Growth-Enhancing Technologies: A Strategy to Reduce the Environmental Footprint of Canadian Beef Production

by

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ABSTRACT

An examination of the relationship between growth-enhancing technologies (GET's) and the environmental footprint of beef production systems revealed that cattle backgrounded and finished with GET's had 3 to 7% lower GHG emissions (kg CO_2e kg boneless beef^{-1}) and 3 to 8% lower NH_3 emissions (kg NH_3 kg boneless beef^{-1}). In addition, GET-treated cattle required 5 to 11% less land (ha kg boneless beef^{-1}) and 6 to 12% less water (m^3 H_2O kg boneless beef^{-1}) compared to GET-free cattle. These environmental impacts, along with economic viability and consumer preference and acceptance, must be assessed in a whole-system approach to determine the long-term sustainability of GET-free production in Canadian beef production.

FOREWORD

This thesis follows a manuscript style format based on the Canadian Journal of Animal Science guide for manuscript preparation. It consists of an abstract, introduction, materials and methods, results, discussion, and conclusion and has not been submitted for publication at this time. Animal production data used in this thesis was sourced from Ribeiro et al. (2020).

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DEDICATION

This thesis is lovingly dedicated to my daughter, Anna Préjet,

And to the hard-working farmers in my life, especially my dad, Brian Boonstra.

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ABBREVIATIONS

AAFC	Agriculture and Agri-Food Canada
ADG	Average daily gain
AET	Actual evapotranspiration
ATP	Adenosine triphosphate
β-AA	Beta-adrenergic agonist
β-AR	Beta-adrenergic receptor
BW	Body weight
CETA	Comprehensive Economic Trade Agreement
CFIA	Canadian Food Inspection Agency
CH_4	Methane
CO_2	Carbon dioxide
CO_2e	Carbon dioxide equivalent
CP	Crude protein
DDGS	Dried distiller's grains with solubles
DM	Dry matter
DMI	Dry matter intake
DOF	Days on feed
DP	Dressing percentage
EF	Emission factor
EF_{eco}	Ecodistrict emission factor
ET_c	Crop evapotranspiration
EU	European Union

FAO	Food and Agriculture Organization of the United Nations
FE	Feed efficiency
GE	Gross energy intake
GET	Growth-enhancing technology
GHG	Greenhouse gas
GWP	Global warming potential
ha	Hectare
HCON	Control heifer
HCON_AdjMGA	Control heifer adjusted to HMGA weights
HCON_AdjTBA	Control heifer adjusted to HTBA weights
HCW	Hot carcass weight
HGP	Hormonal growth promoters
HMGA	Melengestrol acetate-treated heifer
HTBA	Trenbolone acetate-treated heifer
IPCC	Intergovernmental Panel on Climate Change
K_c	Crop coefficient
$K_{c\ end}$	Crop coefficient, end
$K_{c\ init}$	Crop coefficient, initial
$K_{c\ mid}$	Crop coefficient, middle
LCA	Life cycle analysis
L_{dev}	Development phase
L_{init}	Initial phase
L_{late}	Late phase

L_{mid}	Middle phase
LW	Liveweight
MGA	Melengestrol acetate
N	Nitrogen
N_2O	Nitrous Oxide
NDM	National Drought Model
$NE_{activity}$	Net energy associated with activity
NE_{gain}	Net energy associated with gain
$NE_{maintenance}$	Net energy associated with maintenance
NH_3	Ammonia
NH_3-N	Ammoniacal nitrogen
P_2O_5	Phosphate
PET	Potential evapotranspiration
PI	Protein intake
PR_{gain}	Protein retained for growth
RAC	Ractopamine hydrochloride
REG	Ratio of dietary energy for gain to digestible energy consumed
REM	Ratio of dietary energy for maintenance
SCON	Control steer
SCON_AdjRAC	Control steer adjusted to SRAC weights
SCON_AdjTBA	Control steer adjusted to STBA weights
SRAC	Ractopamine hydrochloride-treated steer
STBA	Trenbolone acetate-treated steer

TAN	Total ammoniacal nitrogen
TBA	Trenbolone acetate
TDN	Total digestible nutrients
WUI	Water use intensity
ZH	Zilpaterol hydrochloride

1 GENERAL INTRODUCTION

The impact of cattle on environmental sustainability is multi-faceted, with both benefits and detriments associated with these complex production systems. Beef production contributes to climate change by emitting potent greenhouse gases (GHG's), utilizing natural resources such as water and land, with the potential for nutrient leaching from excreta (Vermeulen et al. 2012; Sheppard and Bittman 2012; FAO 2013; Smith et al. 2018). In 2011, it was estimated that Canadian cattle produced 12.0 kg carbon dioxide equivalents (CO_2e) kg liveweight $(LW)^{-1}$ (Legesse et al. 2016) and 18.4 kg ammonia (NH_3) animal^{-1} (Legesse et al. 2018b), with land requirements of 104 m^2 kg LW^{-1} and a water footprint of 7989 L kg LW^{-1} at slaughter (Legesse et al. 2018a). In contrast, cattle promote ecosystem biodiversity (Watkinson and Ormerod 2001; Pogue et al. 2018, 2020), and cattle-occupied grasslands sequester an estimated 0.6 gigatonnes CO_2e yr^{-1} globally (FAO 2013). Further, cattle convert human-inedible fibre from processing by-products and surplus food waste to high-quality proteins through natural digestive processes (White and Hall 2017). In the United States, it has been estimated that livestock recycle more than 43.2×10^9 of the plant resources and food waste from which humans can derive little nutritional value (White and Hall 2017). Finally, livestock production provides valuable nutrient inputs for cropping systems in the form of manure.

