for replacement heifers in the baseline scenario. These values were slightly higher than the IPCC (2006) recommendations of 6.5 ± 1% GEI. Similarly, comparison of daily enteric CH_4 emissions by the different animal categories indicated that emissions varied from 0.24 kg $head^{-1}$ d^{-1} for replacement heifers in the split scenario to 0.35 kg $head^{-1}$ d^{-1} for bulls in the baseline scenario. Regardless of the management scenarios used, average estimated EF (kg CH_4 $head^{-1}$ yr^{-1}) were 124, 95, 126 and 37 for beef cows, replacement heifers, bulls and backgrounding animals, respectively.

The estimated Y_m, daily emissions and EF values were within the range of previously reported values for beef cattle (McCaughey et al. 1999; DeRamus et al. 2003; Ominski et al. 2006; Beauchemin et al. 2009; Alemu et al. 2011). Given the variations in management practice, studies in western Canada (McCaughey et al. 1999; Boadi and Wittenberg 2002; Ominski et al. 2006) reported a Y_m value of 9.5, 6.7 and 8.7% GEI, respectively, for lactating cows managed on grass-based pasture, replacement heifers fed on grass-legume hay and backgrounding steers managed on grass pasture. For daily enteric CH_4 emissions, Beauchemin et al. (2009) reported a range of enteric CH_4 emissions from 0.05 to 0.3 kg head^{-1} d^{-1} and from 0.2 to 0.5 kg head^{-1} d^{-1} for beef cattle and lactating dairy cows, respectively. DeRamus et al. (2003) reported an EF of 73 and 107 kg CH_4 head^{-1} yr^{-1} for beef cows and heifers, respectively, when fed high-forage winter diet with minimum protein supplement. These values were relatively lower than the average EF values estimated in our study for beef cows (124 kg CH_4 head^{-1} yr^{-1}) and replacement heifers (95 kg CH_4 head^{-1} yr^{-1}) which could be due to differences in diet quality. However, a western Canadian study by McCaughey et al. (1999) reported an EF of 108.2 kg CH_4 head^{-1} yr^{-1}.

Regardless of the animal categories, differences were observed for the estimated Y_m and EF among the baseline, split and single scenarios (Table 6.6). These differences may be attributed to the variation in the quality of forages produced following liquid hog manure application on grassland which may have a direct impact on enteric CH_4 production from ruminants. The yield, composition and quality of forage in pastures are altered following liquid hog manure application (Blonski et al. 2004; Wilson et al. 2010; Bork and Blonski 2012).

Table 6.6. Estimated gross energy intake (GEI) and emissions from enteric fermentation and manure management using Integrated Components Model (ICM) for beef cattle categories in the simulated cow-calf production system.

	Beef cows			Replacement heifers			Bulls			Backgrounding animals		
Management scenarios[z]	Baseline	Split	Single	Baseline	Split	Single	Baseline	Split	Single	Baseline	Split	Single
GEI, MJ d^{-1}	231	238	252	175	175	185	247	236	252	182	199	200
Y_m, (%GEI)	7.9	7.5	7.5	8.1	7.5	7.7	7.8	7.5	7.5	7.6	7.3	7.4
Enteric CH_4												
kg ($head^{-1}$ d^{-1})	0.3	0.3	0.3	0.3	0.2	0.3	0.4	0.3	0.3	0.3	0.3	0.3
EF, kg ($head^{-1}$ yr^{-1})	127	119	127	98	89	97	131	119	128	37	37	38
Total, kg yr^{-1}	19020	17914	19064	2362	2141	2334	916	831	896	3757	3876	3953
Volatile solid production,												
kg ($head^{-1}$ d^{-1})	3.4	3.6	3.8	2.5	2.6	2.8	3.6	3.5	3.8	2.6	2.7	2.9
Total, t yr^{-1}	182	196	208	21	23	24	9	9	10	49	52	56
Nitrogen intake, g d^{-1}	221.5	266.8	246.7	156.4	211.3	185.6	236.2	268.5	253.8	185.4	188.9	170.0
Nitrogen excretion, g d^{-1}	194.3	239.6	219.4	142.1	197.0	171.3	225.0	257.3	242.6	161.8	165.3	146.4
Solid on-farm stored manure[y]												
Total CH_4, kg yr^{-1}	2009	2130	2290	236	243	252	101	96	105	904	928	1002
Total N_2O, kg yr^{-1}	110	99	91	12	14	11	6	5	5	52	53	47

GEI = Gross energy intake, EF = Emission factor, Y_m = CH_4 conversion factor (%GEI, by diet).

[z]Baseline = no application of liquid hog manure on forage land, Split = application of liquid hog manure on forage land in fall and spring (50% of the manure is applied in fall and 50% in spring), Single = application of liquid hog manure on forage land in spring (100% spring application).

[y]Soild on-farm stored manure N_2O emissions were the sum of direct and indirect N_2O emissions.

Various studies demonstrated that forage availability and quality (*i.e.,* nutrient composition and digestibility) has a significant impact on enteric CH_4 emissions from ruminants (McCaughey et al. 1999; Boadi and Wittenberg 2002; Boadi et al.2002; Ominski et al. 2006; Ominski and Wittenberg 2006). Boadi et al. (2002) indicated that steers grazing during the early period of grazing season had a 44 and 20% less energy lost as Ch4 relative to steers grazing during the mid and late grazing periods, respectively

Regardless of the management scenarios and animal categories, estimated N excretion ranged from 0.14 to 0.26 kg $head^{-1}$ d^{-1} and volatile solid excretion ranged from 2.5 to 3.8 kg $head^{-1}$ d^{-1}. The average daily N excretion per head was highest for bulls (0.25 kg) followed by beef cows (0.22 kg) and replacement heifers and backgrounding animals (0.18 kg). Comparison of the management scenarios indicated that the estimated total daily N excretion was lower for the baseline scenario (0.72 kg) relative to split (0.86 kg) and single (0.78 kg) scenarios. Similarly, total daily volatile soild excretion was higher for split (12.5 kg) and single (13.3 kg) relative to baseline scenario (12.1 kg). Comparison of daily average volatile soild excretion per head for the animal categories were higher for bulls (0.34 kg) followed by beef cows (0.33 kg) and replacement heifers and backgrounders (0.26 kg). The estimated N_2O and CH_4 emissions from on-farm stored solid manure were influenced by the observed variation in N and volatile solid excretion, respectively (Table 6.6). Erickson et al. (2003) reported that for beef cows (544 kg BW) consuming 2.3% of BW with zero N retention, excretion of N varied between 0.17 to 0.25 kg d^{-1} and excretion of volatile solid varied between 4.2 to 5.9 kg d^{-1}. Furthermore, the American Society of Agricultural and Biological Engineers (ASABE 2005) reported an average N excretion value of 0.19 and 0.13 kg d^{-1} and average volatile solid excretion value of 5.9 and 2.3 kg d^{-1}, respectively, for beef cows and growing calves under confined management.

However, they acknowledged that these average estimates may vary with changes in animal genetics, feeding and management strategies and quality of available feeds.

Estimated direct soil N_2O emissions using manure-DNDC, for the grass hay field, were compared to the measured N_2O emission values for the same field reported by Tenuta et al. (2010) for the dates that correspond with the reported measured values. Model estimated values were 0.31, 0.38 and 0.10 g N ha^{-1} d^{-1} for the control, 3.13, 4.37 and 2.96 g N ha^{-1} d^{-1} for split and 9.10, 5.96 and 1.90 g N ha^{-1} d^{-1} for single application of liquid hog manure for 2004, 2005 and 2006, respectively. These values were comparable to those reported by Tenuta et al. (2010) for the field experiment where average measured N_2O emissions were 0.13, 0.61 and 0.25 g N ha^{-1} d^{-1} for the control treatment, 2.36, 2.91, 1.24 g N ha^{-1} d^{-1} for split and 7.63, 5.43 and 1.48 g N ha^{-1} d^{-1} for single treatments for 2004, 2005 and 2006, respectively. The authors explained that monitoring of N_2O emissions were conducted after spring thaw, however they suggested that N_2O emissions prior to spring thaw were minimal.

Average annual soil CH_4 emissions estimated using manure-DNDC model were negative indicating consumption or uptake by the soils: -0.80, -1.0 and -0.91 g C ha^{-1} d^{-1} for control, split and single scenarios, respectively. However, for the same site Tenuta et al. (2010) monitored CH_4 emissions from(May to October, and reported an average emission of 1.6, 3.5 and 2.7 g C ha^{-1} d^{-1} for control, split and single scenarios, respectively. Generally, well aerated soil are a sink for atmospheric CH_4 via methanotrophic oxidation (Le Mer and Roger 2001; Borken and Beese 2006) and the consumption rate could be influenced by N fertilization (Scerlock et al. 2002; Dittert et al. 2005; Tenuta et al. 2010). Methane emissions have been shown to increase following manure application on grass land (*e.g.,* Scerlock et al. 2002; Dittert et al., 2005). Sherlock et al. (2002) reported that CH_4 emissions of 39 g C ha^{-1} d^{-1} were observed

immediately after hog slurry application to a pasture plot and decreased to 10 g C ha^{-1} d^{-1} within 6 h. and then emissions continued at a lower rate for 7 d. Thereafter, the authors observed a net uptake of atmospheric CH_4 by the treated plots. The limitation of monitoring CH_4 for a short period following manure application is that emissions are mainly contributed by volatilization (degassing) of dissolved CH_4 from the applied manure rather than emissions from soil via methanotrophic activity (Sherlock et al. 2002). Conversely, the manure-DNDC model estimates mainly soil CH_4 and does not account for CH_4 emissions contributed by degassing of dissolved CH_4 following manure application (Table 6.1). Therefore, it is challenging to directly compare measurements of CH_4 from field experiments that account for dissolved CH_4 from applied manure with model estimated values which are mainly from soil methanotrophic activity.

6.4.2. Comparison of Model Estimates for Farm GHG Emissions

Table 6.7 summarizes the total GHG emissions from different sources in the production system and farm GHG intensity estimates for the simulated cow-calf production system using ICM, IFSM and Holos model. Observed variation among models in GHG intensity estimates were 3 to 13% for the baseline scenario, and 5 to 10% for split and single scenarios. Total farm GHG emission estimates were generally higher for split and single scenarios compared to that of baseline scenario. Specifically, farm GHG intensity estimates for the baseline scenario were higher using Holos model (31.5 kg CO_2 eq kg^{-1} carcass weight) compared to ICM (27.9 kg CO_2 eq kg^{-1} carcass weight) and IFSM (28.8 kg CO_2 eq kg^{-1} carcass weight). A comparison of model estimates for the different management scenarios indicated that emission intensity estimates using ICM were 66% higher for split and 49% higher for single scenario relative to estimates for the baseline scenario (27.9 kg CO_2 eq kg^{-1} carcass weight). Similarly, IFSM estimates were 45

and 51% higher for split and single scenarios, respectively, compared to the values for the baseline scenario (28.8kg CO_2 eq kg^{-1} carcass weight) estimated by the model. Similarly, when farm GHG emissions intensity was expressed per unit hectare, the estimated values for the baseline scenario were lower relative to the split and single scenarios (Table 6.7) due the lower land productivity and large land size requirement (Table 6.5) to support the production cycle. Comparison of ICM and IFSM for GHG intensity estimates from the split and single scenarios indicated that ICM yielded estimates that were 10% higher for split scenario compared to those generated using IFSM (41.9 kg CO_2 eq kg^{-1} carcass weight). Conversely, for the single scenario, IFSM yielded estimates that were 5% higher than ICM (41.6 kg CO_2 eq kg^{-1} carcass weight).

The observed differences among models in estimating farm GHG intensity are expected as different boundaries, approaches, assumptions and algorithms were used to estimate GHG emissions from the various sources in the production system (Table 6.1). As such, direct comparison among the model estimates is challenging. Differences in enteric CH_4 emissions, for example, may be attributed to differences in methodologies as enteric CH_4 emissions were estimated using VFA stoichiometric model (Bannink et al. 2006) in ICM, an empirical nonlinear equation (Mills et al. 2003) in IFSM and a fixed EF (IPCC 2006) in Holos model. Significant differences in emission estimates have been reported among these methodologies (Benchaar et al. 1998; Kebreab et al. 2008; Legesse et al. 2011; Alemu et al. 2011).

Furthermore, the models vary in the methodology used to estimate N_2O emissions from cropland. Soil direct N_2O emissions were quantified using manure-DNDC in ICM, simplified DAYCENT model in IFSM and a modified IPCC approach in Holos model (Table 6.1). Several local and global scale studies compared DAYCENT and DNDC models and observed differences in estimated N loss from cropland (Del Grosso et al. 2006; 2009; Smith et al. 2008;

Abdalla et al. 2010). Abdalla et al. (2010) reported differences between DAYCENT and DNDC in estimating soil N_2O emissions and grass biomass from pasture land. Similarly, Smith et al. (2008) reported differences between the two models in estimating N_2O emissions from crop land fertilized with liquid hog manure as well as cropland with different tillage practices in Eastern Canada (Quebec and Ontario). The limitations of using IPCC methodology in a whole-farm analysis have been reviewed by Crosson et al. (2011).

The assumptions that were made within the whole-farm models to simulate the beef production system could also account for the observed differences in GHG intensity estimates generated by the models. For example, the calculation of CO_2 emissions from on-farm energy use for imported liquid hog manure in IFSM assumed that supplier of the manure provided the equipment, fuel and labour to transport and spread the manure on farm. As the farm owner provided a service by supplying land for disposal of manure and benefited from the added nutrient, IFSM does not include energy use CO_2 emissions related to imported manure transportation and application. However, ICM incorporated CO_2 emissions from energy use for application of imported liquid hog manure that contributed 107 and 86 kg CO_2 ha^{-1} for the split and single scenarios, respectively. Conversely, IFSM incorporated CO_2 emission related to manufacture of farm machinery that increased CO_2 emissions from secondary sources which is not considered in ICM and Holos model (Table 6.1).

Animal categories also differ between models. Breeding bulls that may contribute up to 3% of total farm enteric CH_4 emissions (Beauchemin et al. 2010) are not included in IFSM but part of ICM and Holos model analysis.

Table 6.7. Greenhouse gas emissions and farm emission intensity for baseline, split and single scenarios estimated using Integrated Components Model (ICM), Integrated Farm System Model (IFSM) and Holos model.

	ICM			IFSM[z]			Holos[y]
Management scenarios[x]	Baseline	Split	Single	Baseline	Split	Single	Baseline
	Emissions (Mg CO_2 eq)[w]						
Enteric CH_4	688	655	692	664	660	688	640
Manure CH_4	81	85	91	34	68	69	36
Manure N_2O	46	43	38	13	24	27	212
Direct N_2O	41	39	35	11	21	23	181
Indirect N_2O	5	4	3	2	3	4	32
Soil CH_4	-7.4	-4.3	-3.0	…	…	…	…
Soil N_2O	80	700	516	177	598	604	91
Direct N_2O	64	402	292	144	215	219	76
Indirect N_2O[v]	16	298	224	33	383	385	15
Energy CO_2	39	54	48	71	43	63	65
Total GHG emissions	926	1,533	1,383	957	1,390	1,449	1,045
GHG intensity, kg CO_2 eq kg^{-1} beef							
Live weight basis	16.7	27.7	25.0	17.3	25.1	26.2	18.9
Carcass weight basis	27.9	46.1	41.6	28.8	41.9	43.6	31.5
GHG intensity, Mg CO_2 eq ha^{-1}	1.0	4.1	3.6	1.1	3.7	3.7	1.1
Productivity (weight (land size)$^{-1}$)							
kg live weight ha^{-1}	60.5	147.99	143.0	60.5	147.9	143.0	60.5
kg carcass weight ha^{-1}	36.3	88.8	85.8	36.3	88.8	85.8	36.3

[z]Net biogenic CO_2 estimates were not included into the total farm GHG estimates.

[y]Holos model was used to estimate emissions only from the baseline scenario because the model has no options to import manure as a N source into the production system.

[x]Baseline = no application of liquid hog manure on forage land, Split = application of liquid hog manure on forage land in fall and spring (50% of the manure is applied in fall and 50% in spring), Single = application of liquid hog manure on forage land in spring (100% spring application).

[w]CO_2 eq was calculated using a Global Warming Potential of 1 for CO_2, 25 for CH_4 and 298 for N_2O (IPCC, 2007).

[v]Soil indirect N_2O emissions for ICM and IFSM were calculated from model-estimated volatilization and leaching losses and IPCC default emission factors for leaching (0.0075 kg N_2O-N) and volatilization (0.01 kg N_2O-N).

6.4.3. Contribution of Various GHG Sources to Whole-farm Emissions

The breakdown of emission by source (CO_2 eq) is reported in Figure 6.2. For the baseline scenario, enteric CH_4 was the primary contributor of total farm GHG emissions contributing 74, 69 and 61% of the total emission using ICM, IFSM and Holos models, respectively. The second largest contributor of GHG for the control scenario was N_2O (20%) using Holos model, CH_4 (9%) using ICM from on-farm stored solid manure and soil N_2O (15%) using IFSM. The contribution of emission sources to the total farm emission estimated using Holos model differed from the values reported by Beauchemin et al. (2010). Presumably, this variation may be due to the aforementioned differences between the two studies. Moreover, the estimated contribution of enteric CH_4 for the baseline scenario using ICM was higher than the previously reported ranges of estimates, using IPCC methodology (40 to 70% of total emissions) for a North American beef production system (Johnson et al. 2003 and Vergé et al. 2008). The higher proportion observed in our study may be attributed to the higher EF used by the COWPOLL model (7.3 to 8.1%, Table 6.6) compared to the default IPCC EF value (6%) used in the previous reports. In another study, Alemu et al. (2011) also reported higher estimates of enteric CH_4 emissions using COWPOLL model compared to IPCC Tier 2 for similar beef production system. Contribution of N_2O to the overall emission intensity in the literature conducting whole-farm GHG analysis varies considerably from 27% (Beauchemin et al. 2010) to 52% (Johnson et al. 2002). Contribution of N_2O in the baseline scenario estimated using Holos model (29%) was comparable to the value (27%) reported by Beauchemin et al. (2010). However, in the current study about 70% of the N_2O was contributed by manure N_2O from on-farm produced solid manure relative to 85% in Beauchemin et al. (2010) study.

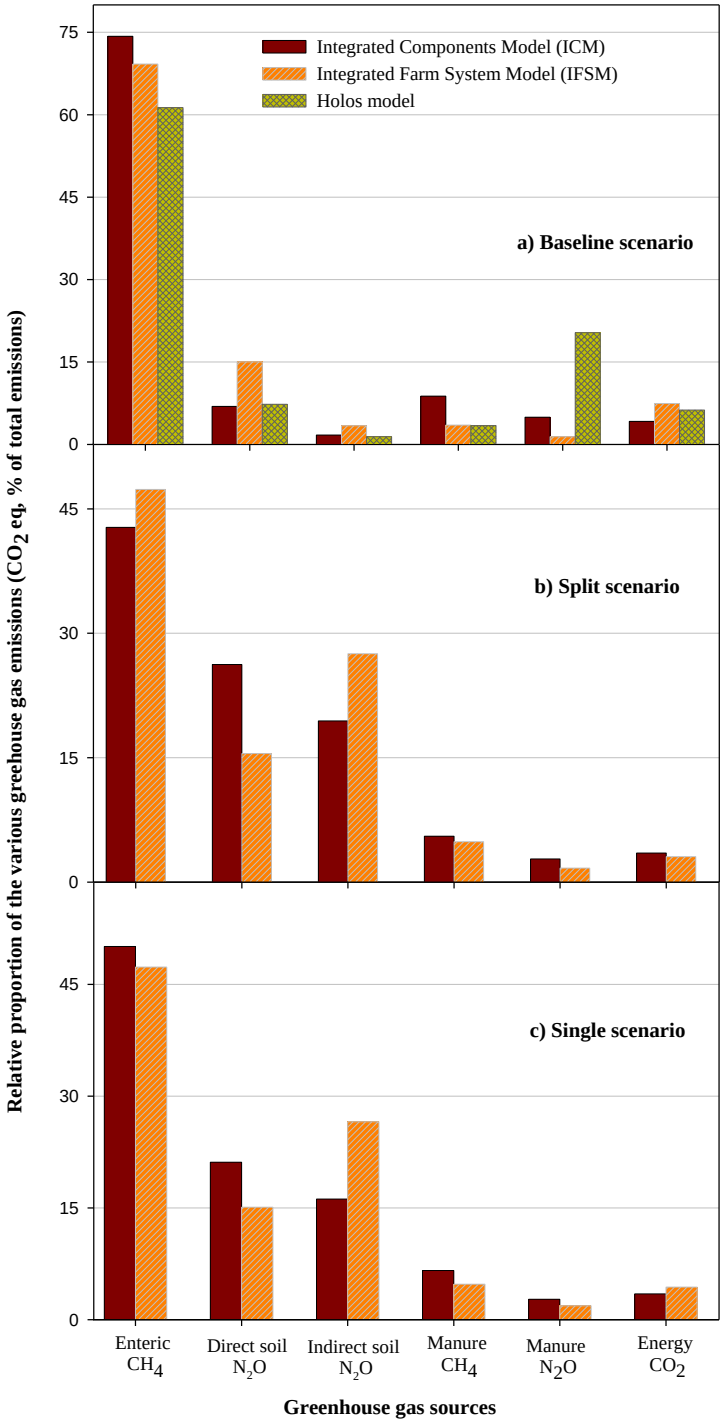

Figure 6.2. Relative proportion of the various GHG emissions (CO_2 eq, % of total emissions) in a cow-calf production system estimated using Integrated Components Model (ICM), Integrated Farm System Model (IFSM) and Holos model for (a) baseline scenario, (b) split scenario and (c) single scenario. The Holos model was used only for the baseline scenario as the model does not have an option to import liquid hog manure into the system.