Increasing beef production while improving environmental sustainability, including a reduction in GHG emissions, NH_3 emissions, and natural resource use, is essential to meet the growing demand for animal-based proteins associated with population growth and globalization. Improvements in production efficiencies in the agriculture industry, including the beef sector, have occurred through continued research and developments in animal and crop genetics, biotechnology, and engineering (Piesse and Thirtle 2010; Brameld and Parr 2016; Balafoutis et al. 2017). Long-term improvements in production parameters (i.e., reproductive efficiency, average daily gain, carcass weight, and feed conversion) have resulted in an overall reduction in the environmental footprint of beef in Canada (Legesse et al., 2016, 2018a, 2018b). These data suggest that sustainable intensification, including the use of growth-enhancing technologies (GET's), which promote increased production efficiency and cattle performance (Ribeiro 2020), is an avenue by which the livestock sector can lower its environmental footprint.

However, consumer approval is a potential limiting factor for the continued use of GET's. In addition to the extrinsic cues (i.e., flavour, colour, marbling, and price) which impact the decision-making of consumers (Garmyn 2020), a feel-good factor from the food purchased and consumed is desired. Therefore, factors such as product source, sustainability, animal welfare, and management techniques, as well as whether the product is deemed "natural" and healthy are also considered. With data from social media platforms collected between January 2017 and January 2019, the Canadian Center for Food Integrity indicated that the number of Canadians discussing the relationship between agriculture and climate change, and agriculture and hormone-use exceeded 2.5 million and 950 thousand, respectively (Canadian Centre for Food Integrity 2019). Further, package labelling has a direct effect on purchasing decisions (Tait et al. 2018), and the abundance of labels present in the marketplace (i.e., "natural," "hormone-free," "free-range," "organic," "antibiotic-free") can increase the anxiety associated with decision-making (Capper 2013). Domestic demand for GET-free beef has increased; nearly a quarter of surveyed Canadians indicated a willingness to pay an additional 25% premium for hormone or antibiotic-free beef (PeopleTalking: Market Research Services 2012). In a global survey, 58% of consumers indicated a preference towards beef raised without the use of hormones or antibiotics compared to conventional beef (Nielsen Global Health and Ingredient-Sentiment Survey, 2016). It is important to note, however, that consumer preference survey results are not always an accurate reflection of in-store purchasing behaviour (Tait et al. 2018) but may still represent a growing consumer desire to see GET's eliminated from beef production systems.

Internationally, the European Union (EU) banned GET-treated beef imports in 1989 (Lusk et al. 2003), restricting the import of the majority of North American beef. Canadian producers must comply with EU regulations through the comprehensive "Canadian Program for Certifying Freedom from Growth Enhancing Products for the Export of Beef to the EU" (CFIA 2016), which grants Canadian producers partial access to the EU market if the requirements of the program are met. In 2017, the Comprehensive Economic Trade Agreement (CETA) between Canada and the EU established a tariff-rate quota for 50,000 tonnes of beef over a 5-yr phase-in (Global Affairs Canada 2017), providing an incentive for producers to raise GET-free cattle and federally inspected processing facilities to become certified to export beef to the EU.

While the domestic demand shift and increased international market access signify an excellent opportunity for Canadian beef producers and exporters, there is concern regarding the environmental impact of GET-removal due to a loss in production efficiency. Several studies have used a life cycle assessment (LCA) approach to estimate the impact of GET's on environmental sustainability in North America and have demonstrated that their use leads to a reduction in GHG emissions (Capper 2012; Basarab et al. 2012; Capper and Hayes 2012a; Stackhouse et al. 2012; Webb 2018), land (Capper 2012; Basarab et al. 2012; Capper and Hayes 2012), water (Capper 2012; Capper and Hayes 2012), and energy use (Capper 2012; Capper and Hayes 2012; Webb 2018). A limited number of studies (Cooprider et al. 2011; Stackhouse-Lawson et al. 2013) have directly measured the impact of GET-use on the environment in live animal trials with and without the use of GET's.

The objective of this study was to model the environmental footprint of beef production, including GHG and NH_3 emissions, as well as water and land use intensity, utilizing a dataset comprised of four trials with the same trial design, occurring over four years from a production study in which cattle were managed with or without the use of GET's. The outcomes of this research will provide science-based information for producers and consumers regarding GET-use on environmental sustainability. This is a critical first step in empowering producers to discuss on-farm management practices that have led to improved sustainability, as well as assisting consumers in the decision-making process regarding their food purchases and consumption.

2 LITERATURE REVIEW

2.1 Canadian beef industry

The Canadian beef industry contributes to the global food supply by exporting more than 400,000 tonnes of beef to 62 countries annually, valued at $3.1 billion (CCA 2020). Of the 193,492 farms across Canada, approximately 36,000 raise beef cattle (Statistics Canada 2015). Canadian beef production is diverse due to variations in landscapes across geographical areas, climatic zones and ecoregions, resources available for inputs, and management systems (Sheppard et al. 2015). There are three primary sectors, or operation types, that characterize beef production in the Canadian context: i) cow-calf, ii) backgrounding, and iii) finishing. A brief description of each follows:

Cow-calf operations consist of permanent herds of reproductively active females that ideally give birth annually. Canadian cow-calf operations have an average reproductive efficiency of 89% (89 calves born 100 cows^{-1}) and a death loss of approximately 3% (Canfax 2013). Calves are weaned between six and eight months of age (213 to 292 kg) and sold to either backgrounding or finishing operations (Sheppard et al. 2015). Cattle in this sector consume forage-based diets, grazing tame or native pasture for a 4-month period during the summer, and annually harvested forage or cropland for an additional 1.3 months (Sheppard et al. 2015).

Backgrounding is a relatively low-input intermediate stage of cattle production where weaned calves are overwintered and fed forage-based, moderately low-energy diets for a 140 to 150-d period (Sheppard et al. 2015). The primary goal of backgrounding is to feed the weaned calves to a sufficient size (~408 kg) to justify the advancement to finishing (Sheppard et al. 2015; The Canadian Cattlemen's Association, 2020).

Finishing operations utilize high energy diets to achieve the desired market weight using intensive and efficient feeding systems known as feedlots. The average daily gain (ADG) in the finishing phase is 1.4 and 1.3 kg d^{-1} for steers and heifers, respectively (Sheppard et al. 2015). Cattle finishing in Canada occurs primarily in feedlots, however, there are a small number of pasture-finishing operations in Canada, with the highest proportion in the Atlantic region (26%) and the lowest in the prairies (<4%, Sheppard et al. 2015). Almost 70% of feedlots in Canada are in Alberta, with an average capacity of 2,000 hd (Canfax 2020).

2.2 Improvements in animal efficiency

Growth-enhancing technologies (GET's) have been adopted in the cattle sector to improve efficiency and profitability through increased ADG and improved feed efficiency (FE; Johnson et al. 2013; Webb 2018). Ionophores, beta-adrenergic agonists (β-AA), and hormonal growth promoters (HGP) are among the products most frequently used in North America.

2.2.1 Beta-adrenergic agonists

Beta-adrenergic agonists are synthetic phenethenolamine compounds similar in shape and structure to the catecholamines, epinephrine and norepinephrine (Johnson et al. 2013). Catecholamines regulate glucose and free fatty acid levels, therefore playing an essential role in animal metabolism (NRC, 1994). To function, β-AA's must bind to the beta-adrenergic receptors (β-AR), which are present on nearly all mammalian cells. There are different types of receptors (i.e., β1-AR, β2-AR, and β3-AR), and the type and distribution within animal organs has an impact on the overall magnitude of metabolic responses (Mersmann 1998). Skeletal muscles and mammalian adipose tissue cells have a high abundance of β2-AR. The binding of β-AA to β2-AR activates the enzyme, adenyl cyclase which increases the production of cyclic adenosine monophosphate (Mersmann 1998). Cyclic adenosine monophosphate binds to protein kinase, resulting in protein phosphorylation and thereby increasing the transcriptional activity necessary to promote protein synthesis (Mersmann 1998; Thompson et al. 2016). Therefore, β-AA's increase muscle mass accumulation, as β-AR activation results in the regulation of pathways that control protein accretion (Johnson et al. 2013). In addition to the increase in the rate of protein synthesis and skeletal muscle mass, there is a reduction in the rate of protein degradation and a stimulation of lipolysis, resulting in a reduction in adipose tissue mass (Johnson et al. 2013).

The dose and duration of β-AA administration can impact cattle performance, muscle yield, and quality (Strydom 2016). Beta-adrenergic agonists have the most significant impact on cattle performance during the first few weeks after administration, but as time progresses, the magnitude of response slowly diminishes, and therefore, they are typically fed in the final 20 to 40-d of the finishing period (NRC, 1994; Strydom 2016).

The β-AA approved for use in the Canadian cattle industry is ractopamine hydrochloride (RAC; Strydom 2016). The approved RAC brands in Canada are Optaflexx 100, Actogain 100, Ractopamine 100, and Ractopamine 4 (CFIA 2012b). Recommended inclusion rates of RAC to improve FE and carcass leanness in cattle weighing > 400 kg are 20 to 30 mg kg^{-1} and 10 to 30 mg kg^{-1} RAC for administration 24 or 28-d before slaughter, respectively (CFIA 2012b). Zilpaterol hydrochloride (ZH) is no longer marketed to cattle, however it has been shown to improve ADG, FE, carcass leanness, and dressing percentage (DP; CFIA 2012c).