For split and single scenarios estimates using ICM and IFSM, enteric CH_4 and soil N_2O were the major contributors to the total farm GHG emissions followed by on-farm stored solid manure and energy use (Figure 6.2). Relative to the baseline scenario, the application of liquid hog manure in split and single scenarios increased contribution of direct soil N_2O emissions by 0.1 to 19% and indirect soil N_2O emissions by 14 to 26%. Contribution of soil N_2O estimated using ICM was higher for split (46%) than single scenario (37%) due to the higher emissions from direct and indirect soil N_2O for the split compared to the single scenarios (Table 6.7, Figure 6.2). Emission estimates for direct soil N_2O emissions using ICM was 1.96 kg N ha^{-1} for split compared to 1.50 kg N ha^{-1} for single. Conversely, proportional contribution of soil N_2O estimated using IFSM was relatively similar for split (43%) and single scenarios (42%). However, unlike the ICM, soil N_2O estimates using IFSM were mainly contributed as a result of indirect emission for both the split (28%) and single (27%) scenarios (Figure 6.2). More specifically emissions from NH_3-N volatilization loss from field applied manure accounted for approximately 97% of the indirect emissions. Johnson et al. (2002) conducted a whole-farm analysis of GHG emissions for a cow-calf through feedlot production system in the U.S., where on-farm produced solid manure was deposited on pasture. These authors reported that enteric CH_4 contributed 36% and N_2O contributed 52% of the total farm GHG emissions. Of the total N_2O emissions, up to 54% resulted from manure application or deposition during grazing. A higher estimate (60%) for relative contribution of N_2O emissions was also reported by Flessa et al. (2002) for a conventional beef farm that applied on-farm produced slurry to the cropland. Overall, application of liquid hog manure increased direct and indirect soil N_2O emissions and thus its contribution to the overall farm GHG intensity.

Various mitigation strategies have been reported to minimize N loss including those used

to reduce the amount of N applied to land in manure (*e.g.,* dietary measures, manure treatments) and those strategies that are used to minimize losses once manure has been land applied (*e.g.,* method, rate and time of application) (VanderZaag et al. 2011). However, there is little information regarding the impact of these management strategies on total farm GHG emissions. In the current study, estimates of total GHG emissions using ICM and IFSM were not consistent as ICM estimated a higher emission intensity for split (27.7 kg CO_2 eq kg^{-1} live weight) whereas IFSM estimated a higher emission intensity for the single scenario (26.2 kg CO_2 eq kg^{-1} live weight). Inconsistency has also been observed among prior field scale studies monitoring the impact of time and amount of manure application on GHG emissions. Allen et al. (1996) reported higher soil N_2O emissions for animal manure or faeces applied on grassland in UK during fall than spring application. Whereas, Chadwick et al. (2000) reported higher soil N_2O and CH_4 emissions for slurry applied on grassland in spring compared to fall and summer application. Similarly, for liquid hog manure applied on an annual crop (maize) in Quebec, Rochette et al. (2004) reported a two fold increase in soil N_2O emissions for spring application compared to fall application (1.74% of total hog slurry-applied N). The rate of manure application also influences the amount of C and N added to the soil. For surface applied slurry on pasture, Menzi et al. (1998) reported a linear relationship between rate of manure application and cumulative NH_3 loss. Rochette et al. (2000) observed an increase in N loss as N_2O-N (from 1.23 to 1.65%) when application rate of pig slurry was doubled. Combining rate and time of application, Tenuta et al. (2010) reported lower N loss as direct soil N_2O emissions (0.29% of total hog slurry-applied N) for liquid hog manure applied on grassland in a split application where half of the required manure is applied in spring and the rest in fall compared to applying all the required manure once in spring (0.51% of total hog slurry-applied N). Further component

and whole-farm assessment are required regarding the time and amount of livestock manure application in beef production systems in order to fully understand net GHG emissions.

6.4.4. Comparison of Model Estimates to Earlier GHG Intensity Estimates

Figure 6.3 depicts farm GHG emissions intensity estimates reported in previous whole-farm studies for different beef production systems. Given the diversity and complexity of beef production systems, it is not surprising that these differences exist among whole-farm studies in the type of production system (*i.e.*, cow-calf, backgrounding, feedlot, cow-calf and backgrounding, cow-calf through feedlot), as well as management practices (*e.g.*, feeding, housing) and boundaries used in the system. Further, differences also exist among studies in the type of models used, the EF values used to estimate emissions, as well as functional units used to express the environmental impact of products produced from the system. As such, direct comparison of farm GHG intensity estimates from the current study with estimates reported in prior studies could be misleading. However, considering these differences, a relative comparison was made between model estimates in the current study and estimates from prior studies to assess the validity of the results and to draw some general conclusions.

Estimates of GHG intensity from beef production systems at the farm gate using whole-farm approach reported in other studies range from 8.4 kg CO_2 eq kg^{-1} carcass weight for a pasture finished feedlot operation (Subak 1999) to 37.5 kg CO_2 eq kg^{-1} carcass weight for a cow-calf operation (Phetteplace et al. 2001).

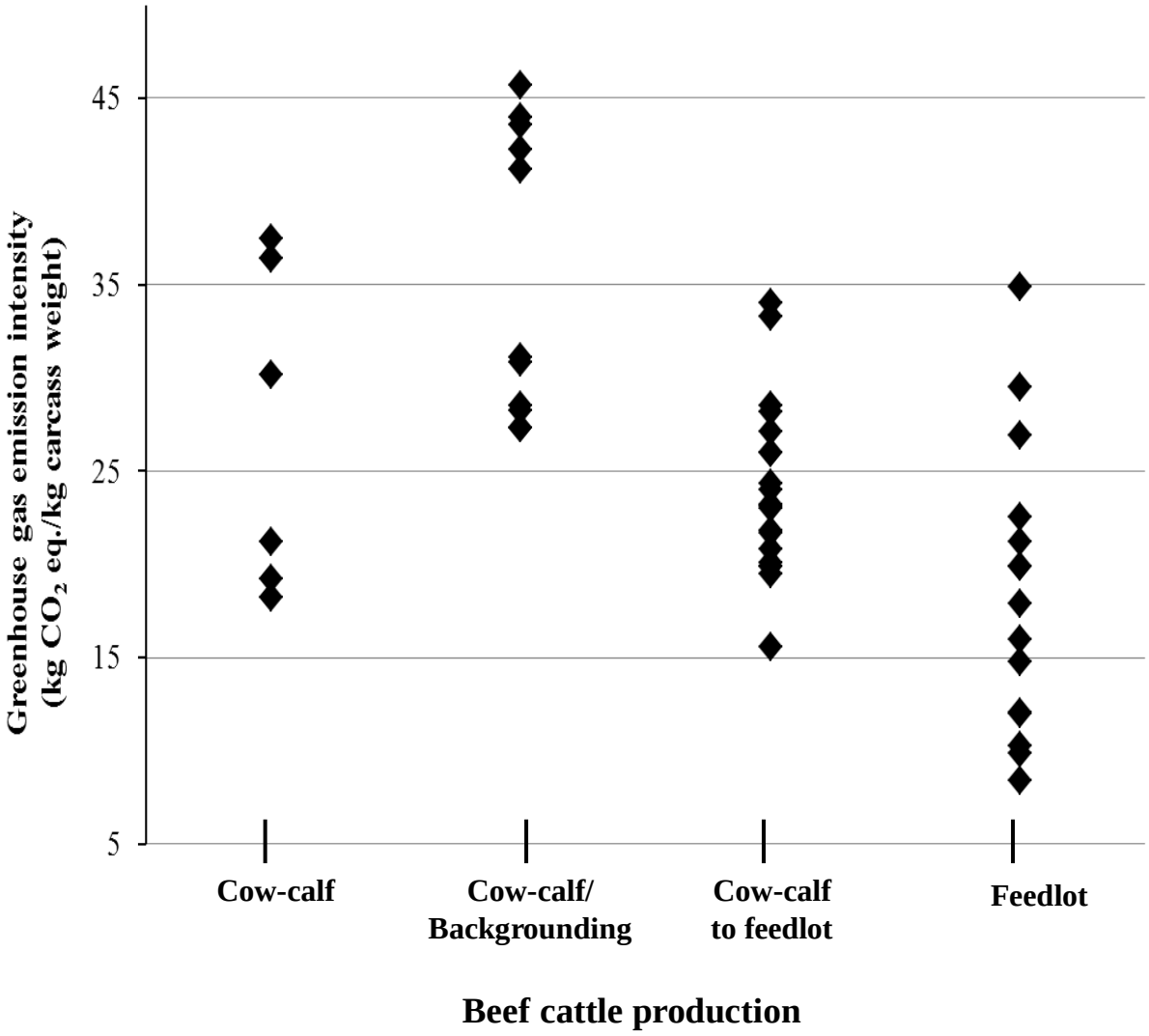

Figure 6.3. Farm greenhouse gas emissions intensity (expressed per unit kg carcass weight exported out of the farm gate) for cow-calf, cow-calf/backgrounding, cow-calf to feedlot and feedlot production systems. Data was organized from Subak (1999), Phetteplace et al. (2001), Johnson et al. (2002), Cederberg and Stadig (2003), Casey and Holden (2006a; b), Ogino et al. (2007), Pelletier et al. (2010), Phetteplace et al. (2010), Nguyen et al. (2010), Crosson et al. (2010), Peters et al. (2010), Veysset et al. (2010), White et al. (2010), Beauchemin et al. (2010; 2011) Cederberg et al. (2011) and Basarab et al. (2012). Emission intensity estimates from the current study were also incorporated in the graph under cow-calf/backgrounding production system.

A much higher intensity (44 kg CO_2 eq kg^{-1} carcass weight) has been reported by Cederberg et al. (2011) for Brazilian beef production that includes land use change due to deforestation (28.5 kg CO_2 eq kg^{-1} carcass weight excluding land use change). Casey and Holden (2006b) reported a GHG intensity of 29 to 31 kg CO_2 eq kg^{-1} carcass weight from pasture-based, conventional sucker-beef production in Ireland. However, for a similar production system using different approaches, Foley et al. (2011) reported an emission intensity of 23.1 kg CO_2 eq kg^{-1} carcass weight. In the Canadian beef production system, Verge et al. (2008) reported a GHG intensity value of 10 kg CO_2 eq kg^{-1} live weight while Beauchemin et al. (2010) reported a value of 21.7 kg CO_2 eq kg^{-1} carcass weight. Beauchemin et al. (2010) conducted a LCA on simulated beef and cropping production over eight years using the Holos model and reported a carbon foot print of 13.0 kg CO_2 eq kg^{-1} live weight and 21.7 kg CO_2 eq kg^{-1} carcass weight. Compared to the Beauchemin et al. (2010) estimates, the emission intensity estimates from the baseline scenario using Holos model in our study were relatively higher, 18.9 kg CO_2 eq kg^{-1} live weight and 31.5 kg CO_2 eq kg^{-1} carcass weight. This variation is likely due to the difference in the simulated beef production system. Beauchemin et al. (2010) simulated emissions from cow-calf through feedlot, whereas our analysis did not include a feedlot phase. The higher efficiency of feedlot phase could influence GHG intensity (Johnson et al. 2002; Capper 2011; Figure 6.2). Phetteplace et al. (2001) reported 33% higher emission intensity for conventional cow-calf production than cow-calf through feedlot production where the feedlot operation was part of the production system (15.5 kg CO_2 eq kg^{-1} live weight gain).

Ogino et al. (2007) reported emission intensity of 36.4 kg CO_2 eq kg^{-1} carcass weight for a Japanese cow-calf production system where cows and calves were managed in the barn and calves were sold at the age of 8 months. Even though the analyzed Japanese production system

resembles our system in some aspects, such as simulating the cow-calf production and selling backgrounded animals, the estimated emission intensity was higher compared to our baseline intensity estimates. However, it should be noted that in addition to using a retail beef yield percentage of 40% to calculate the GHG intensity per kg carcass produced, the Japanese beef simulation used a linear equation, based on DM intake, to calculate enteric CH_4 emissions, sold calves earlier (8 months), assumed a 14 month calving interval, transported manure off farm as compost, did not include carcass produced from culled animals (cows), transported feed from longer distances and used a global warming potential of 23 for CH_4 and 296 for N_2O.

There is a paucity of information regarding recycling of livestock manure in the production system from a whole-farm perspective of analyzing GHG emissions (Petersen et al. 2007). As such, it is difficult to find studies reporting whole-farm GHG intensity from beef production that use animal manure as N source in order to compare intensity estimates from split and single scenarios as was done in the current study. Casey et al. (2006a, b) examined application of on-farm produced cattle slurry twice a year at the rate of 50 m^3 ha^{-1} combined with synthetic fertilizer and reported emission intensities ranging from 24 to 31 kg CO_2 eq kg^{-1} carcass weight. These values are lower than the intensity values for split and single scenario using ICM and IFSM in the current study in which all the required N in the farm was sourced from animal manure (imported liquid hog manure and on-farm produced solid manure). Higher soil N_2O emissions have been reported from soils that received livestock manure compared to synthetic fertilizer. Smith et al. (2008) compared N_2O emissions from maize field that received synthetic fertilizer and pig slurry at different application rates. They reported 0.23 kg ha^{-1} of N_2O emissions for the field that received synthetic fertilizer (150 kg ha^{-1}) and 1.21 and 3.1 kg ha^{-1} for the maize field that received pig slurry at the rates of 60 and 120 t ha^{-1}, respectively. Therefore,

the higher GHG intensity estimates for split and single scenarios in the current study may be associated with the use of liquid hog manure on forage land as a consequence of increased soil N_2O emissions.

6.4.5. Land Use Efficiency

In order to support the production cycle within the simulated cow-calf production system, the control scenario required bout 528 to 541 hectare more land compared to the split and single scenarios (Table 6.5). .The variation may be due to differences in land productivity following liquid hog manure application (Wilson et al. 2010). When emission intensity was expressed per unit land area, baseline, split and single scenarios had an average value of 1.1, 3.9 and 3.6 Mg CO_2 eq ha^{-1} respectively (Table 6.7). For the baseline scenario, the estimated value was less than the previously reported values that ranged between 1.34 and 1.80 t CO_2 eq ha^{-1} for North American beef production managed on pasture and mixed hay (Johnson et al. 2002; Beauchemin et al. 2010). In the current study, for the baseline scenario, naturalized grassland with lower land productivity (Table 6.5) was used for pasture and hay production which increased the amount of land required to support the production cycle and therefore reduced land use efficiency. Conversely, land use efficiency values for the split and single scenarios were comparable to values reported by Flessa et al. (2002), Casey et al. (2006b) and Foley et al. (2011), which ranged between 3.32 and 5.89 t CO_2 eq ha^{-1}, for European beef production that applied on-farm produced slurry to cropland. In terms of carcass productivity per unit land area, regardless of the models used, the baseline scenario had the lower average productivity (36.3 kg carcass ha^{-1}) relative to split (88.8 kg carcass ha^{-1}) and single (85.8 kg carcass ha^{-1}) scenarios., Beauchemin et al. (2010) reported carcass productivity of 61.4 kg carcass ha^{-1} for conventional extensive beef production in Southern Alberta based on native prairie pasture and dryland crop production.

6.4.6. Implications

Various mitigation strategies have been proposed and implemented to minimize GHG from animal agriculture (Boadi et al. 2004c; Beauchemin et al. 2009; Eckard et al. 2010). However, often the strategies are applied to a single farm component (*e.g*., animal, soil) and as such it is difficult to evaluate their impact from a whole-farm perspective. In a review of on-farm level modelling approach by Schils et al. (2007), it was concluded that a whole-farm approach is a powerful tool for the development of cost-effective GHG mitigation options, as relevant interaction between farm components are revealed. Our study clearly demonstrates the potential to integrate detailed process-based farm component models such as COWPOLL and manure-DNDC into a whole-farm model in order to analyze total farm GHG emissions. Given the complexity of animal agriculture, incorporation of process-based models in a whole-farm analysis provides an opportunity to incorporate detailed information regarding the production system under investigation, thereby providing greater flexibility to incorporate different management scenarios (Janzen et al. 2006). The current study also revealed that in addition to variation due to differences in the management practices employed in the production system, the type of models used to estimate whole-farm GHG emissions intensity are also a source of variation.

In western Canada there is a lack of knowledge regarding GHG emissions after application of livestock manure to grassland, and even less is known about its environmental impact from the whole-farm perspective. Although variation was observed among models in estimating GHG intensity for the studied scenarios, application of liquid hog manure increased the contribution of direct and indirect soil N_2O emissions which contributed to the observed higher intensity estimates for split and single scenarios. Mitigation strategies in these scenarios should target

N_2O emissions related to manure application and CH_4 emissions from enteric fermentation.

Loss of N from land applied manure is a function of time, rate and methods of application (Menzi et al. 1998; Rochette et al. 2004; 2008c; Tenuta et al. 2010; Rotz et al. 2011b). In our study liquid hog manure was surface applied without incorporation which could have increased the loss of N through volatilization (Rochette et al. 2008c). A whole-farm simulation study by Rotz et al. (2011b) compared the environmental performance of different manure application methods for different production systems. Ammonia nitrogen (NH_3-N) loss in a swine and cow-calf beef operation under grass production was reduced by 48% with shallow disc injection compared to surface broadcast without incorporation and band application with aeration. Moreover, VanderZaag et al. (2011) conducted a simulation study in eastern Canada using a simplified approach and reported an 18% reduction in total GHG emissions from the swine sector could be achieved by shifting the application of manure from fall to spring and incorporating all manure within one day of application. In contrast, studies also indicated that incorporation or injection of slurry has a risk of increasing soil N_2O and CH_4 emissions (Velthof et al. 2003; Rodhe et al. 2006). This implies that there is a need for an integrated approach to assess the impact of application techniques as well as time and amount of manure application on GHG emissions for a production system applying livestock manure as a N source. Implementation of process-based whole-farm models could provide more flexibility to incorporate the various factors that influence GHG production.

In the current study, enteric CH_4 emission was one of the major contributors to the total GHG emissions (43 to 74%) which is mainly contributed by beef cows (68 to 69%). Thus, strategies to reduce emissions might best be aimed at reducing enteric CH_4 emissions. Various dietary and animal husbandry practices that reduce enteric CH_4 emissions have been reviewed

(Boadi et al. 2004c; Beauchemin et al. 2009; Eckard et al. 2010). Beauchemin et al. (2011) conducted a whole-farm simulation study using a LCA to evaluate the impact of various mitigation strategies for enteric CH_4 production (dietary modification and improved animal husbandry) on net farm GHG emissions. They reported that total farm emission can be reduced up to 8% by applying individual mitigation options on caw-calf operation and by combining the mitigation options, emissions can be reduced by up to 17%. These authors also indicated that the biggest reductions in GHG emissions are achieved when the strategies target reducing enteric CH_4 emissions from cow-calf production rather than feedlot operation, where application of individual mitigation strategies resulted in less than a 2% reduction.

Cow-calf producers in western Canada are moving away from overwintering cows in feedlot to in-field wintering system in which beef cattle are fed directly on pasture and manure is deposited directly in the field (McCartney 2011; AAFC 2011). This management practice avoids accumulation of manure during winter management when beef cattle are managed in drylot and its associated GHG emissions. In our study, on-farm managed solid manure contributed 4 to 9% CH_4 and 1 to 20% N_2O emissions. Prior studies indicated that winter feeding of beef cattle on pasture reduced enteric CH_4 production (Takahashi et al. 2002; Bernier, 2012), increased nutrient recycling efficiency (Jungnitsch et al. 2011; Kelln et al. 2012) and reduced winter feeding cost (Kelln et al. 2011). Animals managed outdoor under cold temperature produce less enteric CH_4 (Kennedy and Milligan 1978; Takahashi et al. 2002; Bernier et al. 2012) which could influence the total farm GHG emissions. Therefore, it would be worth exploiting this management practice in terms of its impact on total farm GHG emissions.