There is substantial evidence that β-AA's improve performance and carcass outcomes in finishing cattle (Mersmann 1998; Ribeiro 2020). For example, Scramlin et al. (2010) randomly assigned steers (n = 300) grouped by body weight (BW), body condition score, and breed type to one of the following treatments: 1) Control: no β-AA; 2) RH: 200 mg hd^{-1} RAC for 33-d; and 3) ZH: 75 mg hd^{-1} ZH for 30-d, with β-AA removed 3-d before the required withdrawal period. Feeding β-AA significantly increased final BW (546.62 kg, 549.75 kg, and 554.15 kg for control, ZH, and RAC, respectively) and improved FE (0.128, 0.131, and 0.107 for ZH, RAC, and control treatments, respectively). Further, hot carcass weight (HCW) was 12.8 kg and 5.3 kg greater in ZH and RAC-supplemented cattle compared to control. The ZH resulted in less total fat (14.19%) compared with RAC (15.78%) and control (15.68%), as well as increased ribeye area and yield, respectively (3.19%; 68.93%) compared to RAC (3.08%; 66.84%) and control (3.10%; 66.80%). However, marbling, skeletal maturity, lean maturity, and pH values did not differ significantly across treatments. Zilpaterol hydrochloride (6.89 kg) and RAC (5.36) had higher shear force values than the control treatment (4.66 kg), indicating that the use of β-AA may negatively impact tenderness (Scramlin et al. 2010).

A reduction in meat quality and tenderness are among the reasons ZH was removed from the NA market, which also included several animal health concerns (i.e., reduced DMI, heart rate, hoof health, and handling safety; Tucker et al. 2015). Further, concerns regarding the safety of consuming meat products containing residues of β-AA influenced the ban on ZH (Authority et al. 2016) and RAC use and sale in Europe and many other countries (Burnett et al. 2012). To export beef to a jurisdiction with regulations banning the presence of RAC, Canadian producers must comply with the "Canadian Beta Agonist-Free Beef Certification Program," an industry-driven initiative monitored by the Canadian Food Inspection Agency (CFIA; CFIA 2017).

2.2.2 *Hormonal growth promoters*

Natural and anabolic HGP's are used in the beef industry to enhance ADG, FE, and muscle growth in feedlot cattle (Keane and Drennan 1987; Pampusch et al. 2008). There are three natural HGP's approved for use in the cattle industry: progesterone, testosterone, and estradiol and three anabolic HGP's: trenbolone acetate (TBA), zeranol, and melengestrol acetate (MGA; Health Canada, 2012).

Route of administration: Implanted

Implants are small pellets made from a compressed powder containing varying amounts and combinations of HGP's and other compounds (i.e., tylosin tartrate, an antimicrobial used to reduce the risk of infection and abscesses at the administration site), which are administered subcutaneously between the cartilage ribs of the ear using an implant gun. The implant slowly dissolves, releasing the compounds into the bloodstream, and the HGP's are carried by special binding proteins to body tissues (Johnson et al. 2013). Implant products are available for use in each stage of beef production, with product selection based on animal age, breed, and, most importantly, gender.

There are 23 commercially available implants (estrogenic, androgenic, and combination products) in Canada, which are manufactured by three companies (Elanco, Merck Animal Health, and Zoetis). Estrogenic implants contain only estrogenic HGP's (estradiol and zeranol). The approved estrogenic implant products in Canada are Compudose, Component E-C, and Component E-S (Elanco Animal Health, Mississauga, ON), Ralgro (Merck Animal Health, Kirkland, QC), and Synovex S and Synovex C (Zoetis Canada Inc., Kirkland, QC). Zeranol stimulates the pituitary gland to increase somatotropin production, resulting in improved ADG and FE, and is safe for use in all ages and genders of beef cattle (Merck Animal Health n.d.).

Trenbolone acetate, a synthetic androgenic agent, mimics natural androgens such as testosterone (Duckett and Pratt 2014), leading to improved ADG, FE, and protein deposition, as well as increased lean meat and reduced carcass fat (Hancock et al. 1991). In heifers, TBA implants improve ADG, FE, final BW, and HCW, but have been shown to result in lower yield grades (Wagner et al. 2007). Combination implants contain both androgenic and estrogenic HGP (i.e., typically TBA and estradiol) in a single implant and have been shown to provide an additive response compared to single HGP products (Johnson et al. 2013). Commercial combination implants available for Canadian beef producers include several Synovex (Zoetis Canada Inc., Kirkland, QC; https://www2.zoetisus.com/products/beef/synovex-implants) products; i) Synovex Choice (100 mg TBA/14 mg estradiol benzoate), ii) Synovex Plus (200 mg TBA/28 mg estradiol benzoate) iii) Synovex One Feedlot (200 mg TBA/28 mg estradiol benzoate), iv) Synovex H (200 mg TBA/20 mg estradiol benzoate), and v) Synovex One Grass (150 mg TBA/21 mg estradiol benzoate extended-release implants). Other commercial combination implants include Component (Elanco Animal Health, Mississauga, ON; https://www.elanco.ca/products-services/beef) products; i) Component TE-100 with Tylan (100 mg TBA, 10 mg estradiol, and 29 mg tylosin tartrate), ii) Component TE-200 with Tylan (200 mg TBA, 20 mg estradiol, plus one pellet containing 29 mg of tylosin tartrate), iii) Component TE-H with Tylan (140 mg TBA, 14 mg estradiol USP, and 29 mg tylosin tartrate), iv) Component TE-S with Tylan (120 mg TBA, 24 mg estradiol, and 29 mg tylosin tartrate), v) Component TE-G with Tylan (40 mg TBA and 8 mg estradiol, plus one pellet containing 29 mg tylosin tartrate).

Route of administration: In feed

Melengestrol acetate, a synthetic progestin, is delivered as a feed additive (CFIA 2012a), as the product is orally active, and therefore implanting is not required (Johnson et al. 2013). The

MGA product approved for use in Canada is MGA 100 Premix, which is fed to feedlot heifers at a rate of 0.40 mg MGA hd^{-1} d^{-1} for growth stimulation, improved feed utilization, and estrus suppression (CFIA 2012a). Wagner et al. (2007) performed a meta-analysis to evaluate the effects of MGA with five treatments i) MGA only, ii) MGA and TBA implant, iii) MGA and estrogenic implant, iv) MGA and TBA plus estrogenic implants, and v) Control. Response to MGA differed by treatment; MGA only and heifers with MGA plus TBA implants had significantly higher ADG than combination and estrogenic implanted heifers. Heifers fed MGA with or without the addition of implants had increased HCW and carcass fat than MGA-free treatments but exhibited no improvement in grade (Wagner et al. 2007). Results of the analysis suggested that the effects of MGA on cattle performance were greater in non-implanted and TBA-implanted heifers compared to MGA use with a combination implant (i.e., TBA plus estrogenic implants).

2.2.3 ***Ionophores***

Ionophores, a class of antibiotic, are used in the North American cattle industry to enhance productivity and prevent coccidiosis (Cameron and McAllister 2016). Seven ionophores (monensin, lasalocid, salinomycin, narasin, maduramicin, laidlomycin, and semduramycin) are available globally for use in the livestock industry, with the exception of the EU (Novilla et al. 2017). Monensin is the most commonly used ionophore in North America. Commercially available ionophores (liquid or dry form) used in Canadian cattle diets include Rumensin, Bovatec, and Cattlyst, containing monensin, lasalocid, and laidlomycin propionate, respectively (Hersom and Thrift 2012; CFIA 2015).

Structurally, ionophores are large molecules with linear backbones that form rings around polar cations, resulting in lipid-soluble complexes that diffuse across lipid barriers of gram positive bacteria at a high rate (thousands $second^{-1}$; Pressman 1976). The most commonly used ionophores in the cattle sector are naturally occurring structures containing a single terminal carboxyl group.

As summarized by Pressman (1976), the mechanism of action of carboxylic ionophores is as follows: a positively charged ionophore inside a cellular membrane releases a proton, trapping the now negatively charged ionophore within the membrane. A complex cation, such as Na^+ or K^+, is bound by the negatively charged ionophore, therefore becoming a neutral zwitterion. The neutral zwitterion then diffuses across the membrane and exchanges the intercellular cation with a proton, and the cycle continues. Protons accumulate in the cell, shifting the overall charge and resulting in adenosine triphosphate (ATP) synthase activation by gram-positive bacteria to remove the protons from the cell and synthesize ATP (Thompson et al. 2016). The change in charge shifts the microbial population of the rumen, increasing propionic acid and decreasing acetic and butyric acid concentrations (Duffield et al. 2012; Hersom and Thrift 2012).

Duffield et al. (2012) conducted a meta-analysis reviewing 114 peer-reviewed papers and 203 trial reports to evaluate the effect of monensin (mean dose of 28.1 mg kg $feed^{-1}$) on dry matter intake (DMI), ADG, and FE in beef cattle (steers and non-replacement heifers). Monensin decreased DMI by 0.27 kg, increased ADG by 0.029 kg d^{-1}, and although the magnitude of the effect on FE was dose-dependent, monensin reduced feed:gain by an average of 0.53 kg of feed kg BW $gain^{-1}$.

A more recent meta-analysis by Cernicchiaro et al. (2016) compared the effects of two ionophores either alone or in combination with other compounds (laidlomycin with or without chlortetracycline and monensin with or without tylosin) on ADG, DMI, FE, and carcass characteristics in finishing beef steers. The inclusion of laidlomycin significantly improved ADG and DMI compared to monensin. However, there were no differences in FE based on liveweight (LW) and carcass-adjusted measurements between the treatments (Cernicchiaro et al. 2016).