6.5. CONCLUSIONS

Our study clearly indicated the possibility of integrating the extant process-based farm

component models into a whole-farm model in order to analyze farm GHG intensity. Given the complexity of beef production system and the various factors influencing GHG emissions, application of process-based models in a whole-farm GHG analysis provides the opportunity to incorporate detailed management information for specific production systems. Furthermore, the environmental benefits of potential production system needs to be fully assessed by including the various benefits such as carbon sequestration and other ecosystem services including biodiversity and wildlife habitat. This can only be achieved by developing a more robust ecosystem-level models that incorporates detailed system-specific information which could be achieved by integrating the existing process-based farm component models.

The GHG intensity per unit live weight leaving the farm gate, regardless of the models used, varied from 16 to 19, 25 to 27 and 25 to 26 kg CO_2 eq, respectively, for the baseline, split and single scenarios. Relative to the baseline scenario, application of liquid hog manure in split and single scenarios increased GHG emission intensity due to the higher contribution of soil N_2O emissions. However, split and single scenarios had higher carcass productivity per unit land area due to improved land productivity following liquid hog manure application. Differences were observed among model estimates in terms of the contribution of different GHG sources to the total farm emissions due to differences in assumptions, approaches and algorithms used in the model. These differences could influence identification of the key areas in the beef production system where management systems are recommended to improve the environmental efficiency. Furthermore, limited information is available on the environmental impact of hog manure application on grassland from a whole-farm perspective, therefore, there is a need for further studies.

7. GENERAL DISCUSSION

Several studies have demonstrated the challenges and limitations of rumen stoichiometric VFA modelling, the impacts of types of models on enteric CH_4 inventory, as well as the importance of using whole-farm modelling for estimation of total farm GHG emissions. The following sections further discuss the challenges of modelling rumen fermentation, the impact of model selection on enteric CH_4 inventory as well as importance and challenges of whole-farm modelling of GHG emissions.

7.1. Challenges of Modelling Rumen Fermentation

Fermentation in the rumen is a complex process involving degradable dietary components and microbial activities that produce VFAs, CH_4, CO_2 and NH_3 as end products. Accurate representation of this complex process using mathematical models is also challenging as limited information regarding many of the processes including rumen microbiota and their metabolism as well as production and metabolism of rumen VFAs are available.

7.1.1. Modelling rumen volatile fatty acid production

One of the significant weaknesses in current models of rumen function is accurate representation of VFA dynamics. Our study showed that despite the various efforts made to improve the VFA stoichiometric models, their prediction accuracy remains low. Wide variation in estimating individual VFA proportions (root mean square prediction error (RMSPE) ranging from 5.2 to 43.2%) was observed among the evaluated VFA models, Murphy et al. (1982, MUR), Bannink et al. (2006, BAN), Sveinbjörnsson et al. (2006, SVE) and Nozière et al. (2010, NOZ). Moreover, less than 23% of the variation in rumen VFA production was accounted for by the models.

Several studies investigated the likely sources of error to explain the inability of rumen

fermentation models to predict VFA molar proportions (Bannink et al. 1997a; Nagorcka et al. 2000). Bannink et al. (1997a) conducted a simulation study and reported that inadequate representation of VFA coefficients that relate VFA formed to type of substrate fermented, inadequate representation of VFA production, and inadequate representation of VFA absorption through the rumen wall were among the most probabilistic causes. With the aim of improving the prediction performance of the BAN model and addressing the mentioned probable sources of variation, Bannink et al. (2008) developed new stoichiometric coefficients using the same dataset used to develop the VFA coefficients for the BAN model. The improvements made included the addition of a sigmoidal relationship between rumen pH and fraction of substrate converted to acetate propionate and butyrate as well as a nonlinear relationship between VFA concentration and VFA absorption (Bannink et al. 2008). For the BAN, MUR and SVE models the effect of rumen pH on rumen degradation was addressed by categorizing the diet into forage and concentrate, whereas the relationship between VFA concentration and VFA absorption was assumed to be linear. However, in agreement with our study, evaluation of the new VFA coefficients of Bannink et al. (2008) still showed poor prediction accuracy (Morvay et al. 2011). The authors reported a wide variation in estimating individual VFA molar proportions (RMSPE ranging from 7.2 to 105.9%) among the six VFA stoichiometric models evaluated in their study. Variation in degradation rate in the rumen cannot be fully explained by rumen pH or types of diet. Relating fermentation to rumen pH, for example, neglects other factors that affect fermentation, such as buffering from feed and saliva (Giger-Reverdin et al. 2002). Overall, there are gaps in knowledge to represent the rumen microbial fermentation process, such as VFA production using mathematical models.

These gaps in knowledge may be addressed through advancements in technology and

evolving data on rumen fermentation and rumen microbial activity that provide a better understanding of rumen ecosystems. The effect of differences in fermentation pattern among microbial types on rumen fermentation profile is not incorporated in any of the models evaluated in our study. As such, the evolving availability of rumen microbial community data is encouraging (Wright and Klieve 2011) and could provide additional information on rumen microbial metabolism and subsequent impact on rumen fermentation. The emerging new DNA sequencing technology, for example, will further provide details regarding rumen microbes as it provides abetter understanding of their physiology, metabolism and overall role in the rumen ecosystem (Buddle et al. 2011). Furthermore, genome sequencing is particularly useful for rumen methanogens as they are difficult to culture. Overall, it will improve our knowledge of proportional redistribution of organic matter by rumen fermentation into VFA, microbial matter and CH_4. Generally, as Bannink et al. (2011) described omics-techniques (genomics, transcriptomics, proteomics, metabolomics) have to resulted in a paradigm shift in mathematical modelling of physiological functions.

Despite the reported variations in absorption rates among VFAs in the rumen (Dijkstra et al. 1993; Lopez et al. 2003), identical fractional rate of absorption for all types of VFA was assumed in the models evaluated in our study. With the intent of improving the assumptions associated with VFA absorption, Storm et al. (2011) reported a positive correlation between rumen epithelial blood flow and absorption kinetics of VFA from the rumen and recommended the inclusion of rumen epithelial blood flow in stoichiometric rumen VFA models. Incorporation of additional concepts, theories and parameters into the VFA stoichiometric models could improve the prediction accuracy of the models. However, as the number of parameters in the model increase, so does the model complexity, including the requirement for detailed input data

which are currently limiting. For example, integrating the stoichiometric VFA models evaluated in our study with the mechanistic model for absorption and intra-epithlial metabolism of VFA (Bannink et al. 2011) could improve the assumption of identical fractional rate of absorption and, as such, the prediction accuracy of the models. Although attempts have been made to implement mechanistic approaches to estimate VFA production in the rumen, their applicability in the current whole-rumen models is still limited due to lack of sufficient detailed data regarding rumen fermentation and metabolism.

7.1.2. Estimation of enteric methane production

Accurate prediction of enteric CH_4 production in the rumen requires accurate representation of VFA stoichiometry. Enteric CH_4 production has a direct relationship with VFA dynamics in the rumen (McAllister et al. 1996; Moss et al. 2000) as the H_2 balance is influenced by VFA stoichiometry (Ellis et al. 2008). In our study, for stoichiometric VFA models with better VFA prediction accuracy, BAN (RMSPE ranging from 5.5 to 10.7%) and NOZ (RMSPE ranging from 5.2 to 14.7%), the associated CH_4 production values had better agreement with measured values (R^2 = 0.36 to 0.51). The methanogenesis model, used to estimate enteric CH_4 production, has been developed based on the net amount of H_2 produced through microbial fermentation in the rumen assuming that formation of CH_4 as a final sink of excess H_2 production (Mills et al. 2001). As such, in addition to the inaccuracies of VFA stoichiometric models in predicting VFA production, the assumption upon which the methanogenesis model was developed on could also have contributed to the lack of agreement between the measured and model predicted CH_4 production values. Compounds other than CH_4, such as nitrates and sulphates, also serve as a H_2 sink (Ellis et al. 2008) and are unaccounted for in the model. Therefore, in order to improve the prediction accuracy of enteric CH_4 production, methanogenesis models need to incorporate all

parameters that regulate the H_2 balance in the rumen.

7.2. Choice of Model and Enteric Methane Inventory (Mechanistic vs. Empirical)

Enteric CH_4 production is estimated using either empirical or mechanistic whole-rumen models. Empirical models relate animal and/or dietary factors to CH_4 output directly whereas, mechanistic whole-rumen models address the key processes in the rumen fermentation, such as degradation and outflows of substrates, growth and outflow of rumen micro-organisms and profile of VFA formed. The profile of VFA formed in the rumen, for mechanistic models, was assessed using the stoichiometric VFA models. In our study, COWPOLL and MOLLY models used the BAN and MUR stoichiometric VFA models, respectively, to simulate VFA dynamics. Evaluation of the mechanistic whole-rumen models for their prediction accuracy of VFA and other parameters indicated that the models performed poorly when predicting VFA profile, however, other parameters such as duodenal flow of neutral detergent fiber and total non-ammonia nitrogen were predicted satisfactorily (Neal et al. 1992; Offner and Sauvant 2004). Given the poor prediction performance of the stoichiometric VFA models observed in our study, poor performance of mechanistic whole-rumen models in estimating VFA prediction profile was not surprising.

For estimated enteric CH_4 production, differences (7 to 63%) were observed among the mechanistic whole-rumen models (COWPOLL and MOLLY) and empirical models (Ellis et al. (2009), Ellis and IPCC Tier 2). The observed differences among the models have an implication on management and/or policy if the model estimated values are used in management and/or policy making decisions. As such, in the process of finding a better enteric CH_4 production estimator, empirical (linear and/or non-linear) and mechanistic whole-rumen models have been evaluated using independent data (Benchaar et al. 1998; Kebreab et al. 2008). The results

indicated better estimates of enteric CH_4 production using mechanistic whole-rumen models (*i.e.*, COWPOLL and MOLLY) compared to empirical models (*i.e.*, IPCC Tier 2). It has been reported that dynamic mechanistic whole-rumen models explained more variations (over 70%) than empirical models (42 to 57%) (Banchaar et al. 1998). Even though model evaluation was not conducted in our study, model estimated CH_4 conversion rate values (Y_m, % GEI) were compared with the values from Canadian research studies. Overall, the calculated average deviation from the Canadian research studies were smaller for the COWPOLL and MOLLY models compared to the IPCC Tier 2 and Ellis models. Similarly, using the same models, Legesse et al. (2011) estimated enteric CH_4 production from cow-calf operation and reported higher differences (26 to 35%) among the model estimated average enteric CH_4 emissions values. They also indicated that emissions estimates were lower for Ellis and IPCC Tier 2 models compared to COWPOLL and MOLLY models. As such, even though the use of models depend on the objective (*e.g.*, inventory) and the available data, the observed differences among the models in estimating enteric CH_4 production demonstrated that caution needs to be taken when selecting models.

In addition to their prediction accuracy, model selection criteria for estimating enteric CH_4 production and/or reporting enteric CH_4 inventory need to incorporate the practical applicability and flexibility of the models. For practical applications, such as rapid diet evaluation or large scale inventory purposes, the empirical models tend to be well suited as they provide quick solutions based on very limited information which may be a challenge for mechanistic rumen models. To date, provincial and national enteric CH_4 inventory from livestock production in Canada are reported using the empirical approach of IPCC Tier 1 and 2 that implement constant emission factors (EF). However, enteric CH_4 inventory from Manitoba beef cattle from 1990 to

2008 in our study indicated that average emissions estimates by IPCC Tier 2 and Ellis models were lower (0.4 to 0.8 Mt CO_2 eq) compared to estimates by COWPOLL and MOLLY models. These differences among model inventory estimates may also reflect the uncertainties in the current Canadian regional and/or national enteric CH_4 inventory estimates.

Given the various factors affecting enteric CH_4 production, such as variations in diet composition and dietary characteristics as well as animal types (*i.e.*, DM intake, productivity), the use of constant EF by the IPCC Tier 2 model to estimate enteric CH_4 production from beef operations could affect emissions estimates. Karimi-zindashty et al. (2012) conducted a sensitivity analysis using data from Canadian beef cattle and demonstrated that IPCC default parameters, such as Y_m, EF and coefficients for calculating net energy for maintenance are the greatest sources of uncertainty in implementing IPCC Tier 2 methodology to conduct enteric CH_4 inventory. Environment Canada (2011) reported that enteric CH_4 EF-related uncertainty for beef cattle ranges from 8 to 17%. Moreover, for linear and/or non-linear equations (*e.g.*, Ellis), extrapolating the models beyond the data range and/or production system upon which the models are developed, further decrease reliability in predicting CH_4 emissions (Mills 2008). The Ellis model used in our study, for example, is developed based on the data collected from growing and feedlot beef cattle. As such, using the model to estimate enteric CH_4 emissions from the 29 beef cattle categories in our study may be the main reason for the lower emissions estimates compared to the other models, IPCC Tier 2, COWPOLL and MOLLY. Conversely, mechanistic whole-rumen models (e.g., COWPOLL, MOLLY) address the key processes in the rumen fermentation and therefore, provide an opportunity to incorporate the aforementioned factors that affect enteric CH_4 production. In addition, mechanistic whole-rumen models also provide opportunities to evaluate effective emission mitigation strategies. For example, dynamic

mechanistic whole-rumen models have been used to identify effective nutritional strategies that will serve to decrease emissions of both enteric CH_4 and N pollutants (Benchaar et al. 2001; Bannink et al. 2010; Ellis et al. 2012). Ellis et al. (2012) conducted a simulation study using the COWPOLL model and demonstrated that the use of high-sugar grasses to increase water soluble carbohydrate content in the dietary DM of dairy cows diet, at the expense of crude protein in the diet, minimized N excretion but increased enteric CH_4 production. Such a tool can, therefore, form an integrated system to help producers and policy makers to manage GHG emissions from animal agriculture, thereby increase environmental sustainability.

Attempts to upscale mechanistic whole-rumen model predictions for regional and/or national enteric CH_4 inventory have been very limited as a consequence of the need for detail inputs by the models and limited available information. However, the IPCC methodology recommends the use of Tier 3 approach which includes sophisticated models that consider detailed dietary information and seasonal variability for improved estimation of enteric CH_4 inventory (IPCC 2006). Countries, such as the U.S. (U.S. EPA 2011) and The Netherlands (Bannink et al. 2011) are implementing the IPCC Tier 3 approach to conduct enteric CH_4 inventory from dairy and beef cattle production by implementing mechanistic rumen models. Enteric CH_4 inventory from 1990 to 2008 for Manitoba beef cattle, using mechanistic whole-rumen models (COWPOLL and MOLLY), was conducted in our study, demonstrating the potential use of IPCC Tier 3 methodology in Canadian regional and/or national enteric CH_4 emissions quantification protocol in order to improve enteric CH_4 inventory. Further effort may be required in developing whole-rumen models as well as obtaining detailed animal and dietary input information at the regional and/or national level.

7.3. Holistic Approach of Farm Greenhouse Gas Assessment

In recent years there has been an increased interest in developing whole-farm models to assess total farm GHG emissions and simulate the impact of various management and/or nutritional strategies on GHG emissions. Previously, use of a sectoral approach of quantifying GHG emissions as well as developing models based on the individual components in the production system has limited ability to investigate total farm GHG emissions and draw general conclusions about the production system. Accordingly, we developed a process-based, whole-farm model using different farm component models (COWPOLL, manure-DNDC and some aspects of IPCC) to assess total farm GHG emissions from Canadian cow-calf operation, which is the major contributor of total GHG emissions (80% of total emissions) within the Canadian beef production system (Beauchemin et al. 2010). This could serve as the first step towards developing an integrated process-based, whole-farm model based on Canadian production systems for accurate assessment of farm GHG emissions. Beef production system has the highest carbon foot print compared to other production systems such as dairy, swine and poultry (De Vries and De Boer 2010; Crosson et al. 2011). For Canadian beef production system, protein-based GHG intensity is four times higher for beef compared to milk production (Dyer et al. 2010). As such, given its significant carbon footprint, accurate estimation of total farm GHG emissions from beef production systems is important especially for countries like Canada where beef production plays a significant role in the agricultural sector the government is committed to GHG reduction.

In addition to estimating total farm GHG emissions, whole-farm modelling provides an opportunity to identify the relative proportional contribution of GHG emissions for the different farm components in a production cycle. This helps to direct the mitigation effort towards those

components which contribute the greatest emissions. In our study, cow-calf operations that do not apply liquid hog manure on forage land (baseline scenario) enteric CH_4 was the major contributor (63 to 76%) of the total farm GHG emissions and therefore, mitigation strategies to reduce total farm GHG emissions need to be directed towards reduction of enteric CH_4 production. Conversely, for split and single scenarios, mitigation strategies need to target both enteric CH_4 as well as soil N_2O emissions as they were the major contributors. Using life cycle assessment for Canadian beef production system, by applying Holos model to estimate farm GHG estimations, Beauchemin et al. (2010; 2011) identified enteric CH_4 as the largest contributing sources of farm GHG (63% of total emissions). Subsequently, by targeting enteric CH_4 production, the authors applied various management and nutritional strategies to reduce enteric CH_4 emissions from cow-calf operation and reported a 17% reduction in total farm GHG emissions (assuming effects were additive).

The necessity for whole-farm analysis of livestock manure recycling in the production system has been reported (Petersen et al. 2007). Moreover, application of manure as a substitute for synthetic fertilizer has been proposed as the potential management practices to reduce the carbon foot print of livestock products (Hermansen and Kristensen 2011). Evaluation of these management strategies could be worthwhile in provinces like Manitoba where approximately 80% of the pastureland is unimproved (native land, Small and McCaughey 1999; Manitoba Agriculture Review 2002). The farm GHG intensity estimates for the time and amount of liquid hog manure applied on forage land in our study were inconsistent between Integrated Components Model (ICM) and Integrated Farm System Model (IFSM). However, total farm GHG emissions estimates were higher (32 to 66%) for the split and single scenarios compared to the baseline scenario. This may indicate that GHG emissions associated with manure recycling in

production systems need to be minimized by applying appropriate measures during manure storage and/or during and after land application in order to reduce total farm GHG emissions. These measures may include management strategies ranging from reduction of the amount of N applied to land in manure (*e.g.*, dietary measures, manure treatments) to strategies that are used to minimize losses once manure has been land applied (*e.g.*, method, rate and time of application; VanderZaag et al. 2011).

The whole-farm models used in our study (ICM and IFSM) were process-based, thereby providing opportunities to investigate the various management strategies used to minimize GHG emissions associated with manure recycling in the production system. The COWPOLL model of ICM, for example, can be used to assess the dietary measures to minimize N excretion (Dijkstra et al. 2011; Ellis et al. 2012). Application methods as well as rate, amount and time of manure application can be assessed by the manure-DNDC model that include two methods of manure application, surface spreading and incorporation. For the cow-calf operation in our study, manure was surface applied using drag hose and splash-plate tanker which may increase volatilization loss of N. Using IFSM, Rotz et al. (2011) investigated various manure application techniques on NH_3 volatilization loss and reported 48% reduction for manure applied using shallow disk injection in beef production system compared to broadcast spreading without incorporation. As such, incorporation of manure during or after field application could be used as a managmnet tool to minimize total farm GHG emissions from production systems that recycle manure.

Despite their flexibility in simulating different production systems and GHG mitigation options, the use of process-based whole-farm models has its own challenges. The models require extensive detailed information about the farm components which may not be available at farm level. Further, the models have been developed based on specific production system or set of

parameters which may not apply for all farm scenarios. As such, adapting the model to another production system may require modification of the algorithms used in the model. For example, the IFSM model is based on the U.S. beef production systems and as such, the model algorithms for animal production and management as well as forage production and quality were modified based on Canadian production system in our study. Some of the challenges while simulating the IFSM were: (i) it is impossible to lower the amount of purchased forages to zero because the model was designed to import forage during the winter months to account for the perceived forage deficit that does not always exist. In our study, the quantity of forage produced as hay for winter feeding was adjusted using the size of land assigned for spring, summer and fall grazing periods because the model considers any excess forage from grazing is harvested and stored for winter feeding, and (ii) the model does not provide an opportunity to assign values for the quality of hay produced. We used dates of harvest to adjust the stage of maturity and as such, the quality of forage harvested for hay production.

Overall, given the complexity of agroecology there is a need for a joint effort among researchers and modellers from different disciplines (*e.g.*, animals, plants, soil, economics, mathematics, biology) in order to fully understand the system and develop an integrated model for holistic GHG emissions analysis. The ICM used in our study, requires further effort to integrate the different component models using common programming language so that the nutrient flow within the system as well as the ease use of the model will be improved. Moreover, beyond GHG emissions, assessment of a given production systems need to incorporate the other potential benefits of the systems, such as ecosystem services (*e.g.*, biodiversity and wildlife habitat, Power 2010) and carbon sequestration. Otherwise we might make erroneous decisions regarding mitigation strategies for the various production systems. Conversely, even though the

evolving computing capacity may provide opportunities to development detailed process-based holistic models, incorporating detail information of the various components within the production systems increase the models` complexity and hinder their applicability. However, classifying models for research, which are more complex (*e.g.*, MELODIE, Chardon et al. 2012), and those for practical application as a decision support system (*e.g.*, Holos, Little et al. 2008) may solve this problem.