2.3 Consumer perceptions of growth-enhancing technologies

Although there are proven productivity benefits of GET-use in the cattle industry, consumers have concerns regarding their use, which may impact their purchasing decisions and the future demand for conventionally raised beef (Webb 2018). Conventional food products are being out-marketed and out-advertised by items deemed “hormone-free” and “natural.” Special interest groups and promoting agencies further assist in altering consumer perceptions (Capper 2013) through marketing of GET’s as negative products, which has led many consumers to believe that hormone use in cattle production compromises food safety, animal care and results in adverse environmental outcomes. For example, ionophores have been proposed to pose a risk of promoting antimicrobial resistance even though there is no evidence that they promote resistance to medically important antimicrobials (Russell and Houlihan 2003). As ionophores are not used in humans, their use in cattle is unlikely to have an adverse effect on human health (Cameron and McAllister 2016).

A study of Canadian Conversations online on Food and Farming published by the Canadian Centre for Food Integrity reported that 950,000 Canadians discussed hormone usage in food production between January 2017 and January 2019 (Canadian Centre for Food Integrity 2019). In addition to hormone use, 2.5 million Canadians discussed the relationship between agriculture and climate change (Canadian Centre for Food Integrity 2019). These statistics are an indication that consumers are interested in the production techniques used to produce the food they purchase, as well as the role agriculture plays in climate change and the environment.

2.4 Environmental concerns associated with beef production

Several drivers influence consumer decisions regarding food choices, including source of product, welfare of the animal, taste, nutrition, health; with the environmental sustainability of beef production often foremost in the mind of many consumers. This concern primarily focuses on the role of greenhouse gas (GHG) emissions in climate change, as well as water use and nutrient release (nitrogen (N), phosphorus, and ammonia; NH_3) into the environment. Therefore, it is vital for the general public to understand and have access to science-based information to better inform their purchasing decisions.

2.4.1 ***Greenhouse gas emissions***

The primary GHG's associated with climate change include carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O).

<u>Carbon dioxide</u>

The naturally occurring processes of the carbon cycle includes the uptake of CO_2 by land vegetation during photosynthesis, and CO_2 production by metabolic processes within animals and the decomposition of organic matter by microorganisms (Rörsch et al. 2005). A significant source of anthropogenic (human-made) CO_2 production is the burning of fossil fuels (Rörsch et al. 2005; Yue et al. 2015).

Fossil fuels are non-renewable, depleting energy sources originating from organic matter. The main types of fossil fuels are coal, petroleum, and natural gas (Bhatia 2014). Fossil fuel use has increased immensely in the last century to meet global energy demands (Figure 2.1). In 2019, CO_2 emissions were responsible for 80% of all Canadian GHG emissions, and the majority was emitted as a result of fossil fuel combustion (Environment and Climate Change Canada 2021).

Globally, fossil fuels and industrial processes are responsible for approximately 80% of the total anthropogenic GHG emissions (Environment and Climate Change Canada 2010).

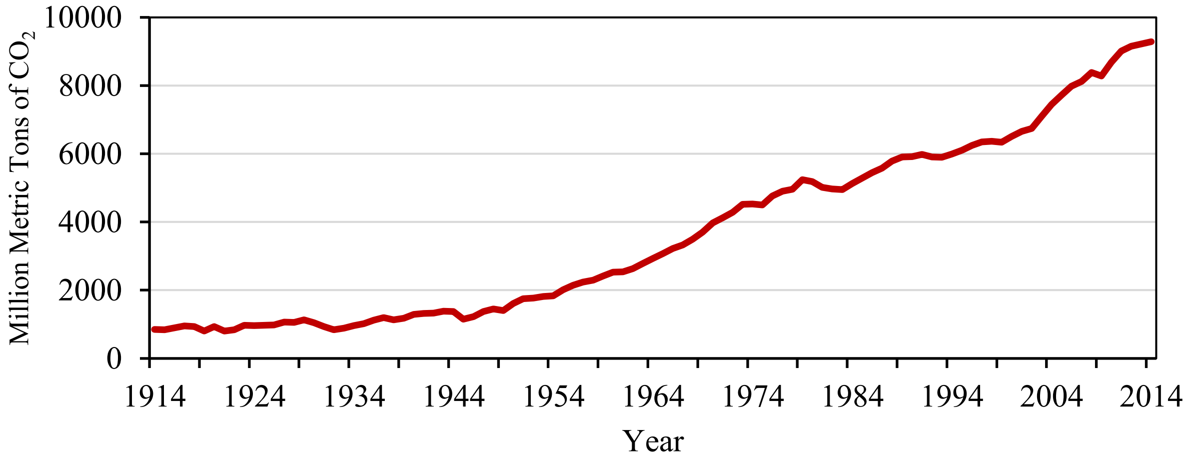

Figure 2.1 Total carbon emissions from fossil fuel consumption, adapted from Boden et al. (2017).

Energy generated from burning fossil fuels is used directly as a heat source (for spaces and process heating) or converted to energy for vehicles, industrial processes, and electrical power generation (Bhatia 2014). It is evident that CO_2 emissions would be substantially reduced if fossil fuel use was substituted with other forms of alternative, non-CO_2 emitting energy sources such as solar and wind (Khan et al. 2021).

<u>Methane</u>

Methane is one of the most abundant organic compounds on earth and is the main component of natural gas, a commonly used source of "clean-burning" energy (Basile 2013). As the second most common GHG in Canada, CH_4 is responsible for approximately 15% of Canada's total GHG emissions (Environment and Climate Change Canada 2019). The impact of atmospheric CH_4 emissions on climate change (referred to as its global warming potential; GWP^{100}) is 28 times more significant than CO_2 (IPCC 2021). However, the atmospheric lifespan of CH_4 is only a decade on average, compared to CO_2 which lasts thousands of years (US EPA 2015). Although GWP^{100}, is frequently used in the published literature to convert the emission rate to carbon dioxide equivalence (CO_2e; Legesse et al. 2016; Lynch et al., 2020), it does not consider differences in impact of CH_4 and CO_2 on climate change. More recently, the term GWP* has been introduced to measure and reflects the effect of half-life on atmospheric CH_4 concentrations (Lynch et al., 2020).

Biogenic and thermogenic sources of CH_4 are responsible for atmospheric emissions. Biogenic sources include those arising from microbial conversions in wetlands, landfills, thawing permafrost (due to climate change) and agricultural production, while thermogenic sources of CH_4 are associated with energy and industry (Allen 2016; IPCC 2021). Landfills, wastewater, animal waste management systems, coal mining, and oil and production facilities are the main anthropogenic CH_4 sources in Canada (Environment and Climate Change Canada 2019). Of the global CH_4 emissions, 50 to 65% are related to human activities (US EPA 2015).

Nitrous oxide

Nitrous oxide is the most significant ozone-depleting compound emitted to the atmosphere due to its potency (Cayuela et al. 2014) and has a GWP^{100} of 295 to 298 times CO_2 and an atmospheric lifetime of 114 years (US EPA 2015). Nitrous oxide emissions occur naturally through the N cycle and via anthropogenic activities such as fuel combustion, wastewater treatment, chemical production, and the agriculture industry (US EPA 2015). The application of synthetic and natural fertilizers in the production of crops is directly linked to N_2O emissions from the abundance of nitrogen N in the applied products (Venterea et al. 2012).

2.4.1.1 *The carbon footprint associated with Canadian beef production*

Vermeulen et al. (2012) reported that food systems account for 19 to 29% of global anthropogenic GHG emissions, of which 80 to 86% arise from agriculture. Agriculture accounts for 8.1% of total Canadian GHG emissions, comprised of those arising from agricultural soils (41%), crop residue burning (0.1%), enteric fermentation (41%), and manure management (13%; Environment and Climate Change Canada 2021).

Life cycle assessments (LCA) are used to estimate on-farm GHG emissions. Beauchemin et al. (2010) conducted a LCA of GHG emissions from beef production in Western Canada by simulating a 120-hd cow-calf to finish operation, which also included bulls and replacement heifers as well as cropland required to produce feed and pasture for grazing. To account for total lifetime GHG emissions from all animal classes in the operation, the LCA was conducted over an 8-yr cycle (Beauchemin et al. 2010). Using the GHG modeling software, Holos (www.agr.gc.ca/holos-ghg), the GHG intensity estimated for the period was 22 kg CO_2e kg^{-1} carcass weight. Enteric CH_4 was the largest source of GHG emissions, accounting for 63% of emissions. The breakdown of enteric CH_4 by animal type indicated that cows, bulls, calves, backgrounders, and finishers accounted for 79%, 3%, 2%, 7%, and 9%, respectively. Overall GHG emissions by animal class were 61% for the cow-calf herd, 19% for breeding stock, 8% for backgrounders, and 12% for finishers. Reducing CH_4 emissions is instrumental in reducing total GHG emissions, as the CH_4 produced by enteric fermentation has been reported to account for more than 60% (Beauchemin et al. 2010) and 73% (Legesse et al. 2016) of the total GHG emissions from cattle.

Legesse et al. (2016) evaluated and compared the environmental footprint of Canadian cattle production between 1981 and 2011. Over the 30 years, total GHG emissions from Canadian beef cattle production increased by 28%. However, a reduction of 18% (12.7 CO_2e in 2011 vs. 15.6 CO_2e in 1981) was measured on an intensity basis of CO_2 kg LW^{-1}. More specifically, CH_4, N_2O, and CO_2 emissions decreased by 18%, 19%, and 16%, respectively (Legesse et al. 2016) from 1981 to 2011.

A similar LCA study conducted by Alemu et al. (2017a) reported the GHG emissions from 295 cow-calf operations in Canada. The LCA results indicated a wide variation in emissions, as management style, geographical location, feed type, etc., significantly varied among farms. However, the mean GHG emission estimate from cow-calf operations was 23.9 kg CO_2e kg LW $sold^{-1}$ (median=23.4, ranging from 16.3 to 37.8). In the study, CH_4 (enteric and manure) accounted for 69% of the total GHG emissions and N_2O accounted for 24% of the total GHG emissions. Soil N_2O varied among farms, depending on the cropping system, as no emissions were produced on the farms which did not produce annual crops. Furthermore, on-farm energy use accounted for 5% of total GHG emissions (Alemu et al. 2017a). Differences in GHG emissions between scenarios were due to animal and land productivity, manure management (i.e., stockpiled manure resulted in lower emissions than composted or deep-bedded packs), and annual crop production (Alemu et al. 2017a). The authors noted that farms with the lowest-emissions did not necessarily have fewer animals, but used practices associated with increased productivity, including implementing an earlier calving season, a higher cull rate, and using more perennial forage than annual cropland in their feeding systems (Alemu et al. 2017a).