8. CONCLUSIONS AND FUTURE STUDIES

The following conclusions can be drawn from the thesis:

1. Prediction accuracy of current rumen VFA models (MAR, BAN, SAV and NOZ) was low and the correlation between predicted and measured molar proportions of VFA was poor.
2. Molar proportion of VFA was better predicted by BAN (RMSPE between 5.5 and 13.3%) and NOZ (RMSPE between 5.2 and 14.7%) models compared to MUR and SAV models.
3. Enteric CH_4 production was better predicted using BAN model than MUR, SVA and NOZ.
4. Large differences were observed among model-estimated enteric CH_4 production from beef cattle using dynamic mechanistic (MOLLY and COWPOLL) and empirical models (IPCC Tier 2 and Ellis).
5. Differences were observed between methane conversion factor (Y_m, % gross energy intake) values estimated using MOLLY, COWPOLL, IPCC Tier 2 and Ellis models and the average values reported by Canadian research studies.
6. Application of liquid hog manure on cropland in cow-calf production system increased the total farm GHG intensity by 32 to 66% relative to the baseline management scenario as a consequence of an increase in the relative contribution of soil N_2O emissions.
7. Enteric CH_4 emissions was the major contributor of the total farm GHG emissions for the baseline scenario whereas soil N_2O and enteric CH_4 were the major contributors of the total farm GHG emissions for the split and single scenarios.

8. The types of whole-farm models used to analyze farm GHG emissions affects the estimated total farm GHG intensity and estimates of the relative contribution of the different GHG sources in the production system due to the difference in approaches, assumptions and algorithms.

FUTURE STUDIES

1. Development of Volatile Fatty Acid Stoichiometric Coefficients Using Beef Cattle Diet

The existing stoichiometric coefficients are developed using data collected from dairy cows (Bannink et al. 2006; 2008; Sveinbjörnsson et al. 2006), sheep and beef cattle (Murphy et al. 1982) as well as ovine, growing buffalo and cows (Nozière et al. 2011). Universal application of the existing VFA coefficients for different animal categories that are using different types of diet could be one of the potential sources of error as rumen VFA production profile varies with diet composition and animal type. Colucci et al. (1984) and De Boer et al. (1984) reported that rumen digestion kinetics of small ruminant differs from cattle, especially when they are feeding forage. Sharp et al. (1982) and Sutton (1980, 1985) indicated that production of VFA and their relative concentrations has close relationship to forage diets than concentrate. Therefore, for beef cattle, fed high forage diets, VFA coefficients developed using stoichiometric models based on VFA concentration may have a better representation.

2. Inclusion of the Effect of Cold Winter Temperature on Enteric Methane Production in Mechanistic Whole-rumen Models

Outdoor winter feeding is becoming the common winter management practices in western Canada beef cattle operations (McCartney 2011; AAFC 2011). Enteric CH_4 emission from rumen fermentation is affected by the temperature within which the animals are managed (Bernier et al. 2012). Studies revealed that animals managed under cold temperature have lower enteric CH_4 emissions (Takahashi et al. 2002; Bernier et al. 2012) which could be attributed to rumen digestion kinetics linked to digesta passage rate and VFA production profile. Moreover, enteric CH_4 production has an inverse relationship ($r = -0.53$) with the rate of digesta passage

(Okine et al. 1989). In sheep, Kennedy and Milligan (1978) reported that fermentation activity in the rumen at constant feed intake was decreased by about 1/3 as a result of cold exposure due to faster rumen digesta passage. Therefore, application of the current mechanistic whole-rumen models, without inclusion of a temperature factor, to estimate enteric CH_4 emissions from beef cattle in western Canada where animals are exposed to ambient temperature of about -20^0C during the winter period could overestimate enteric CH_4 production.

3. Implementation of the IPCC Tier 3 Approach for Enteric Methane Inventory from Canadian Beef and Dairy Production

Regional and national inventory of enteric CH_4 emissions from beef and dairy cattle in Canada is conducted using the IPCC Tier 2 methodology that implements constant EF values. However, given the major contribution of enteric CH_4 emissions from beef and dairy cattle (~33% of the total agricultural GHG emissions, Environment Canada 2011) and the variation of EF with variables such as feed intake, diet composition, and animal category, accurate estimation of enteric CH_4 emissions is paramount. As such, the IPCC Tier 3 methodology should be incorporated in into the Canadian enteric CH_4 inventory protocol. The IPCC Tier 3 methodology makes the use of local data from experiments, monitoring and validated calculation methods by incorporating mechanistic modelling. This could be achieved either by modifying the extant mechanistic rumen models for Canadian production (*e.g.*, COWPOLL) or by developing new models. Recently, Bannink et al. (2011) described the implementation of IPCC Tier 3 methodology in the Dutch protocol of inventory of enteric CH_4 by using a mechanistic rumen model (Dijkstra et al. 1992; Mills et al. 2001) that considers microbial rumen fermentation processes.

4. Assessment of the Impacts of Greenhouse Gas Mitigation Options Using Whole-farm

Modelling

One of the advantages of whole-farm modelling is that it provides an opportunity to test the impact of different mitigation strategies on total farm GHG emissions and provide answer for the "*what-if*" questions. Strategies influencing enteric CH_4 production (Beauchemin et al. 2011) and ammonia nitrogen volatilization loss following slurry application (Rotz et al. 2011b) have been evaluated using whole-farm models. Given the higher contribution of enteric CH_4 and soil N_2O for the production system that utilized liquid hog manure as N source (chapter 6), management strategies that target enteric CH_4 and soil N_2O (individually and/or in combination) need to be explored using whole-farm models. Some of the strategies may include different application techniques, time and amount of manure application (Rotz et al. 2011b; VanderZaag et al. 2011) as well as management strategies influencing enteric CH_4 emissions, such as dietary manipulation (*e.g.*, supplementation of polyunsaturated lipid, feeding dry distillers grain) and improved animal husbandry (*e.g.*, reproductive performance, stocker management, winter feeding management; Beauchemin et al. 2011). Moreover, it is also valuable to compare farm GHG emissions estimates between production systems that recycle livestock manure as N source with those systems that apply synthetic fertilizer.

5. Economic Analysis for the Simulated Mitigation Strategies

In the process of evaluating various management strategies to develop best management practices the strategies must consider economic benefit with acceptable environmental impact. This will influence the adoption and implementation of the developed mitigation technologies by farmers. As such, whole-farm models (*e.g.*, Holos, ICM) need to incorporate economic parameters in order to evaluate the economic consequences of management decisions utilized in different production systems.

6. Integration of the Component Models into a Single Program and Incorporation of other Farm Components

The farm component models that were used in the ICM (COWPOLL, manure-DNDC, some aspects of IPCC) needs to be written using single programming language in order to facilitate its ease of use and minimize the cumulative error. This may require collaborative effort of different researchers and modelers including the model founders. Moreover, some parameters (*e.g.*, manure management, animal category) in the models need to be modified to accommodate the different management practices utilized in beef cattle production systems.

In addition to GHG analysis, evaluation of the sustainability of a production system requires incorporation of the other components of the system that influence the sustainability of the system (*e.g.*, carbon sequestration, ecosystem services including biodiversity and wildlife habitat). Therefore, whole-farm models need to be extended beyond GHG analysis and incorporate those components to tell the whole story of the production system.

9. REFERENCES

Agriculture and Agri Food Canada (AAFC). 2011. Sustainable management on nutrients on the landscape for in-field livestock winter feeding system. [Online] Available: http://www4.agr.gc.ca/AAFC-AAC/display-afficher.do?id=1294669298001&lang=eng. [2012 May 10].

Abdalla, M., Jones, M., Yeluripati, J., Smith, P., Burke, J. and Williams, M. 2010. Testing DayCent and DNDC model simulations of N_2O fluxes and assessing the impacts of climate change on the gas flux and biomass production from a humid pasture. Atmosp. Environ. **44**: 2961-2970.

Abdalla, A. L., Sutton, D. J., Phipps, R. H. and Humphries, D. J. 1999. Digestion in the rumen of lactating dairy cows given mixtures of urea-treated whole-crop wheat and grass silage. Anim. Sci. **69:** 203-212.

Adams, B.W., Ehlert, G., Stone, C., Lawrence, D., Alexander, M., Willoughby, M., Hincz, C., Moisey, D., Burkinshaw, A., Carlson, J. and France, K. 2009. Rangeland health assessment for grassland, forest and tame pasture. Alberta Sustainable Resource Development Lands, Division Rangeland Management Branch, Alberta, Pub. No. T/044, 152 pp. [Online] Available: http://www.srd.alberta.ca/LandsForests/GrazingRangeManagement/documents/RangelandHealthAssessmentforGrasslandForestTamePasture-Revised-Apr2009.pdf. [2012 January 20].

Agriculture and Agri-Food Canada (AAFC). 1990-2008. Annual livestock and meat report. [Online] Available: http://www.agr.gc.ca/redmeat/almrcalendar.htm. [2009 August 2].

Alberta Agriculture, Food and Rural Development (AAFRD). 2003. CowBytes©, beef cattle ration balancer (V. 4. 6 .8), Agriculture and Rural Development, Alberta, Canada.

Alemu, A. W., Dijkstra, J., Bannink, A., France, J. and Kebreab, E. 2011. Rumen stoichiometric models and their contribution and challenges in predicting enteric methane production. Anim. Feed Sci. Technol. **166-167**: 761-778.

Alemu, A. W., Ominski, K. H. and Kebreab, E. 2011. Estimation of enteric methane emissions trends (1990-2008) from Manitoba beef cattle using empirical and mechanistic models. Can. J. Anim. Sci. **91**: 305-321.

Allen, A., Jarvis, S. and Headon, D. 1996. Nitrous oxide emissions from soils due to inputs of nitrogen from excreta return by livestock on grazed grassland in the UK. Soil Biol. Biochem. **28**: 597–607.

Alexandratos, N. 2009. World food and agriculture to 2030/50. Highlights and views from mid-2009. FAO expert meeting on how to feed the world in 2050. [Online] Available: ftp://ftp.fao.org/docrep/fao/012/ak969e/ak969e00.pdf. [2012 July 05].

Amon, B., Kryvoruchko, V., Amon, T. and Zechmeister-Boltenstern, S. 2006. Methane, nitrous oxide and ammonia emissions during storage and after application of dairy cattle slurry and influence of slurry treatment. Agric. Ecosyst. Environ. **112**: 153–162.

Amon, B., Amon, T., Boxberger, J. and Alt, C. 2001. Emissions of NH_3, N_2O and CH_4 from dairy cows housed in a farmyard manure tying stall (housing, manure storage, manure spreading). Nutr. Cycling Agroecos. **60**: 103-113.

Annison, E. F., Bickerstaffe, R. and Linzell, J. L. 1974. Glucose and fatty acid metabolism in cows producing milk of low fat content. J Agri. Sci. (Camb.), **82:** 87-95.

Archibeque, S., Haugen-Kozyra, K., Johnson, K., Kebreab, E. and Powers-Schilling, W. 2012. Near-Term Options for Reducing Greenhouse Gas Emissions from Livestock Systems in the United States beef, dairy, and swine production systems. Report, Page 82 *in* L. P. Olander and A. Van de Bogert, eds. Nicholas Institute for Environmental Policy Solutions, July 2012.

Archimède, H. Sauvant, D. and Schmidely. P. 1997. Quantitative review of ruminal and total tract digestion of mixed diet organic matter and carbohydrates. Reprod. Nutr. Dev. **37**, 183-189.

Argyle, J. L. and Baldwin, R. L. 1988. Modelling of rumen water kinetics and effects of rumen pH changes. J. Dairy Sci. **71**: 1178–1188.

ASABE. 2005. Manure production and characteristics. ASABE Standard D384.2 MAR2005. American Society of Agricultural and Biological Engineers. St. Joseph, MI.

Aschenbach, J. R., Penner, G. B. Stumpff, F. and Gäbel, G. 2011. Role of fermentation acid absorption in the regulation of ruminal pH. J. Anim. Sci. **89**: 1092-1107.

Asgedom, H. and Kebreab, E. 2011. Beneficial management practices and mitigation of greenhouse gas emissions in the agriculture of the Canadian Prairie: a review. Agron. Sustainable Dev. **31**:433–451.

Asman, W. A. H. 1998. Factors influencing local dry deposition of gasses with special reference to ammonia. Atmos. Environ. **32**:415-421.

Axelsson, J. 1949. The amount of produced methane energy in the European metabolic experiments with adult cattle. Ann. R. Agric. Coll. Sweden **16:** 404–419.

Baldwin, R. L. 1995. Modeling ruminant digestion and metabolism. Chapman & Hall, London, UK. 578 pp.

Baldwin, R. L., Thornley, J. H. M and Beever, D. E. 1987. Metabolism of the lactating cow. II. Digestive elements of a mechanistic model. J. Dairy Res. **54**: 107–131.

Ball B. C., Dobbie K. E., Parker J. P. and Smith K. A. 1997. The influence of gas transport and porosity on methane oxidation in soils. J. Geophys. Res. Atmos. **102**: 23301–23308.

Bannink, A., De Visser, H., Klop, A., Dijkstra, J. and France, J. 1997a. Causes of inaccurate prediction of volatile fatty acids by simulation models of rumen function in lactating cows. J.

Theor. Biol. **189**: 353–366.

Bannink, A., De Visser, H. and van Vuuren, A. M. 1997b. Comparison and evaluation of mechanistic rumen models. Br. J. Nutr. **78**: 563–581.

Bannink, A., France, J., López, S., Gerrits, W. J. J., Kebreab, E., Tamminga, S. and Dijkstra, J. 2008. Modelling the implication of feeding strategy on rumen fermentation and functioning of the rumen wall. Anim. Feed Sci. Technol. **143**: 3-26.

Bannink, A., Kogut, J., Dijkstra, J., France, J., Kebreab, E., van Vuuren, A. M. and Tamminga, S. 2006. Estimation of the stoichiometry of volatile fatty acid production in the rumen of lactating cows. J. Theor. Biol. **238**: 36–51.

Bannink, A., Kogut, J., Dijkstra, J., France, J., Tamminga, S. and Van Vuuren, A. M. 2000. Modeling production and portal appearance of volatile fatty acids in dairy cows. Pages 87-102 *in* J. P. McNamara, J. France and D. E. Beever, eds. Modelling nutrient utilization in farm animals. CABI publishing, Wallingford, UK.

Bannink, A., Smits, M. C. J., Kebreab, E., Mills, J. A. N., Ellis, J. L., Klop, A., France, J. and Dijkstra, J. 2010. Simulating the effects of grassland management and grass ensiling on methane emission from lactating cows. J. Agric. Sci. **148**: 55–72

Bannink, A., van Schijndel, M. W. and Dijkstra, J. 2011. A model of enteric fermentation in dairy cows to estimate methane emission for the Dutch National Inventory Report using the IPCC Tier 3 approach. Anim. Feed Sci. Technol. **166-167**: 603-618.

Barcroft, J., McAnally, R. A. and Phillipson, A. T. 1944. Absorption of volatile fatty acids from the alimentary tract of sheep and other animals. J. Exp. Biol. **20**: 120–129.

Basarab, J., Baron, V., López-Campos, Ó., Aalhus, J., Haugen-Kozyra, K. and Okine, E. 2012. Greenhouse gas emissions from calf- and yearling-fed beef production systems, with and without the use of growth promotants. Anim. **2**: 195-220.

Basarab, J. A., Gould, S. R. and Weisenburger, R. D. 1984. Growth response of beef cattle at pasture to Zeranol or Progesterone-estradiol implants. Can. J. Anim. Sci. **64**: 119-126.

Basarab, J. A., Okine, E. K., Baron, V. S., Marx, T., Ramsey, P., Ziegler, K. and Lyle, K. 2005. Methane emission from enteric fermentation in Alberta`s beef cattle population. Can. J. Anim. Sci. **85**: 501-512.

Basset-Mens, C., Ledgard, S. and Boyes, M. 2009. Eco-efficiency of intensification scenarios for milk production in New Zealand. Ecol. Econ. **68**: 1615–1625.

Beauchemin, K. A., Kreuzer, M., O'Mara, F. and McAllister, T. A. 2008. Nutritional management for enteric methane abetment: a review. Aust. J. Exp. Agric. **48**: 21-27.

Beauchemin, K. A., McAllister, T. A. and McGinn, S. M. 2009a. Dietary mitigation of enteric methane from cattle. CAB Reviews: Perspectives in Agriculture, Veterinary Science, Nutrition

and Natural Resources **4 (035):** 18-27.

Beauchemin, K. A. and McGinn, S. M. 2005. Methane emissions from feedlot cattle fed barley or corn diets. J. Anim. Sci. **83**: 653-661.

Beauchemin, K. A. and McGinn, S. M. 2006. Methane emissions from beef cattle: Effects of fumaric acid, essential oil, and canola oil. J Anim. Sci. **84**:1489-1496.

Beauchemin, K. A., McGinn, S. M., Benchaar, C. and Holtshausen, L. 2009b. Crushed sunflower, flax, or canola seeds in lactating dairy cow diets: effects on methane production, rumen fermentation, and milk production. J. Dairy Sci. **92**: 2118-2127.

Beauchemin, K. A., Janzen, H. H., Little, S. M., McAllister, T. A. and McGinn, S. M., 2010. Life cycle assessment of greenhouse gas emissions from beef production in western Canada: a case study. Agric. Syst. **103**: 371-379.

Beauchemin, K. A., Janzen, H. H., Little, S. M., McAllister, T. A. and McGinn S. M. 2011. Mitigation of greenhouse gas emissions from beef production in Western Canada – Evaluation using farm-based life cycle assessment. Anim. Feed Sci. Technol. **166-167**: 663-677.

Beauchemin, K. A., Yang, W. Z. and Rode, L. M. 1999. Effect of grain source and enzyme additive on the site and extent of nutrient digestion in dairy cows. J. Dairy Sci. **82**: 378-390.

Beaulieu, M. S. 2004. Manure storage in Canada. Statistics Canada. Catalogue no. 21-021-MIE. Minister of Industry, Ottawa, pp 52.

Benchaar, C., Pomar, C. and Chiquette, J. 2001. Evaluation of dietary strategies to reduce methane production in ruminants: A modelling approach. Can. J. Anim. Sci. **81**:563–574.

Benchaar, C., Rivest, J. Pomar, C. and Chiquette. J. 1998. Prediction of methane production from dairy cows using existing mechanistic models and regression equations. J. Anim. Sci. **76**: 617–627.

Berg, W., Brunsch, R. and Pazsiczki, I. 2006. Greenhouse gas emissions from covered slurry compared with uncovered during storage. Agric. Ecosyst. Environ. **112**:129-134.

Bergman, E. N. 1975. Production and utilization of metabolites by the alimentary tract as measured in portal and hepatic blood. Pages 292-305 *in* I. W., McDonald and A. C. I., Warner, eds. Digestion and metabolism in the ruminant. University of New England Publishing Unit. Armidale.

Bergman, E. N. 1990. Energy contribution of volatile fatty acids from the gastrointestinal tract in various species. Physiological reviews. **70**: 567-590.

Bergman, E. N., Reid, R. S., Murray, M. G., Brockway, J. M. and Whitelaw, F. G. 1965. Interconversions and production of volatile fatty acids in the sheep rumen. Biochem. J. **97**: 53–58.

Bernier, J. N., Undi, M., Plaizier, J. C., Wittenberg, K. M., Donohoe, G. R. and Ominski, K. H. 2012. Impact of prolonged cold exposure on dry matter intake and enteric methane 1 emissions of beef cows overwintered on low-quality forage diets with and without 2 supplemented wheat and corn dried distillers grain with solubles (DDGS). Can. J. Anim. Sci. **92**: 493_500.

Berntsen, J., Petersen, B. M., Jacobsen, B. H., Olesen, J. E. and Hutchings , N. J. 2003. Evaluating nitrogen taxation scenarios using the dynamic whole farm simulation model FASSET. Agric. Syst. **76**: 817–839.

Bibby, J. and Toutenburg, H. 1977. Prediction and improved estimation in linear models. John Wiley and Sons. London, UK.

Blaxter, K. L., and Clapperton, J. L. 1965. Prediction of the amount of methane produced by ruminants. Br. J. Nutr. **19**: 511–521.

Blonski, L. J., Bork, E. W. and Blenis, P. V. 2004. Herbage yield and crude protein concentration of rangeland and pasture following hog manure application in southeastern Alberta. Can. J. Plant Sci. **84**: 773–783.

Boadi, D. A. and Wittenberg, K. M. 2002. Methane production from dairy and beef heifers fed forages differing in nutrient density using the sulfur hexafluoride (SF_6) tracer gas technique. Can. J. Anim. Sci. **82**:201-206.

Boadi, D. A., Wittenberg, K. M. and McCaughey, W. P. 2002. Effect of grain supplementation on methane production of grazing steers using the sulphur (SF_6) tracer gas technique. Can. J. Anim. Sci. **82**: 151-157.

Boadi, D. A. Benchaar, C., Chiquette, J. and Massé, D. 2004c. Mitigation strategies to reduce enteric methane emissions from dairy cows: Update review. Can. J. Anim. Sci. **84**: 319-336.

Boadi, D. A., Ominski, K. H., Wittenberg, K. M. and Fulawka, D. L. 2004a. Improving estimates of methane emissions associated with enteric fermentation of cattle in Canada by adopting an IPCC Tier-2 Methodology. Solicitation number: K2362-3-0088. Final report submitted to Environment Canada, Oct. 20, 2004.

Boadi, D. A., Wittenberg, K. M., Scott, S. L., Burton, D., Buckley, K., Small, J. A. and Ominski, K. H. 2004b. Effect of low and high forage diet on enteric and manure pack greenhouse gas emission from a feedlot. Can. J. Anim. Sci. **84**: 445-453.

Bork, E. W. and Blonski, L. J. 2012. Short-term native grassland compositional responses following liquid hog manure application. Can. J. Plant Sci. **92**: 55-65.

Borken W. and Beese F. 2006. Methane and nitrous oxide fluxes of soils in pure and mixed stands of European beech and Norway spruce. Eur. J. Soil Sci. **57:** 617–625.

Bosch, M. W., Tamminga, S., Post, G., Leffering, C. P. and Muylaert, J. M. 1992. Influence of stage maturity of grass silages on digestion process in dairy cows. 1. Composition, nylon bag

degradation rates, fermentation characteristics, digestibility and intake. Livest. Prod. Sci. **32**: 245-264.

Bouwman, A. F., Van Vuuren, D. P., R. Derwent, G. and Posch, M. 2002. A global analysis of acidification and eutrophication of terrestrial ecosystems. Water, Air, and Soil Pollution **141**:349–382.

Bratzler, J. W. and Forbes, E. B. 1940. The estimation of methane production by cattle. J. Nutr. **19**: 611-613.

Broderick, G. A., Luchini, N. D., Reynal, S. M., Varga, G. A. and Ishler, V. A. 2008. Effect on production of replacing dietary starch with sucrose in lactating dairy cows. J. Dairy Sci. **91**: 4801-4810.

Brown, H. A., Wagner-Riddle, C. and Thurtell, G. W. 2000. Nitrous oxide flux from solid dairy manure in storage as affected by water content and redox potential. J. Environ. Qual. **29**: 630–638.

Buddle, B. M., Denis, M., Attwood, G. T., Altermann, E., Janssen, P. H., Ronimus, R. S., Pinares-Patiño, C. S., Muetzel, S. and Wedlock, D. N. 2011. Strategies to reduce methane emissions from farmed ruminants grazing on pasture. Vet. J. **188**: 11–17.

Callison, S. L., Firkins, J. L., Eastridge, M. L. and Hull, B. L. 2001. Site of nutrient digestion by dairy cows feed corn of different particle sizes or steam-rolled. J. Dairy Sci. **84**: 1458-1467.

Canfax Research Services. 2009. Trends, cycles and seasonality in the cattle industry. Calgary, Alberta, 2009.

Capper J. L. 2011. Replacing rose-tinted spectacles with a high-powered microscope: The historical vs. modern carbon footprint of animal agriculture. Anim. Front. **1**:26–32.

Carlberg, J. G. and Brewin, D. G. 2005. Managing an industry in crisis: BSE in Canada. [Online] available: http://ageconsearch.umn.edu/bitstream/35529/1/sp05ca03.pdf. [2009 Nov. 10].

Carlberg, J. G., Brewin, D. G. and Rude., J. I. 2009. Managing a border threat: BSE and COOL effects on the Canadian beef industry. Rev. Agric. Econ. **31**: 952-962.

Carroll, E. J. and Hungate, R. E. 1954. The magnitude of the microbial fermentation in the bovine rumen. Appl. Microbiol. **2**: 205-214.

Casey, J. W. and Holden, N. M. 2006a. Quantification of GHG emissions from sucker-beef production in Ireland. Agric. Syst**. 90**: 79-98.

Casey, J. W. and Holden, N. M. 2006b. Greenhouse gas emissions from conventional, agri-environmental scheme, and organic Irish suckler-beef units. J. Environ. Qual. **35**: 231–239.

Cederberg, C. and Stadig, M. 2003. System expansion and allocation in life cycle assessment of milk and beef production. Int. J. LCA. **8**: 350-356.

Cederberg, C., Persson, M., Neovius, K., Molander, S. and Clift, R. 2011. Including carbon emissions from deforestation in the carbon footprint of Brazilian beef. Environ. Sci. Technol. **45**: 1773-1779.

Chadwick, D., Pain, B. and Brookman, S. 2000. Nitrous oxide and methane emissions following application of animal manures to grassland. J. Environ. Qual. **29**: 277–287.

Chardon, X., Rigolot, C., Baratte, C., Espagno, S., Raison, C., Martin-Clouaire, R., Rellier6, J.-P., Le Gall, A., Dourmad, J. Y., Piquemal, B., Leterme, P., Paillat, J. M., Delaby, L., Garcia, F., Peyraud, J. L., Poupa, J. C., Morvan, T. and Faverdin, P. 2012. MELODIE: a whole-farm model to study the dynamics of nutrients in dairy and pig farms with crops. Anim. doi:10.1017/S1751731112000687.

Chaves, A. V., Thompson, L. C., Iwaasa, A. D., Scott, S. L., Olson, M. E., Benchaar, C., Veira, D. M. and McAllister, T. A. 2006. Effect of pasture type (alfalfa vs. grass) on methane and carbon dioxide production by yearling beef heifers. Can. J. Anim. Sci. **86**: 409-418.

Chianese, D. S., Rotz, C. A. and Richard, T. L. 2009. Simulation of carbon dioxide emissions from dairy farms to assess greenhouse gas reduction strategies. Trans. ASABE **52**:1301-1312.

Clark, H., Kelliher, F. and Pinares-Patiño, C. 2011. Reducing CH_4 emissions from grazing ruminants in New Zealand: challenges and opportunities. Asian-Aust. J. Anim. Sci. **24**: 295 – 302.

Clark, J. H., Klusmeyer, T. H. and Cameron, M. R. 1992. Microbial protein synthesis and flow of nitrogen fractions to the duodenum of dairy cows. J. Dairy Sci. **75**: 2304-2323.

Colucci, P. E. Macleod, G. K., Grovum, W. L. and McMillan, I. 1984. Comparative digestion and digesta kinetics in sheep and cattle. Can. J. Anim. Sci., **64**: 173–174.

Conrad, R. 1989. Control of methane production in terrestrial ecosystems. Pages 39–58 *in* M. O. Andreae and D. S. Schimel, eds. Exchange of trace gases between terrestrial ecosystems and the atmosphere. Biddles Ltd., Guildford, UK.

Corley, R. N. and Murphy, M. R. 2004. An in vitro technique for measuring the production rate of volatile fatty acids in the rumen under dynamic conditions. Small Rumin. Res. **54**: 219-225.

Crocker, L. M., DePeters, E. J., Fadel, J. G., Perez-Monti, H., Taylor, S. J., Wyckoff, J. A. and Zinn, R. A. 1998. Influence of processed corn grain in diets of dairy cows on digestion of nutrients and milk production. J. Dairy Sci. **81**: 2394-2407.

Crosson, P., Foley, P. A., Shalloo, L., O'Brien, D. and Kenny, D. A., 2010. Greenhouse gas emissions from Irish beef and diary production systems. Advances in animal biosciences. Food, feed, energy and fibre from land – a vision for 2020. Pages 350-380 *in*: Proceedings of the British Society of Animal Science and the Agricultural Research Forum,. Cambridge University Press, Cambridge, UK.

Crosson, P., Shalloo, L., O'Brien, D., Lanigan, G. J., Foley, P. A., Boland, T. M., and Kenny, D. A. 2011. A review of whole farm systems models of greenhouse gas emissions from beef and dairy cattle production systems. Anim. Feed Sci. Technol. **166-167**: 29-45.

Danfær, A., Huhtanen, P., Udén, P., Sveinbjörnsson, J. and Volden, H. 2006. The Nordic dairy cow model, Karoline – description. Pages 383-406, *in* E. Kebreab, J. Dijkstra, A. Bannink, W. J. J. Gerrits and J. France, eds. Nutrient digestion and utilization in farm animals: modelling approach. CAB International, Wallingford, UK.

De Boer, J. L., Cottyn, B. G., Boucque, C. V., Aerts, J. V. and Buysse, F. X. 1984. Comparative digestibility by sheep and cows and consequences on energy value. Can. J. Anim. Sci., **64**: 175–176.

De Haan, M. H. A., Schils, R. L. M., Hemmer, J. G. A., Dasselaar, A. van den ol-van, de Boer, J. A., Evers, A. G., Holshof, G. J., iddelkoop, C. van, and Zom, R. L. G. 2007. DairyWise, model documentation. Wageningen Animal Science Group. [Online] Available: http://edepot.wur.nl/35296. [2012 Apr. 26].

Del Grosso, S. J., Ojima, D. S., Parton, W. J., Mosier, A. R., Petereson, G. A. and Schimel, D. S. 2002. Simulated effects of dryland cropping intensification on soil organic matter and greenhouse gas exchanges using the DAYCENT ecosystem model. Environ. Pollut. **116**: S75-S83.

Del Grosso, S. J., Ojima, D. S., Parton, W. J., Stehfest, E., Heistemann, M., DeAgelo, B. and Rose, S. 2009. Global scale DAYCENT model analysis of greenhouse gas emissions and mitigation strategies for cropped soils. Global Planet. Change. **67**: 44–50.

Del Grosso, S. J., Parton, W. J., Mosier, A. R., Walsh, M. K., Ojima, D. S. and Thornton, P. E. 2006. DAYCENT National-Scale Simulations of Nitrous Oxide Emissions from Cropped Soils in the United States. J. Environ. Qual. **35**: 1451–1460.

Demeyer, D. 1991. Quantitative aspects of microbial metabolism in the rumen and hind gut. Pages 217-237 *in* J. P. Jouany, ed. Rumen microbial metabolism and ruminant digestion. NRIA Editions, Paris, France.

Demeyer, D. and Van Nevel, C. J. 1975. Methanogenesis, an integrated part of carbohydrate fermentation, and its control. Pages 363-383 *in* I. W. McDonald and A. C. I. Warner, eds. Digestion and metabolism in the ruminant. The University of New England Publishing Unit, Armidale, NSW, Australia.

Denman, K.L., Brasseur, G., Chidthaisong, A., Ciais, P., Cox, P. M., Dickinson, R. E., Hauglustaine, D., Heinze, C., Holland, E., Jacob, D., Lohmann, U., Ramachandran, S., da Silva Dias, P. L., Wofsy, S. C. and Zhang, X., 2007. Couplings between changes in the climate system and biogeochemistry. Pages 499-587 *in*: S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K. B. Averyt, M. Tignor, H. L. Miller, eds. Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge, UK and

New York, NY, USA.

DeRamus, H. A., Clement, T. C., Giampola, D. D. and Dickison, P. C. 2003. Methane emissions of beef cattle on forages: Efficiency of grazing management systems. J. Environ. Qual. **32**: 269-277.

De Visser, H., Klop, A., Van der Koelen, C. J. and Van Vuuren, A. M. 1998a. Starch supplementation of grass harvested at two stages of maturity prior to ensiling: intake, digestion, and degradability by dairy cows. J. Dairy Sci. **81**: 2221–2227.

De Visser, H., Klop, A., Van der Meulen, J. and Van Vuuren, A. M. 1998b. Influence of maturity of grass silage and flaked corn starch on the production and metabolism of volatile fatty acids in dairy cows. J. Dairy Sci. **81**: 1028–1035.

Dijkstra, J. 1994. Production and absorption of volatile fatty acids in the rumen. Livest. Prod. Sci. **39**: 61-69.

Dijkstra J., Boer, H., Van Bruchem, J., Bruining, M. and Tamminga, S. 1993. Absorption of volatile fatty acids from the rumen of lactating dairy cows as influenced by volatile fatty acid concentration, pH and rumen liquid volume. Br. J. Nutr. **69**: 385-96.

Dijkstra, J., Forbes, J. M. and France, J. 2005. Quantitative aspects of ruminant digestion and metabolism, second edition. CABI Publishing, Wallingford, UK. 734 pp.

Dijkstra, J., Kebreab, E., France, J. and Bannink, A. 2008. Modelling protozoal metabolism and volatile fatty acid production in the rumen. Pages 170-188 *in*: J. France and E. Kebreab, eds. Mathematical modelling in animal nutrition. CABI International, Wallingford, UK.

Dijkstra, J., Kebreab, E., Mills, J. A. N., Pellikaan, W. F., Lopez, S., Bannink, A. and France, J. 2007. Predicting the profile of nutrients available for absorption: from nutrient requirement to animal response and environmental impact. Anim. **1**: 99-111.

Dijkstra, J., Neal, H. D., Beever, D. E. and France, J. 1992. Simulation of nutrient digestion, absorption and outflow in the rumen: Model description. J. Nutr. **122**: 2239_2256.

Dijkstra, J., Oenema, O. and Bannink, A. 2012. Dietary strategies to reduce nitrogen excretion from cattle: implication for methane emissions. Current Opinion in Enviro. Sust. **3**: 414-422.

Dittert, K., Lampe, C., Gasche, R., Butterbach-Bahl, K., Wachendorf, M., Papen, H., Sattelmacher, B., Taube, F. 2005. Short-term effects of single or combined application of mineral N fertilizer and cattle slurry on the fluxes of radiatively active trace gases from grassland soil. Soil Biol. Biochem. **37**: 1664–1674.

Dumas, A., Dijkstra, J. and France, J. 2008. Mathematical modelling in animal nutrition: a century review. J. Agri. Sci. **146**: 123-142.

Dyer, J. A. and Desjardins. R. L. 2003. Simulated farm fieldwork, energy consumption and related greenhouse gas emissions in Canada. Biosyst. Eng. **85**: 503-513.

Dyer, J. A. and Desjardins. R. L. 2006. An integrated index of electrical energy use in Canadian agriculture with implications for greenhouse gas emissions. Biosyst. Eng. **95**: 449-460.

Dyer, J. A., Vergé, X. P. C., Desjardins, R. L. and Worth, D. E. 2010. The protein-based GHG emission intensity for livestock products in Canada. J. Sust. Agri. **34:** 618-629.

Eckard, R. J., Grainger, C. and de Klein, C. A. M. 2010. Options for the abetment of methane and nitrous oxide from ruminant production: A review. Livest. Sci. **130**: 47-56.

Ellert, B. H. and Janzen, H. H. 2008. Nitrous oxide, carbon dioxide and methane emissions from irrigated cropping systems as influenced by legumes, manure and fertilizer. Can. J. Soil Sci. **88**: 207_217.

Ellis, J. L., Bannink, A., France, J., Kebreab, E. and Dijkstra, J. 2010. Evaluation of enteric methane prediction equations for dairy cows used in whole farm models. Glob. Change Biol. **16**: 3246-3256.

Ellis, J. L., Dijkstra, J., France, J., Parsons, A. J., Edwards, G. R., Rasmussen, S., Kebreab, E. and Bannink, A. 2012. Effect of high-sugar grasses on methane emissions simulated using a dynamic model. J. Dairy Sci. **95**: 272–285.

Ellis, J. L., Dijkstra, J., Kebreab, E., Bannink, A., Odongo, N. E., McBride, B. W. and France, J. 2008. Aspects of rumen microbiology central to mechanistic modelling of methane production in cattle. J. Agri. Sci. **146**: 213-233.

Ellis, J. L., Kebreab, E., Odongo, N. E., Beauchemin, K., McGinn, S., Nkrumah, J. D., Moore, S. S., Christopherson, R., Murdoch, G. K., McBride, B. W., Okine E. K. and France. J. 2009. Modeling methane production from beef cattle using linear and nonlinear approaches. J. Anim. Sci. **87**: 1334-1345.

Ellis, J. L., Kebreab, E., Odongo, N. E., McBride, B. W., Okine, E. K. and France, J. 2007. Prediction of methane production from dairy and beef cattle. J. Dairy Sci. **90**: 3456–3466.

Ellis, S., Webb, J., Misselbrook, T. and Chadwick, D. 2001. Emission of ammonia (NH_3), nitrous oxide (N_2O) and methane (CH_4) from a dairy hardstanding in the UK. Nutr. Cycling Agroecos. **60**:115-122.

Environment Canada. 2012a. National Climate Data and Information Archive. [Online] Available http://www.climate.weatheroffice.ec.gc.ca/climate_normals/stnselect_e.html. [2012 April 15].

Environment Canada. 2012b. Precipitation product samples. [Online] Available http://www.ec.gc.ca/natchem/default.asp?lang=En&n=B385159B-1. [2012 April 18].

Environment Canada. 2011. National Inventory Report 1990-2009 (Part 1, 2 and 3): Greenhouse gas sources and sinks in Canada. The Canadian Government`s Submission to the UN Framework Convention on Climate Change. Greenhouse Gas Division. Environment Canada, Gatineau, QC. [Online] Available http://www.ec.gc.ca/ges-ghg/. [2012 April 18].

Environment Canada. 2010. National Inventory Report 1990—2008 (Part 1, 2 and 3): Greenhouse Gas Sources and Sinks in Canada. Greenhouse gas division. Environment Canada, Gatineau, QC.

Environment Canada. 2009. Weather office. [Online] Available: www.climate.weatheroffice.ec.gc.ca. [2009 August 11].

Erickson, G. E., Auvermann, B., Eigenberg, R. A., Greene, L.W., Klopfenstein, T. J. and Koelsch, R. K. 2003. Proposed beef cattle manure excretion and characteristics standard for ASAE, in: proceedings of the Ninth International Animal, Agricultural and Food Processing Wastes. North Carolina USA, pp. 269-276.

Fahey, G. C. and Berger. L. L.1988. Carbohydrate nutrition in ruminants: pages 125-133 *in*: D. C. Church, ed. The ruminant animal: digestive physiology and nutrition. Prentice Hall: Englewood Cliffs, NJ.

Fernandez, I., Nozière, P. and Michalet-Doreau, B. 2004. Site and extent of starch digestion of whole-plant maize silages differing in maturity stage and chop length, in dairy cows. Livest. Prod. Sci. **89**: 147-157.

Finlay, B. J., Esteban, G., Clarke, K. J., Williams, A. G., Embley, T. M. and Hirt, R. P. 1994. Some rumen ciliates have endosymbiotic methanogens. FEMS Microb. Letters **117**: 157-162.

Flessa H., Dörsch, P., Beese, F., K.onig, H. and Bouwman, A. F. 1996. Influence of cattle wastes on nitrous oxide and methane fluxes in pasture land. J. Environ. Qual., **25**: 1366–1370.

Flessa, H., Ruser, R., Dörsch, P., Kamp, T., Jimenez, M. A., Munch, J. C. and Beese. F. 2002. Integrated evaluation of greenhouse gas emissions (CO_2, CH_4, N_2O) from two farming systems in southern Germany. Agri. Ecosys. Environ. **91**: 175-189.

Foley, P. A., Crossonb,P., Lovetta, D. K., Bolanda, T. M., O'Maraa, F. P. and Kennya, D. A. 2011. Whole-farm systems modelling of greenhouse gas emissions from pastoral suckler beef cow production systems. Agri. Ecosys. Environ. **142**: 222-230.

Foley, A. E., Hristov, A. N., Melgar, A., Ropp, J. K., Etter, R. P., Zaman, S., Hunt, C. W., Huber, K. and Price, W. J. 2006. Effect of barley and its amylopectin content on ruminal fermentation and nitrogen utilization in lactating dairy cows. J. Dairy Sci. **89**: 4321-4335.

Foley, J. A., Ramankutty, N., Brauman, K. A., Cassidy, E. S., Gerber, J. S., Johnston, M., Mueller, N. D., O'Connell, C., Ray, D. K., West, P. C., Balzer, C., Bennett, E. M., Carpenter, S. R., Hill, J., Monfreda, C., Polasky, S., Rockström, J., Sheehan, J., Siebert, S., Tilman, D., and Zaks, D. P. M. 2011. Solution for a cultivated planet. Nature **478**: 337-342.

France J. and Dijkstra. J. 2005. Volatile fatty acid production. Pages. 157-174 *in:* J. Dijkstra, J. M. Forbes, and J. France, eds. Quantitative aspects of ruminant digestion and metabolism. Second ed. CAB International, New York.

France, J., Gill, M., Dhanoa, M. S. and Siddons, R. C. 1987. On solving the fully-interchanging N-compartment model in steadystate tracer kinetic studies with reference to VFA absorption from the rumen. J. Theor. Biol., **125**: 193-211.

France, J. and E. Kebreab. 2008. Mathematical Modelling in Animal Nutrition. CAB International, Wallingford, UK. Pp 574.

France, J. and Siddons, R. C. 1993. Volatile fatty acid production. Pages 107-121 *in* J. M. Forbes and J. France, eds. Quantitative aspects of ruminant digestion and metabolism. CAB International, New York.

France, J. and Thornley, J. H. M. 1984. Mathematical models in agriculture: a quantitative approach to problems in agriculture and related sciences. London; Boston: Butterworths.

Friggens, N. C., Oldham, J. D., Dewhurst, D. R. and Horgan, G. 1998. Proportions of volatile fatty acids in relation to the chemical composition of feeds based on grass silage. J. Dairy Sci. **81**: 1331-1344.

Garnsworthy, P. C. 2004. The environmental impact of fertility in dairy cows: A modeling approach to predict methane and ammonia emission. Anim. Feed Sci. and Technol. **112**: 211-223.

Gerber, P. J. and Steinfeld, H. 2008. Global environmental consequences of the livestock sector`s growth. Outlook Agri. **37**: 7-13.

Gerber, P., Vellinga, T., Opio, C., Henderson, B., and Steinfeld, H. 2010. Greenhouse gas emissions from the dairy sector: A life cycle assessment. FAO Food and Agriculture Organisation of the United Nations, Animal Production and Health Division, Rome, Italy.

Giger-Reverdin, S., Duvaux-Ponter, C., Sauvant, D., Martin, O., Prado, I. N. and R. Müller. 2002. Intrinsic buffering capacity of feedstuffs. Anim. Feed Sci. Technol. **96**:83–102.

Gill, M., Smith, P. and Wilkinson, J. M. 2010. Mitigating climate change: the role of domestic livestock. Anim. **4**: 323-333.

Government of Manitoba 2008. The Climate Change and Emissions Reductions Act. 2008. C.C.S.M. c. C135. [Online] Available: http://web2.gov.mb.ca/laws/statutes/ccsm/c135e.php. [2010 August 12].

Grainger, C., Clarke, T., McGinn, S. M., Auldist, M. J., Beauchemin, K. A., Hannah, M. C., Waghorn, G. C., Clark, H. and Eckard, R. J. 2007. Methane emissions from dairy cows measured using the sulfur hexafluoride (SF_6) tracer and chamber techniques. J. Dairy Sci. **90**: 2755–2766.

Grant, R. F. and Pattey, E. 2003. Modelling variability in N_2O emissions from fertilized agricultural fields. Soil Biol. Biochem. **35:** 225-243.

Greenfield, T. L., Baldwin, R. L. and Erdman, R. A. 2001. Ruminal fermentation and

intestinal flow of nutrients by lactating cows consuming brown midrib corn silages. J. Dairy Sci. **84**: 2469-2477.

Gregorich, E. G., Rochette, P., VandenBygaart, J. A. and Angers D. A. 2005. Greenhouse gas contributions of agricultural soils and potential mitigation practices in Eastern Canada. Soil Tillage Res **83**: 53–72.

Guan, H. T., Wittenberg, K. M., Ominski, K. H. and Krause, D. O. 2006. Efficacy of ionophores in cattle diets for mitigation of enteric methane. J. Anim. Sci. **84**: 1896–1906.

Harper, L. A., Sharpe, R. R. and Parkin, T. B. 2000. Gaseous nitrogen emissions from anaerobic swine lagoons: ammonia, nitrous oxide, and dinitrogen gas. J. Environ. Qual. **29**: 1356–1365.

Hegarty, R. S. and Nolan, J. V. 2007. Estimation of ruminal methane production from measurement of volatile fatty acid production. Pages 69-103 *in* H. P. S. Makkar and P. E. Vercoe, eds. Measuring methane production from ruminants. Springer, The Netherlands.

Helgason, B. L., Janzen, H. H., Chantigny, M. H., Drury, C. F., Ellert, B. H., Gregorich, E. G., Lemke, R. L., Pattey, E., Rochette P. and Wagner-Riddle, C. 2005. Toward improved coefficients for predicting direct N_2O emissions from soil in Canadian agroecosystems. Nutr. Cycling Agroecosys. **72**: 87-99.

Hemansen, J. E. and Kristensen. T. 2011. Management options to reduce the carbon footprint of livestock products. Anim. Front. **1**: 33-39.

Herrero, M., Thornton, P. K., Gerber, P. and Reid R. S. 2009. Livestock, livelihoods and the environment: understanding the trade-offs (Review). Curr. Opin. Environ. Sustainability **2**: 111-120.

Hindrichsen, I. K., Wettstein, H. R., Machmüller, A. and Kreuzer, M. 2006. Methane emission, nutrient digestion and nitrogen turnover in dairy cows and their slurry at different milk production scenarios with and without concentrate supplementation. Agric. Ecosyst. Environ. **113**: 150-161.

Hobson, P. N. and Stewart. C. S. 1997. The rumen microbial ecosystem. Blackie Academic and Professional. Chapman and Hall, New York, NY.

Holter, J. B. and Young, A. J. 1992. Methane prediction in dry and lactating dairy cows. J. Dairy Sci. **75**: 2165-2175.

Hopkins L. A. 1985. Soils of the rural municipalities of Ste Anne and la Broquerie, and part of the local government district of stuartburn. Report No. D49. Canada-Manitoba Soil Survey, Winnipeg, MB.

Hristov, A. N., Etter, R. P., Ropp, J. K. and Grandeen, K. L. 2004. Effect of dietary crude protein level and digestibility on ruminal fermentation and nitrogen utilization in lactating dairy cows. J. Anim. Sci. **82**: 3219-3229.

Hristov, A. N. and Ropp, J. K. 2003. Effect of dietary carbohydrate composition and availability on utilization of ruminal ammonia nitrogen for milk protein synthesis in dairy cows. J. Dairy Sci. **86**: 2416-2427.

Huang, Q., Wohlgemut, O., Cicek, N., France, J. and Kebreab, E. 2010. A mechanistic model for predicting methane emissions from anaerobic manure storage systems. Can J Soil Sci **90**: 507–516.

Huhtanen, P., Nousiainen, J. and Rinne, M. 2006. Recent development in forage evaluation with special reference to practical applications. Agric. and Food Sci. **15**: 293-323.

Huhtanen, P. and Sveinbjörnsson, J. 2006. Evaluation of methods for estimating starch digestibility and digestion kinetics in ruminants. Anim. Feed Sci. Technol. **130**: 95–113.

Hume, I. D. 1977. Production of volataile fatty acids in two species of wallaby and in sheep. Comp. Biochem. Physiol. **56A**: 299-304.

Hungate, R. E. 1966. The rumen and its microbes. Academic press. New York.

Intergovernmental Panel on Climate Change (IPCC). 1995. Climate change 1995: The science of climate change. Summary for policymakers and technical summary of the working group I Report. p. 22. Cambridge University Press, Cambridge, UK. [Online] Available: http://www.ipcc.ch/ipccreports/sar/wg_I/ipcc_sar_wg_I_full_report.pdf. [2010 Dec. 5].

Intergovernmental Panel on Climate Change (IPCC), 2006. 2006. IPCC Guidelines for National Greenhouse Gas Inventories. [Online] Available: http://www.ipcc-nggip.iges.or.jp/public/2006gl/index.html. [2012 Apr. 10].

Intergovernmental Panel on Climate Change (IPCC). 2007. IPCC fourth assessment report: Climate Change 2007. Contribution of working group I to the fourth assessment report of the Intergovernmental Panel on Climate Change, 2007. Cambridge University Press, Cambridge, United Kingdom and New York, USA.

Ipharraguerre, I. R., Clark, J. H. and Freeman, D. E. 2005a. Varying protein and starch in the diet of dairy cows. I. Effects on ruminal fermentation and intestinal supply of nutrients. J. Dairy Sci. **88**: 2537-2555.

Ipharraguerre, I. R., Clark, J. H. and Freeman, D. E. 2005b. Rumen fermentation and intestinal supply of nutrients in dairy cows fed rumen-protected soy products. J. Dairy Sci. **88**: 2879-2892.

Ipharraguerre, I. R., Shabi, Z., Clark, J. H. and Freeman, D. E. 2002. Ruminal fermentation and nutrient digestion by dairy cows fed varying amounts of soyhulls as replacement for corn grain. J. Dairy Sci. **85**: 2890-2904.

ISO, 2006a. Environmental Management – Life Cycle Assessment: Principles and Framework. (ISO 14040:2006). European Committee for Standardization, Brussels, Belgium.

ISO, 2006b. Environmental Management – Life Cycle Assessment: Requirements and Guidelines. (ISO 14044:2006). European Committee for Standardization, Brussels, Belgium.

Janzen, H. H., Angers, D.A., Boehm, M., Bolinder, M., Desjardins, R. L., Dyer, J. A., Ellert, B. H., Gibb, D. J., Gregorich, E. G., Helgason, B. L., Lemke, R., Massé, D., McGinn, S. M., McAllister, T. A., Newlands, N., Pattey, E., Rochette, P., Smith, W., VandenBygaart, A. J. and Wang, H. 2006. A proposed approach to estimate and reduce net greenhouse gas emissions from whole farms. Can. J. Soil Sci. **86**: 401-418.

Janzen, H. H., Campbell, C. A., Izaurralde, R. C., Ellert, B. H., Juma, N., McGill, W. B. and Zentner, R. P. 1998. Management effects on soil C storage on the Canadian prairies. Soil Till. Res. **47**: 181-195.

Jarecki, M. K., Parkin, T. B., Chan, A. S. K., Hatfi eld, J. L. and Jones, R. 2008. Greenhouse Gas Emissions from Two Soils Receiving Nitrogen Fertilizer and Swine Manure Slurry. J. Environ. Qual. **37**: 1432-1438.

Johnson, J. M. F., Franzluebbers, A. J., Weyere, S. L. and Reicosky, D. C. 2007. Agricultural opportunities to mitigate greenhouse gas emissions. Environ. Pollut. **150**:107–124.

Johnson, L. M., Harrison, J. H., Davidson, D., Swift, M., Mahanna, W. C. and Shinners, K. 2002a. Corn silage management iii. Effect of hybrid, maturity, and processing on nitrogen metabolism and ruminal fermentation. J. Dairy Sci. **85**: 2928-2947.

Johnson, L. M., Harrison, J. H., Davidson, D., Swift, M., Mahanna, W. C. and Shinners, K. 2002b. Corn silage management ii: effects of hybrid, maturity, and mechanical processing on digestion and energy content J. Dairy Sci. **85**: 2913–2927.

Johnson, L. M., Harrison, J. H., Davidson, D., Swift, M., Mahanna, W. C. and Shinners, K. 2002c. Corn silage management i: effects of hybrid, maturity, and mechanical processing on chemical and physical characteristics. J. Dairy Sci. **85**: 833–853.

Johnson, D. E, Phetteplace, H. W. and Seidl, A. F. 2002. Methane, nitrous oxide and carbon dioxide emissions from ruminant livestock production systems. pages 77-86 *in* J. Takahashi, B. A. Young, Eds. Greenhouse Gases and Animal Agriculture. Amsterdam, The Netherlands.

Johnson, D. E., Phetteplace, H. W., Seidl, A. F., Schneider, U. A. and Mccarl, B. A. 2003. Management variations for U. S. beef production systems: effects on greenhouse gas emissions and profitability. Pages 953-961 *in*: proceeding of the 3rd International Methane and Nitrous oxide Mitigation Conference; 2003 November 17-21; Beijing, china. Coal Institute, Beijing, china.

Johnson, K. A. and Johnson, D. E. 1995. Methane emissions from cattle. J. Anim. Sci. **73**: 2483-2492.

Joy, M. T., DePeters, E. J., Fadel, J. G. and Zinn, R. A. 1997. Effect of corn processing on the site and extent of digestion in lactating cows. J. Dairy Sci. **80**: 2087-2097.

Jungnitsch, P. F., Schoenau, J. J., Lardner, H. A. and Jefferson, P. G. 2011. Winter feeding beef cattle on the western Canadian prairies: Impacts on soil nitrogen and phosphorus cycling and forage growth. Agric. Ecosyst. Environ. **141**: 143–152.

Kagan, M. W. 2000. Nitrous oxide distribution in soil and surface flux. Msc Thesis, University of Manitoba, Winnipeg, MB.

Karimi-Zindashty, Y., Macdonald, J. D., Desjardins, R. L., Worth, D. E., Hutchinson, J. J. and Verge, X. P. C. 2012. Sources of uncertainty in the IPCC Tier 2 Canadian livestock model. J. Agri. Sci. **150:** 556-569.

Kebreab, E., Clark, K., Wagner-Riddle, C. and France, J. 2006b. Methane and nitrous oxide emissions from Canadian animal agriculture: A review. Can. J. Anim. Sci. **86**: 135–158.

Kebreab, E., Dijkstra, J., Bannink, A. and France, J. 2009. Recent advances in modeling nutrient utilization in ruminants. J. Anim. Sci. **87**: E111-E122.

Kebreab, E., France, J., McBride, B. W., Odongo, N. E., Bannink, A., Mills, J. A. N and Dijkstra J.. 2006a. Evaluation of models to predict methane emissions from enteric fermentation in North American dairy cattle. Pages 299-313 *in* E. Kebreab, J. Dijkstra, W. J. J. Gerrits, A. Bannink and J. France, eds. Nutrient Utilization in Farm Animals: Modelling Approach. CAB International, Wallingford, UK.

Kebreab, E., France, J., Mills, J. A. N., Allison, R. and Dijkstra, J. 2002. A dynamic model of N metabolism in the lactating dairy cow and an assessment of impact of N excretion on the environment. J. Anim. Sci. **80**: 248-259.

Kebreab, E., Johnson, A., Archibeque, L., Pape, D. and Wirth, T. 2008. Model for estimating enteric methane emission from United States dairy and feedlot cattle. J. Anim. Sci. **86**: 2738-2748.

Kebreab, E., Mills, J. A. N., Crompton, L. A., Bannink, A., Dijkstra, J., Gerrits, W. J. J. and France, J. 2004. An integrated mathematical model to evaluate nutrient partition in dairy cattle between animal and environment. Anim. Feed Sci. Technol. **112**: 131–154.

Kelln, B., Lardner, H. A., McKinnon, J. J., Campbell, J. R., Larson, K. and Damiran, D. 2011. Effect of winter feeding system on beef cow performance, reproductive efficiency, and system cost. Prof. Anim. Sci. **27:** 410-421.

Kelln, B., Lardner, H., Schoenau, J. and King, T. 2012. Effects of beef cow winter feeding systems, pen manure and compost on soil nitrogen and phosphorous amounts and distribution, soil density, and crop biomass. Nutr. Cycling Agroecosyst. **92**: 183-194.

Kennedy, P. M. and Milligan, L. P. 1978. Effects of cold exposure on digestion, microbial synthesis and nitrogen transformation in sheep. Brit. J. Nutr. **39**: 105-117.

Khorasani, G. R., Okine, E. K., Corbett, R. R. and Kennelly, J. J. 2001a. Nutritive value of peas for lactating dairy cattle. Can. J. Anim. Sci. **81**: 541-551.

Khorasani, G. R., Okine, E. K. and Kennelly, J. J. 2001b. Effect of forage source and amount of concentrate on rumen and intestinal digestion of nutrients in late-lactating cows. J. Dairy Sci. **84**: 1156-1165.

Knowlton, K. F., Glenn, B. P. and Erdman, R. A. 1998. Performance, ruminal fermentation, and site of starch digestion in early lactation cows fed corn grain harvested and processed differently. J. Dairy Sci. **81**: 1972-1984.

Kobayashi, Y. 2010. Abatement of methane production from ruminants: trends in the manipulation of rumen fermentation. Asian-Aust. J. Anim. Sci. **23**: 410–416.

Kohn, R. A. and Boston, R. C. 2000. The role of thermodynamics in controlling rumen metabolism. Pages 11-24 *in:* J. P. McNamara, J. France and D. E. Beever, eds. Modelling nutrient utilization in farm animals. CAB Publishing, Wallingford, UK.

Koong, L. J., Baldwin, R. L., Ulyatt, M. J. and Charlesworth, T. J. 1975. Iterative computation of metabolic flux and stoichiomteric parameters for alternative pathways in rumen fermentation. Comput. Prog. Biomed. **4**: 209–213.

Kopp, J. C., Wittenberg, K. M. and McCaughey, W. P. 2004. Management strategies to improve cow-calf productivity on meadow bromegrass pastures. Can. J. Anim. Sci. **84**: 529–535.

Kriss, M. 1930. Quantitative relations of the dry matter of the food consumed, the heat production, the gaseous outgo, and the insensible loss in body weight of cattle. J. Agric. Res. **40**: 283–295.

Kristensen, N. B. 2001. Rumen microbial sequestration of [2-(13)C] acetate in cattle. J. Anim. Sci. **79**: 2491-2498.

Kristensen N. B. 2005. Splanchnic metabolism of volatile fatty acids in the dairy cow. Anim. Sci. **80**: 3-10.

Kristensen, N. B., GaÈbel, G., Pierzynowski, S. G. and Danfñr, A. 2000a. Portal recovery of short-chain fatty acids infused into the temporarily isolated and washed reticulo-rumen of sheep. British J. Nutr. **84**: 477-482.

Kristensen, N. B. and Harmon. D. L. 2004. Effect of increasing ruminal butyrate absorption on splanchnic metabolism of volatile fatty acids absorbed from the washed reticulorumen of steers. J. Anim. Sci. **82**: 3549–3559.

Kristensen, T., Mogensen, L., Knudsen, M. T. and Hermansen, J. E. 2011. Effect of production system and farming strategy on green house gas emission from commercial dairy farms in a life cycle approach. Livest. Sci. **140**: 136-148.

Kristensen, N. B., Pierzynowski, S. G. and Danfær, A. 2000b. Portal-drained visceral metabolism of 3-hydroxybutyrate in sheep. J. Anim. Sci .**78**: 2223-2228.

Lal, R. 2004. Agricultural Activity and the global carbon cycle. Nutr. Cycling Agroecosyst. **70**:

103-116.

Lassey, K. R. 2008. Livestock methane emission and its perspective in the global methane cycle. Aust. J. Exp. Agric. **48**: 114–118.

Ledgard, S.F., Lieffering, M., McDevitt, J., Boyes, M. and Kemp, R. 2010. A Greenhouse gas footprint study for exported New Zealand lamb. Report prepared for the Meat Industry Association, Ballance Agri-Nutrients, Landcorp and MAF. Pages 26. [Online] Available http://www.mia.co.nz/docs/press_releases/greenhouse_gas_footprint_study_for_exported_nz_lamb._march_2010.pdf. [2012 August 1]

Le Du, Y. L. P., Baker, R. D. and Barker, J. M. 1976. Milk-fed calves. 2. The effect of length of milk feeding period and milk intake upon herbage intake and performance of grazing calves. J. Agric. Sci. **87**: 197-204.

Legesse, G., Smallb, J. A., Scottb, S. L., Crowa, G. H., Block, H. C., Alemu, A. W., Robins, C. D. and Kebreab, E. 2011. Predictions of enteric methane emissions for various summer pasture and winter feeding strategies for cow calf production. Anim. Feed Sci. Technol. **166-167**: 678-687.

Lemke, R. L., VandenBygaart, A. J., Campbell, C. A., Lafond, G. P., McConkey, B. G. and Grant, B. B. 2011. Long-term effects of crop rotations and fertilization on soil C and N in a thin Black Chernozem in southeastern Saskatchewan. Can. J. Soil Sci. **92**: 449-461.

Leng, R. A. and Brett, D. J. 1966. Simultaneous measurements of the rate of production of acetic, propionic and butyric acids in the rumen of sheep on different diet and the correlation between production rates and concentration of these acids in the rumen. Brit. J. Nutr. **20:** 541-552.

Le Mer, J. and Roger, P. 2001. Production, oxidation, emission and consumption of methane by soils: a review. Eur. J. Soil Biol.**37**: 25–50.

Lescoat, P. and Sauvant, D. 1995. Development of a mechanistic model for rumen digestion validated using the duodenal flux of amino acids. Reprod. Nutr. Dev. **35**: 45-70.

Li C. 2007. Quantifying greenhouse gas emissions from soils: Scientific basis and modeling approach. Soil Sci. Plant Nutr. **53**: 344–352.

Li, C. S., Frolking, S. and Frolking, T. A. 1992a. A model of nitrous oxide evolution from soil driven by rainfall events: 1. model structure and sensitivity. J. Geophys. Res. **97**: 9759-9776.

Li, C. S., Frolking, S. and Frolking, T. A. 1992b. A model of nitrous oxide evolution from soil driven by rainfall events: 2. model applications. J. Geophys. Res. **97**: 9777-9783.

Li, C. S., Frolking, S. and Harris, R. 1994. Modeling carbon biogeochemistry in agricultural soils. Global Biogeochem. Cycles. **8**: 237-254.

Li, C. S., Farahbakshazad, N., Jaynes, D. B., Dinnes, D. L., Salas, W. and McLaughlin, D.

2006. Modeling nitrate leaching with a biogeochemical model modified based on observations in a row-crop field in Iowa. Ecol. Model. **196**: 116-130.

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

Liebig, M. A., Gross, J. R., Kronberg, S. L., Phillips, R. L. and Hanson J. D. 2010. Grazing Management contributions to net global warming potential: A long-term evaluation in the northern great plains. J. Environ. Qual. **39**:799–809.

Lin, L. I. K. 1989. A concordance correlation coefficient to evaluate reproducibility. Biometrics **45**: 255-268.

Little, S., Linderman, J., Maclean, K. and Janzen, H., 2008. Holos – A tool to estimate and reduce greenhouse gases from farms. Methodology and algorithms for versions 1.1.x. Agriculture and Agri-Food Canada, Cat. No. A52-136/2008E-PDF. 158 pp.

López, S., Hovell, F D. D., Dijkstra, J. and France, J. 2003. Effects of volatile fatty acid supply on their absorption and on water kinetics in the rumen of sheep sustained by intragastric infusions. J Anim. Sci. **81**: 2609-2616.

Lovett, D. K., Shalloo, L., Dillon, P. and O'Mara, F. P. 2008. Greenhouse gas emissions from pastoral based dairying systems: the effect of uncertainty and management change under two contrasting production systems. Livest. Sci. **116**: 260–274.

Ludington, D. and Johnson, E. L. 2003. Dairy farm energy audit summary. FlexTech Services. New York State Energy Research and Development Authority. Albany, NY. [Online] Available http://www.nyserda.ny.gov/Publications/Research-and-Development/~/media/Files/Publications/Energy/Audit/Reports/dairy-farm-energy.ashx. [2011 December 11].

MacNeil. M. D., Newman. S., Enns, R. M. and Stewart-Smith, J. 1994. Relative economic values for Canadian beef production using specialized sire and dam lines. Can. J. Anim. Sci. **74**: 4ll-417.

Manitoba Agriculture and Food. 1999. Pasture quality survey – Eastern/Interlake. Winnipeg, Manitoba. [Online] Available: http://www.mbforagecouncil.mb.ca/default.aspx. [2009 July 5].

Manitoba Agricultural Review. 2002. Program and Policy Analysis Branch, Manitoba Agriculture and Food, Winnipeg, MB. 70 pp.

Manitoba Agriculture, Food and Rural initiative. 2004. Beef processing in Manitoba: feasibility analysis. Economic Development Initiative Branch, Winnipeg Manitoba. 70 pp.

Manitoba Agriculture, Food and Rural Initiatives (MAFRI). 2008. 2008 Manitoba Agriculture yearbook. [Online] Available http://www.gov.mb.ca/agriculture/statistics/indexyearbook.html. [2012 January 10].

Manitoba Agriculture, Food and Rural Initiatives (MAFRI). 2012. Questions and answers about custom slaughter and marketing at the farm gate. [Online] Available http://www.gov.mb.ca/agriculture/livestock/anhealth/jaa06s12.html. [2012 February 20].

Manitoba Forage Council (MFC). 2006. Manitoba forage benchmarking project interim report. [online] Available: http://www.mbforagecouncil.mb.ca/projectstechnicalinfo/currentongoingprojects/themanitobabenchmarkproject/default.aspx. [2009 September 12].

Manitoba Management Plus Program. 2012. Avilable http://www.mmpp.com/mmpp.nsf/mmpp_index.html. [2012 Novemebr 10].

Martin, S. B. 2004. Manure management in Canada. Statistics Canada, Farm Environmental Management in Canada, Agricultural Division. Catalogue no. 21-021-MIE-No. 002. [online] Available http://publications.gc.ca/collections/Collection/Statcan/21-021-M/21-021-MIE2004001. pdf. [2012 March 12].

Martin, P. A., Chamberlain, D, G., Robertson, S. and Hirst, D. 1994. Rumen fermentation patterns in sheep receiving silage of different chemical composition supplemented with concentrates rich in starch or in digestible fibre. J. Agri. Sci. **122**: 145-150.

Martin, C., Kirstensen, N. B. and Huhtanen. P. 2001. Comparison of non-tracer and tracer methods for determination of volatile fatty acid production rate in the rumen of sheep fed on two levels of intake. Brit. J. Nutr. **86:** 331-340.

Martin, C. Morgavi, D. P. and Doreau, M. 2010. Methane mitigation in ruminants: from microbe to the farm scale. Anim. **4:** 351-365.

Martin, C., Rouel, J., Jouany, J. P., Doreau, M. and Chilliard, Y. 2008. Methane output and diet digestibility in response to feeding dairy cows crude linseed, extruded linseed, or linseed oil. J. Anim. Sci. **86**: 2642-2650.

Martineau, R., Lapierre, H., Ouellet, D. R., Pellerin, D. and Berthiaume, R. 2007. Effect of the method of conservation of timothy on nitrogen metabolism in lactating dairy cows. J. Dairy Sci. **90**: 2870-2882.

Mathot, M., Decruyenaere, V., Stilmant, D. and Lambert, R. 2012. Effect of cattle diet and manure storage conditions on carbon dioxide, methane and nitrous oxide emissions from tie-stall barns and stored solid manure. Agri. Ecosyst. Environ. **148:** 134-144.

McAllan, A. B., Sutton, J. D., Beever, D. E. and Napper, D. J. 1994. Rumen fermentation characteristics and duodenal nutrient flow in lactating cows receiving two types of grass silage with two levels of concentrates. Anim. Feed Sci. Technol. **46**: 277-291.

McAllister, T. A., Okine, E., Mathison, G. W. and Cheng, K. J. 1996. Dietary, environmental and microbial aspects of methane production in ruminants. Can. J. Anim. Sci. **76**: 231-243.

McCartney, D. 2011. Country pasture/forage resource profiles. Agriculture and Agri Food Canada, Lacombe, Alberta. [Online] Available: http://www.fao.org/ag/AGP/AGPC/doc/

Counprof/Canada/Canada.html. [2012 May 13].

McCaughey, W. P., Wittenberg, K. M. and Corrigan, D. 1999. Impact of pasture type on methane production by lactating beef cows. Can. J. Anim. Sci. **79**: 221–226.

McGeough, E. J., Little, S. M., Janzen, H. H., McAllister, T. A., McGinn, S. M., and Beauchemin, K. A. 2012. Life-cycle assessment of greenhouse gas emissions from dairy production in Eastern Canada: A case study. J. Dairy Sci. **9**: 5164–5175.

Menzi, H., Katz, P. E., Fahrni, M., Neftel, A. and Frick, R. 1998. A simple empirical model based on regression analysis to estimate ammonia emissions after manure application. Atmos. Environ**. 32**: 301–307.

Mills, J. A. N. 2008. Modeling methane emission from farm livestock. Pages 189-203 *in* J. France and E. Kebreab, eds. Mathematical modeling in animal nutrition. CAB international, Wallingford, UK.

Mills, J. A. N., Dijkstra, J., Bannink, A., Cammell, S. B., Kebreab, E. and France, J. 2001. A mechanistic model of whole-tract digestion and methanogenesis in the lactating dairy cow: Model development, evaluation, and application. J. Anim. Sci. **79**: 1584-1597.

Mills, J. A. N., Kebreab, E., Yates, C. M., Crompton, L. A., Cammell, S. B., Dhanoa, M. S., Agnew, R. E. and France, J. 2003. Alternative approaches to predicting methane emissions from dairy cows. J. Anim. Sci. **81**:3141–3150.

Minson, D. J. and Wilson, J. R. 1994. Prediction of intake as an element of forage quality. Pages. 533-563 *in:* G. C. Fahey ed. Forage quality, evaluation and utilization. American Society of Agronomy, Madison, WI.

Moe, P. W. and Tyrrell. H. F. 1979. Methane production in dairy cows. J. Dairy Sci. **62**: 1583–1586.

Mohammed, R., McGinn, S. M. and Beauchemin, K. A. 2010. Prediction of enteric methane output from milk fatty acid concentrations and rumen fermentation parameters in dairy cows fed sunflower, flax, or canola seeds. J. Dairy Sci. **94**: 6057–6068.

Moorby, J. M., Dewhurst, R. J., Evans, R. T. and Daneln, J. L. 2006. Effects of dairy cow diet forage proportion on duodenal nutrient supply and urinary purine derivative excretion. J. Dairy Sci. **89**: 3552-3562.

Morvay, Y., Bannink, A., France, J., Kebreab, E., and Dijkstra, J. 2011. Evaluation of models to predict the stoichiometry of volatile fatty acid profiles in rumen fluid of lactating Holstein cows. J. Dairy Sci. **94**: 3063–3080

Moss, A. R., Jouany, J. and Newbold, J. 2000. Methane production by ruminants: its contribution to global warming. Ann. Zootech. **49**: 231–253.

Murphy, M. R., Baldwin, R. L. and Koong, L. J. 1982. Estimation of stoichiometric

parameters for rumen fermentation of roughage and concentrate diets. J. Anim. Sci. **55**: 411–421.

Nagorcka, B. N., Gordon, G. L. R. and Dynes, R. A. 2000. Towards a more accurate representation of fermentation in mathematical models of the rumen. Pages 37-48 *in:* J. P. McNamara, J. France, and D. E. Beever, eds. Modelling Nutrient Utilization in Farm Animals. CAB Publishing, Wallingford, UK.

Nagy, C. N. 2000. Energy and greenhouse gas emission coefficients for inputs used in agriculture. In report to prairie adaptation research collaborative; Centre for Studies in Agriculture, Law and the Environment: Saskatoon, Canada, p. 11.

Narasimhalu, P., Kong, D. and Choo, T. M. 1998. Straw yields and nutrients of seventy-five Canadian barley cultivars. Can. J. Anim. Sci. **78**: 127–134.

National Research Council (NRC). 1982. Nutritional data for United States and Canadian feeds: United States-Canadian table of feed composition. National Academy Press. Washington, DC.

National Research Council (NRC). 1996. Nutrient requirements of beef cattle. National academy press, Washington, DC.

National Research Council (NRC). 2000. Nutrient requirement of beef cattle (revised ed.). National Academic Press, Washington, DC.

National Research Council (NRC). 2001. Nutrient requirement of dairy cattle. National Academic Press, Washington, DC.

Neal, H. D., Dijkstra, J. and Gill, M. 1992. Simulation of nutrient digestion, absorption and outflow in the rumen: Model evaluation. J. Nutr. **122**: 2257-2272.

Netherland Environmental Assessment Agency (NEAA). 2010. Greenhouse gas emission in the Netherlands 1990-2008. National inventory report 2010. [Online] Available: http://www.broeikasgassen.nl/docs/NIR2010_NL_v2.pdf, [2010 Dec. 10].

Nguyen, T. L. T., Hermansen, J. E. and Mogensen, L. 2010. Environmental consequences of different beef production systems in the EU. J. Cleaner Prod. **18**: 756-766.

Nocek, J. E. and Tamminga, S. 1991. Site of digestion of starch in the gastrointestinal tract of dairy cows and its effect on milk yield and composition. J. Dairy Sci. **74:** 3598-3629.

Nozière, P., Glasser, F., Loncke, C., Ortigues Marty, I., Vernet, J. and Sauvant, D. 2011. Modelling rumen volatile fatty acids and its evaluation on net portal fluxes in ruminants. Pages 158-167 *in:* D. Sauvant, J. Van Milgen, P. Faverdin and N. Friggens eds. Modelling nutrient digestion and utilization in farm animals. Wageningen Academic Publishers, The Netherlands.

Nozière, P., Ortigues-Marty, I., Loncke, C., Sauvant, D., 2010. Carbohydrate quantitative digestion and absorption in ruminants: from feed starch and fibre to nutrients available for tissues. Anim. **4**: 1057–1074.

Norton, J. M. 2008. Nitrification in agricultural soils. Pages 173-200 *in:* J. S. Schepers and W. R. Raun, eds. Nitrogen in agricultural system, USA.

Oba, M. and Allen, M. S. 2003. Effect of corn grain conservation method on feeding behaviour and productivity of lactating dairy cows at two dietary starch concentrations. J. Dairy Sci. **86**: 174-183.

O'Brien, D., Shalloo, L., Grainger, C., Buckley, F., Horan, B. and Wallace, M. 2010. The influence of strain of Holstein-Friesian cow and feeding system on greenhouse gas emissions from pastoral dairy farms. J. Dairy Sci. **93**: 3390–3402.

Oenema, O., Velthof, G. and Kuikman, P. 2001. Technical and policy aspects of strategies to decrease greenhouse gas emissions from agriculture. Nut. Cycling Agroecosyst. **60:** 301-315.

Offner, A. and Sauvant, D. 2004. Comparative evaluation of the Molly, CNCPS and LES rumen models. Anim. Feed Sci. Technol. **112**: 107-130.

Offner, A. and Sauvant, D. 2006. Thermodynamic modeling of ruminal fermentations. Anim. Res. **55**: 343-365.

Ogino, A., Orito, H., Shimada, K. and Hirooka, H. 2007. Evaluating environmental impacts of the Japanese beef cow–calf system by the life cycle assessment method. Anim. Sci. J. **78**: 424–432.

Okine, E. K., Mathison, G. W. and Hardin, H. T. 1989. Effects of changes in frequency of reticular contractions on fluid and particulate passage rates in cattle. J. Anim. Sci. **67**: 3388–3396.

Olesen, J. E. and Schelde, K. 2008. FarmGHG,: A model for estimating greenhouse gas emissions from livestock farms. User manual. Aarhus University. [Online] Available: http://agrsci.au.dk/fileadmin/DJF/JPM/Klima/JEO/FarmGHGManual5.pdf. [2012 Apr. 27].

O'Mara, F. P. 2011. The significance of livestock as a contributor to global greenhouse emissions today and in the near future. Anim. Feed Sci. Tech. **166-168**: 7-15.

Ominski, K. H., Boadi, D. A. and Wittenberg, K. M. 2006. Enteric methane emission from backgrounded cattle consuming all-forage diet. Can. J. Anim. Sci. **86**: 393-400.

Ominski, K. H. and Wittenberg, K. M. 2006. Strategies for reducing entric methane emissions in forage-based beef production systems. Pages 261-270 *in:* J. S. Bhatti, R. Lal, M. J. Apps, and M. A. Price, eds. Climate change and managed ecosystems. CRC Press, NW.

Pelletier, N., Pirog, R. and Rasmussen, R. 2010. Comparative life cycle environmental impacts of three beef production strategies in the Upper Midwestern United States. Agric. Sys. **103**: 380–389.

Pelster, D. E., Chantigny, M. H., Rochette, P., Angers, D. A., Rieux, C. and Vanasse, A. 2012. Nitrous oxide emissions respond differently to mineral and organic nitrogen sources in

contrasting soil types. J. Environ. Qual. **41**: 1427-1435.

Peters, G. M., Rowley, H. V., Wiedemann S., Tucker R., Short, M. D. and Schulz, M. 2010. Red meat production in Australia: life cycle assessment and comparison with overseas studies. Environ. Sci. Technol. **44**: 1327–1332.

Peterson, S. O., Amon, B. and Gattinger, A. 2005. Methane oxidation in slurry storage surface crusts. J. Environ. Qual. **34**:455–461.

Petersen, S. O., Sommer, S. G., Béline, F., Burton, C., Dach, J., Dourmad, J. Y., Leip, A., Misselbrook, T., Nicholson, F., Poulsen, H. D., Provolo, G., Sørensen, P., Vinnerås, B., Weiske, A., Bernal, M. P., Böhm, R., Juhász, C. and Mihelic, R. 2007. Recycling of livestock manure in a whole-farm perspective. Livest. Sci. **112**: 180–191.

Phetteplace, H. W., Johnson, D. E. and Seidl, A. F. 2001. Greenhouse gas emissions from simulated beef and dairy livestock systems in the United States. Nutr. Cycling Agroecosyst. **60**: 99-102.

Phipps, R. H., Sutton, D. J., Beever, D. E. and Jones, A. K. 2000. The effect of crop maturity on nutritional value of maize silage for lactating dairy cows. 3. Food intake and milk production. Anim. Sci. **71**: 401-409.

Phipps, R. H., Sutton, D. J., Humphries, D. J. and Jones, A. K. 2001. A comparison of the effects of cracked wheat and sodium hydroxide-treated wheat on food intake, milk production and rumen digestion in dairy cows given maize silage diets. Anim. Sci. **72**: 585-594.

Pitt, R. E., Van Kessel, J. S., Fox, D. G., Pell, A. N., Barry, M. C. and Van Soest, P. J. 1996. Prediction of ruminal volatile fatty acids and pH within the net carbohydrate and protein system. J. Anim. Sci. **74**: 226-244.

Plaizier, J. C., Garner, T., Droppo, T. and Whiting, T. 2004. Nutritional practices on Manitoba dairy farm. Can. J. Anim. Sci.**84**: 501-509.

Po, H. N. and Senozan, N. M. 2001. Henderson-Hasselbalch equation: its history and limitations. J. Chem. Edu. **78**: 1499–1503.

Poore, M. H., Moore, J. A., Eck, T. P., Swingle, R. S. and Theurer, C. B. 1993. Effect of fiber and ruminal starch degradability on the site and extent of digestion in dairy cows. J. Dairy Sci. **76**: 2244-2253.

Power, A. G. 2010. Ecosystem services and agriculture: tradeoffs and synergies. Biolo. Sci. **365**: 2959-2975.

Prairie Provinces Committee on Livestock Development and Manure Management. 2006. Tri-provincial manure application and use guidelines. Manitoba version. [Online] Available http://www.gov.mb.ca/agriculture/livestock/beef/pdf/baa08s01a.pdf. [2012 March 23].

Qiu, X., Eastridge, M. L. and Wang, Z. 2003. Effect of corn silage hybrid and dietary

concentration of forage NDF on digestibility and performance by dairy cows. J. Dairy Sci. **86**: 3667-3674.

Reece, W. O. 2004. Dukes` physiology of domestic animals. Second ed. Cornell University press. London.

Rejis J. W. 2007. Improving slurry by diet adjustments: a novelty to reduce N losses from grassland based dairy farms. PhD thesis, Wageningen University, The Netherlands.

Reynolds, C. K., Aikman, P. C., Lupoli, B. Humphries, D. J. and Beever, D. E. 2003. Splanchnic metabolism of dairy cows during the transition from late gestation through early lactation. J. Dairy Sci. **86**: 1201-1217.

Reynolds, C. K., Crompton, L. A. and Mills, J. A. N. 2009. Improving the Efficiency of Energy Utilization in Cattle. Proceedings of the 30th Western Nutrition Conference, September 23-24, Winnipeg, MB. Pp 55-65.

Robinson, P. H., Fadel, J. G. and Tamminag, S. 1986. Evaluation of mathematical models to describe nuteral detergent residue in terms of its susceptibility to degradation in the rumen. Anim. Feed Sci. Technol. **15**: 249-271.

Robinson, P. H., Tamminga, S. and Van Vuuren, A. M. 1987. Influence of declining level of feed intake and varying the proportion of starch in the concentrate on rumen ingesta quantity, composition and kinetics of ingesta turnover in dairy cows. Livest. Prod. Sci. **17**: 37-62.

Rochette, P., Angers, D. A., Chantigny, M. H., Bertrand, N. and Cote, D. 2004. Carbon dioxide and nitrous oxide emission following fall and spring applications of pig slurry to an agricultural soil. Soil Sci. Soc. Am. J. **68**: 1410-1420.

Rochette, P., van Bochove, E., Prevost, D., Angers, D. A., Cote, D., Bertrand, N., 2000. Soil carbon and nitrogen dynamics following application of pig slurry for the 19th consecutive year: II. Nitrous oxide fluxes and mineral nitrogen. Soil Sci. Soc. Am. J. **64**: 1396-1403.

Rochette, P., Guilmette, D., Chantigny, M. H., Angers, D. A., MacDonald, J. D., Bertrand, N., Parent, L. É., Côte´, D. and Gasser, M. O. 2008c. Ammonia volatilization following application of pig slurry increases with slurry interception by grass foliage. Can. J. Soil Sci. **88**: 585-593.

Rochette, P. and Janzen H. H. 2005. Towards a revised coefficient for estimating for estimating N2O from legumes. Nutr. Cycling Agroecosyst. **73**:171-179.

Rochette, P., Worth, D. E. Huffman, E. C., Brierley, J. A., McConkey, B. G., Yang, J., Hutchinson, J. J., Desjardins, R. L., Lemke, R. L. and Gameds, S. 2008b. Estimation of N_2O emissions from agricultural soils in Canada. II. 1990-2005 inventory. Can. J. Soil Sci. **88**: 655-669.

Rochette, P., Worth, D. E. Lemke, R. L., McConkey, B. G., Pennock, D. J., Wagner-Riddle, C., Desjardins, R. L. 2008a. Estimation of N_2O emissions from agricultural soils in Canada. I.

Development of a country-specific methodology. Can. J. Soil Sci. **88**: 641-654.

Rodhe, L., Pell, M. and Yamulki, S. 2006. Nitrous oxide, methane and ammonia emissions following slurry spreading on grassland. Soil Use Manage. **22**: 229-237.

Rotz, C. A. 2003. How to maintain forage quality during harvest and storage. Proceedings of 2003 Western Canadian Dairy Seminar. Advances Dairy Technol. **15**: 227-236.

Rotz, C. A. 2004. Management to reduce nitrogen losses in animal production. J. Anim. Sci. **82**: E119-137.

Rotz, C. A., Buckmaster, D. R. and Comerford, J. W. 2005. A beef herd model for simulating feed intake animal performance and manure excretion in farm systems. J. Anim. Sci. **83**: 231-242.

Rotz, C. A., Corson, M. S., Chianese, D. S., Montes, F., Hafner, S. D., Jarvis, R., Coiner, C. U. 2011a. The Integrated Farm System Model. Reference Manual, Version 3.4. Pasture Systems and Watershed Management Research Unit. Agricultural Research Service, USDA.

Rotz, C. A., Kleinman, P. J. A., Dell, C. J., Veith, T. L. and Beegle, D. B. 2011b. Environmental and economic comparisons of manure application methods in farming systems. J. Environ. Qual. **40:** 438-448.

Rotz, C. A., Montes, F. and Chianese, D. S. 2010. The carbon footprint of dairy production systems through partial life cycle assessment. J. Dairy Sci. **93**: 1266-1282.

Schils, R. L. M., Olesen, J. E., del Prado, A. and Soussana, J. F. 2007. A review of farm level modelling approaches for mitigating greenhouse gas emissions from ruminant livestock systems. Livest. Sci. **112:** 240-251.

Schils R. L. M., Verhagen, A., Aarts, H. F. M. and Šebek, L. B. J. 2005. A farm level approach to define successful mitigation strategies for GHG emissions from ruminant livestock systems. Nutr. Cycling Agroecosyst. **71**: 163-175.

Sejian, V., Lal, R., Lakritz, J. and Ezeji, T. 2011. Measurement and prediction of enteric methane emission. Int. J. Biometeorol. **55**: 1-16.

Shaffer, M. J., Halvorson, A. D. and Pierce, F. J. 1991. Nitrate leaching and economic analysis package (NLEAP): model description and application. Pages 285-298 *in:* R. F., Follett, D. R., Deeney, R. M., Cruse, Eds. Managing nitrogen for groundwater quality and farm profitability. Soil Sci. Soc. Am., Inc. Madison, WI.

Sharp, W. M., Johnson, R. R. and Owens, F. N. 1982. Ruminal VFA production with steers fed whole or ground corn grain. J. Anim. Sci. **55**:1501-1514.

Sherlock, R. A. and Bright, K. P. 1999. An object-oriented framework for farm system simulation. Pages 783-788 *in:* L. Oxley, F. Scrimgeour, A. Jakeman, Eds. MODSIM 99 Proceedings of the International Congress on Modelling and Simulation. Modelling and

Simulation, Society of Australia and New Zealand.

Sherlock, R. R., Sommer, S. G., Khan, R. Z., Wood, C. W., Guertal, E. A., Freney, J. R., Dawson, C. O. and Cameron, K. C. 2002. Ammonia, methane and nitrous oxide emission from pig slurry applied to a pasture in New Zealand. J. Environ. Qual. **31**: 1491–1501.

Silveira, C., Oba, M., Yang, W. Z. and Beauchemin, K. A. 2007. Selection of barley grain affects ruminal fermentation, starch digestibility, and productivity of lactating dairy cows. J. Dairy Sci. **90**: 2860-2869.

Small, J. A. and McCaughey, W. P. 1999. Beef cattle management in Manitoba. Can. J. Anim. Sci. **79**: 539-544.

Smith, W. N., Desjardins, R. L., Grant, B., Li, C., Corre, M., Rochette, P. and Lemke, R. 2002. Testing the DNDC model using N_2O emissions at two experimental sites. Can. J. Soil Sci. **82**: 365-374.

Smith, W. N., Grant, B. B., Desjardins, R. L., Rochette, P., Drury, C. F. and Li, C. 2008. Evaluation of two process-based models to estimate soil N_2O emissions in Eastern Canada. Can. J. Soil Sci. **88**: 251-260.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, B. and Sirotenko, O. 2007: Agriculture. In Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change B. Metz, O. R. Davidson, P. R. Bosch, R. Dave, L. A. Meyer, eds. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Statistics Canada. 2009. Canadian Socio-Economic Information Management System: CANSIM. Number of cattle by class and farm type, annual (head), 1930-2009. [updated daily]. [Online] Available: http://cansim2.statcan.ca/. [2009 June 12].

Steinfeld, H., Gerber, P., Wassenaar, T., Castel, V., Rosales, M. And de Haan, C. 2006. Livestock's Long Shadow: Environmental Issues and Options. FAO, Rome, Italy.

Stensig, T. and Robinson, P. H. 1997. Digestion and passage kinetics of forage fiber in dairy cows as affected by fiber-free concentrate in the diet. J. Dairy Sci. **80**: 1339-1352.

Stewart, A. A., Little, S. M., Ominski, K. H., Wittenberg, K. M., Janzen, H. H., 2009. Evaluating greenhouse mitigation practices in livestock systems: an illustration of a whole-farm approach. J. Agric. Sci. **147**: 367-382.

Storm, A. C., Hanigan , M. D. and Kristensen N. B. 2011. Effects of ruminal ammonia and butyrate concentrations on reticuloruminal epithelial blood flow and volatile fatty acid absorption kinetics under washed reticulorumen conditions in lactating dairy cows. J. Dairy Sci. **94**: 3980-3994

St-Pierre, N. R. 2003. Reassessment of biases in predicted nitrogen flows to the duodenum by

NRC 2001. J. Dairy Sci. **86**: 344-350.

Subak, S. 1999. Global environmental costs of beef production. Ecol. Econ. **30**: 79-91.

Sutton, J. D. 1980. Digestion and end-product formation in the rumen from production ration. Pages 271-290 *in:* Y. Ruckebusch and P. Thivend, eds. Digestive physiology and metabolism in ruminants. AVI Pub. Co., Inc., Westport, CT, USA.

Sutton J. D. 1985. Digestion and absorption of energy substrates in the lactating cows. J. Dairy Sci. **68**: 3376-3393.

Sutton, J. D., Cammell, S. B., Beever, D. E., Humphries, D. J. and Phipps, R. H. 1998. Energy and nitrogen balance of lactating dairy cows given mixtures of urea-treated whole-crop wheat and grass silage. Anim. Sci. **67**: 203-212.

Sutton, J. D., Cammell, S, B., Phipps, R. H., Beever, D. E. and Humphries, D. J. 2000. The effect of crop maturity on the nutritional value of maize silage for lactating dairy cows. 2. ruminal and post-ruminal digestion. Anim. Sci. **71**: 391-400.

Sutton, J. D., Dhanoa, M. S., Morant, S. V., France, J., Napper, J. D. and Schuller, E. 2003. Rates of production of acetate, propionate, and butyrate in the rumen of lactating dairy cows given normal and low-roughage diets. J. Dairy Sci. **86**: 3620-3633.

Sveinbjörnsson, P., Huhtanen, P. and Udén, J., 2006. The Nordic dairy cow model, Karoline-development of volatile fatty acid sub-model. Pages 383-406 *in:* E. Kebreab, J. Dijkstra, A. Bannink, W. J. J. Gerrits and J. France, eds. Nutrient digestion and utilization in farm animals: modeling approach. CAB Publishing, Wallingford, UK.

Takahashi, T., Suzuki, Y. and Horiguchi, K. 2002. Effect of heat exposure on methane emission from expiratory gas in sheep fed with high concentrate diets. Proc. 1st International Conference on Greenhouse Gases and Animal Agriculture, Obihiro, Japan. 7-11, Nov. 2001.

Taylor, C. C. and Allen, M. S. 2005a. Corn grain endosperm type and brown midrib 3 corn silage: site of digestion and ruminal digestion kinetics in lactating cows. J. Dairy Sci. **88**: 1413–1424.

Taylor, C. C. and Allen, M. S. 2005b. Corn grain endosperm type and brown midrib 3 corn silage: ruminal fermentation and N partitioning in lactating cows. J. Dairy Sci. **88**: 1434-1442.

Taylor, C. C. and Allen, M. S. 2005c. Corn grain endosperm type and brown midrib 3 corn silage: feeding behaviour and milk yield of lactating cows. J. Dairy Sci. **88**: 1425–1433.

Tedeschi, L. O., Fox, D. G., Sainz, R. D., Barioni, L. G., de Medeiros, S. R. And Boni, C. 2005. Mathematical models in ruminant nutrition. Sci. Agri. (Piracicaba, Braz.) **62**: 79-91.

Tenuta, M., Mkhabela, M., Tremorin, D., Coppi, L., Phipps, G., Flaten, D. and Ominski, K. 2010. Nitrous Oxide and Methane Emission from a Coarse-texture Grassland Soil Receiving Hog Slurry. Agric. Ecosyst. Environ. **138**: 35-43.

The AEgis Technologies Group, inc. 2008. acslX user`s guide (Version 2.4). Huntsville, USA.

Thorburn, P. J., Biggs, J. S., Collins, K. and Probert, M. E. 2010. Using the APSIM model to estimate nitrous oxide emissions from diverse Australian sugarcane production systems. Agric. Ecosys. Environ. **136**: 343-350.

Thornley, J. and France, J. 2007. Mathematical models in agriculture: Quantitative methods for the plant, animal and ecological sciences. CAB International. Wallingford, UK.

Thornton, P. K. 2010. Livestock production: recent trends, future prospects. Phil. Trans. Reasarch. Soc. B, **365:** 2853–2867.

Tothi, R., Lund, P., Weisbjerg, M. R. and Hvelplund, T. 2003. Effect of expander processing on fractional rate of maize and barley starch degradation in the rumen of dairy cows estimated using rumen evacuation and in situ techniques. Anim. Feed Sci. Technol. **104**: 71-94.

Torrent, J. and Johnson, D. E. 1994. Methane production in the large intestine of sheep. Pages 391–394 *in* J. F Aquilera, ed. Energy metabolism of farm animals EAAP Publication No. 76 Granada, Spain: CSIC Publishing Services.

Traxler, M. J., Fox, D. G., Van Soest, P. J., Pell, A. N., Lascano, C. E., Lanna, D. P., Moore, J. E., Lana, R. P., Velez, M. and Flores, A. 1998. Predicting forage indigestible NDF from lignin concentration. J. Anim. Sci. **76**: 1469-1480.

Tremorin, D. A., Tenuta, M., Mkhabela, M., Donald N. Flaten, D. N. and Ominski, K. H. 2012. Nitrous oxide emissions from feces and synthetic urine of cattle grazing forage grass fertilized with hog slurry. Anim Feed Sci. Technol. (In press).

Ueda, K., Ferlay, A., Chabrot, J., Loor, J. J., Chilliard, Y. and Doreau, M. 2003. Effect of linseed oil supplementation on ruminal digestion in dairy cows fed diets with different forage: concentrate ratios. J. Dairy Sci. **86**: 3999-4007.

United Nations. 1998. Kyoto protocol to the United Nations framework convention on climate change. [Online] Available: http://unfccc.int/resource/docs/convkp/kpeng.pdf [2012 May 10].

United States Environmental Protection Agency (US EPA). 2010. Inventory of U. S. greenhouse gas emission and sinks: 1990 – 2008. EPA report 430-R-10-006. [Online] Available: http://www.epa.gov/climatechange/emissions/downloads10/US-GHG-Inventory-2010_Report.pdf. [2010 Dec. 10].

United States Environmental Protection Agency (US EPA). 2012. Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990–2010. [Online] Available: http://epa.gov/climatechange/emissions/usinventoryreport.html. [2012 May 1].

VanderZaag, A. C., Jayasundara, S. and Wagner-Riddle, C., 2011. Strategies to mitigate nitrous oxide emissions from land applied manure. Anim. Feed Sci. Technol. **166-167**, 464-479.

Van Soest, P. J., Van Amburgh, M. E., Robertson, J. B. and Knaus, W. F. 2005. Validation

of the 2.4 times lignin factor for ultimate extent of NDF digestion, and curve peeling rate of fermentation curve in to pools. Pages 139-149 *in* Proceedings Cornell Nutritional Conference for Feed Manufacturers. 24–26 Oct. 2005. East Syracuse, NY, USA.

Van Vuuren, A. M., Klop, A., Van der Koelen, C. J. and De Visser, H. 1999. Starch and stage of maturity of grass silage: site of digestion and intestinal nutrient supply in dairy cows. J. Dairy Sci. **82**: 143-152.

Velthof, G., Kuikman, P. and Oenema, O. 2003. Nitrous oxide emission from animal manures applied to soil under controlled conditions. Biol. Fertil. Soils **37**: 221–230.

Vergé, X. P. C., Dyer, J. A., Desjardins R. L. and Worth, D. 2007. Greenhouse gas emissions from the Canadian dairy industry in 2001. Agric. Syst. **94**: 683-693.

Vergé, X. P. C., Dyer, J. A., Desjardins, R. L. and Worth, D. 2008. Greenhouse gas emissions from the Canadian beef industry. Agric. Syst. **98:** 126-134.

Voelker, J. A. and Allen, M. S. 2003a. Pelleted beet pulp substituted for high-moisture corn: 2. Effects on digestion and ruminal digestion kinetics in lactating dairy cows. J. Dairy Sci. **86**: 3553-3561.

Voelker, J. A. and Allen, M. S. 2003b. Pelleted beet pulp substituted for high-moisture corn: 3. effects on ruminal fermentation, pH, and microbial protein efficiency in lactating dairy cows. J. Dairy Sci. **86**: 3562–3570.

Veysset P., Lherm M. and Bébin, D. 2010. Energy consumption, greenhouse gas emissions and economic performance assessments in French Charolais suckler cattle farms: Model-based analysis and forecast. Agric. Syst. **103**: 41-50.

Waghorn, G. C. and Hegarty, R. S. 2011. Lowering ruminant methane emissions through improved feed conversion efficiency. Anim. Feed Sci. Technol. **166-167**: 291-301.

Wagner-Riddle, C. and Thurtell, G. W. 1998. Nitrous oxide emissions from agricultural fields during winter and spring thaw as affected by management practices. Nutr Cycling Agroecosyst **52**:151-163.

Wagner-Riddle, C., Kebreab, E., France, J., Clark, K., Rapai, J., 2006. Supporting measurements required for evaluation of greenhouse gas emission models for enteric fermentation and stored animal manure. Pages 17-21 *in:* Discussion Paper for the 2nd meeting of the Animal Production and Manure Management Network, BIOCAP Canada Foundation, Kingston, Ontario.

Waldner, C. L., Kennedy, R. I., Rosengren, L. and Clark E. G. 2009. A field study of culling and mortality in beef cows from western Canada. Can. Vet. J. **50**: 491-499.

Wang, M. 2007. GREET version 1.8a. Argonne National Laboratory. <http://www.transportation.anl.gov/publications/transforum/v8/v8n2/greet_18b.html>

Wastney, M. E., Palliser, C. C., Lile, J. A., Macdonald, K. A., Penno, J. W. and Bright, K. P. 2002. A whole-farmmodel applied to a dairy system. Proc. N. Z. Soc. Anim. Prod., **62**: 120-123.

Webb, J., Sommer, S. G., Kupper, T., Groenestein, K., Hutchings, N. J., Eurich-Menden, B., Rodhe, L., Misselbrook, T. H. and Amon, B. 2012. Emissions of ammonia, nitrous oxide and methane during the management of solid manures. pages 67-107 *in*: E. Lichtfouse, ed. Agroecology and strategies for climate change. Sustainable agriculture reviews. Springer, Netherlands.

Well, R. and Butterbach-Bahl, K. 2010. Indirect emissions of N_2O from N deposition and leaching of agricultural N. Pages 162-189 *in*: Nitrous oxide and Climate Change, K. A., Smith, ed. Earthscan, London.

West, T. O. and Post, W. M. 2002. Soil organic carbon sequestration rates by tillage and crop rotation: a global data analysis. Soil Sci. Soc Am. J. **66**:1930-1940.

Weymann, D., Well, R., Flessa, H., von der Heide, C., Deurer, M., Meyer, K., Konrad, C. and Walther, W. 2008. Groundwater N_2O emission factors of nitrate-contaminated aquifers as derived from denitrification progress and N_2O accumulation. Biogeosci. **5**: 1215-1226.

Whalen, S. C. 2005. Biogeochemistry of methane exchange between natural wetlands and the atmosphere. Environ. Eng. Sci. **22**: 73–94.

White, T. A., Snow, V. O. and King, W. McG. 2010. Intensification of New Zealand beef farming systems. Agric. Syst. **103:** 21-35.

Whitman W. B., Bowen T. L. and Boone D. R. 2006. The methanogenic bacteria. Pages 165–207 *in:* M. Dworkin, ed. The prokaryotes, Springer-Verlag, New York, NY.

Wiens, M. J., Entz, M. H., Wilson, C. and Ominski, K. H. 2008. Energy requirements for transport and surface application of liquid pig manure in Manitoba, Canada. Agric. Syst. **98**: 74-81.

Wilkerson, V. A., Glenn, B. P. and McLeod, K. R. 1997. Energy and nitrogen balance in lactating cows fed diets containing dry or high moisture corn in ether rolled or ground form. J. Dairy Sci. **80**: 2487-2496.

Williams, A. G. and Coleman, G. S. 1997. The rumen protozoa. Pages 73-139 *in* P. N. Hobson and C. S. Stewart, eds. The Rumen Microbial Ecosystem. Blackie Academic and Professional, London, UK.

Wilson, C., Undi, M., Tenuta, M., Tremorin, D., Coppi, L., Flaten, D., Wittenberg, K. M. and Ominski, K. H. 2011. Utilization of liquid hog manure to fertilize grasslands in southeast Manitoba: Impact of application timing and forage harvest strategy on nutrient utilization and accumulation. Nutr. Cycling Agroecosyst. **91**: 155-171.

Wilson, C., Undi, M., Tenuta, M., Wittenberg, K. M., Flaten, D., Krause, D. O., Entz, M.

H., Holley, R. and Ominski, K. H. 2010. Pasture productivity, cattle productivity and metabolic status following fertilization of a grassland with liquid hog manure: A three-year study. Can. J. Anim. Sci. **90**: 2 33-243.

Wood, J. D., Gordon, R. J., Wagner-Riddle, C., Dunfield, K. E. and Madani, A. 2012. Relationships between dairy slurry total solids, gas emissions, and surface crusts J. Environ. Qual. 41: 694-704

Wright A. G. and Klieve, A. V. 2011. Does the complexity of the rumen microbial ecology preclude methane mitigation? Anim. Feed Sci. Technol. **166– 167:** 248– 253.

Yamulki, S., Jarvis, S. C. and Owen, P. 1999. Methane emission and uptake from soils as influenced by excreta deposition from grazing animals. J. Environ. Qual. **28***:* 676-682.

Yan, T., Agnew, R. E., Gordon, F. J. and Porter, M. G. 2000. Prediction of methane energy output in dairy and beef cattle offered grass silage based diets. Livest. Prod. Sci. **64**: 253-263.

Yang, W. Z., Beauchemin, K. A., Koenig, K. M. and Rode, L. M. 1997. Comparison of hull-less barley, barley, or corn for lactating cows: effect on extent of digestion and milk production. J. Dairy Sci. **80**: 2475-2486.

Yang, W. Z., Benchaar, C., Ametaj, B. N., Chaves, A. V., He, M. L. and McAllister, T. A. 2007. Effect of garlic and juniper berry essential oils on ruminal fermentation and on the site and extent of digestion in lactating cows. J. Dairy Sci. **90**: 5671-5681.

Zhu, J. S., Stokes, S. R. and Murphy, M. R. 1997. Substitution of neutral detergent fiber from forage with neutral detergent fiber from by-products in the diet of lactating cows. J. Dairy Sci. **80**: 2901-2906.