In the United States, a similar study was conducted by Capper (2011) to assess the change in environmental impacts of beef production between 1977 and 2007. The authors reported that the total carbon footprint was reduced by 16.3% over the 30-year timeframe, while CH_4 and N_2O emissions were reduced by 17.7% and 12%, respectively. In their study, Capper (2011) used production data from existing reports and databases to model comparisons between the two years, according to the methodology outlined in Capper et al. (2009). The reduction in GHG emissions arising from the Canadian beef industry have been attributed primarily to increased reproductive efficiency, ADG, slaughter weight, and crop yield (Legesse et al., 2016).

2.4.1.1.1 The Holos model: A method to estimate on-farm GHG emissions

Several of the Canadian studies described above estimated GHG emissions using Holos (www.agr.gc.ca/holos-ghg), an empirical whole-farm environmental modeling program developed by Agriculture and Agri-Food Canada. Holos inputs are on an annual time stamp and account for the cropping system, land use and management changes, as well as livestock production on a monthly basis (Alemu et al., 2017a). The model estimates CH_4, N_2O, and CO_2 emissions from enteric fermentation and manure, N leaching, N runoff from soils (crop, forage, pasture, range), and the re-deposition of volatilized ammonia, and on-farm energy use and herbicide and fertilizer manufacturing, respectively (Alemu et al. 2017a). The algorithms within the model are based on the Intergovernmental Panel on Climate Change (IPCC) Tier II methodology, altered to fit Canadian farming practices (Alemu et al. 2017a). The IPCC Tier II methodology framework encompasses "enhanced characterization for livestock populations," indicating that Tier I excluded such enhanced characterization (IPCC, 2006). To model emissions based on the IPCC Tier II methodology, production parameters are provided for each animal category (IPCC, 2006).

2.4.2 ***Water use***

Increased demand for water, a natural and depleting resource, can be attributed to pollution, climate change, and a growing world population (Arto et al. 2016). A national water footprint can be defined as the total volume of freshwater used to produce goods and services demanded by the nation's population wherever this water has been used (Arto et al. 2016). Further, water types within the associated footprint can be categorized; blue (i.e., surface and groundwater), green (i.e., rainwater), and grey (i.e., related to pollutant loads; Mekonnen and Hoekstra 2011a). On a per-capita basis, Canada's water use is among the largest globally (Arto et al. 2016).

2.4.2.1 The water footprint associated with Canadian beef production

Mekonnen and Hoekstra (2011b) evaluated the blue, green, and grey water footprint of farm animals and farm animal products. Beef cattle were among the eight animal categories evaluated and had the highest water footprint, accounting for nearly a third of the total global livestock water use footprint of 2422 $Gm^3\ yr^{-1}$ (Mekonnen and Hoekstra 2011b). To evaluate the water footprint of livestock, several components, including the indirect water footprint of feed, direct water footprint relating to drinking water, and service water, which is used to clean or maintain the animals' environment must be quantified, with results expressed as $m^3\ yr^{-1}\ hd^{-1}$ (Mekonnen and Hoekstra 2011b). The beef cattle water footprint for the live animal at the end of its life equated to 7477 $m^3\ ton^{-1}$ or 1889 $m^3\ hd^{-1}$ assuming an average animal weight of 253 kg (Mekonnen and Hoekstra 2011b). The authors noted that the largest share of the total water footprint is related to the feed production and therefore, when cattle move from a grazing system to a feedlot with improved feed conversion efficiency, and as animals reach slaughter weight faster, the water footprint is reduced. The global average water footprint associated with beef production was 15400 m^3 ton carcass yield^{-1} (94% green, 4% blue, 3% grey; Mekonnen and Hoekstra 2011b). The beef production water footprint, expressed on a nutritional basis, was 10.19 litres $kcal^{-1}$, 112 litres g protein^{-1} and 153 litres g fat^{-1} (Mekonnen and Hoekstra 2011b).

Mekonnen and Hoekstra (2011a) evaluated the water footprint of crops and derived crop products (i.e., grains, fibres) on a global scale using the calculation framework outlined in their water footprint manual (Hoekstra et al. 2009). The authors estimated the water footprint of 146 primary crops and more than 200 derived products. Evapotranspiration, climate parameters, crop characteristics, and soil water availability were among the most critical factors in evaluating the water footprint of crops. The authors expressed the green and blue water footprints as the total volume (m^3 yr^{-1}) per unit of crop yield (ton yr^{-1}). Grey water, which is an indication of freshwater pollution, was calculated as the fraction of N that was lost through leaching or as surface run off (Mekonnen and Hoekstra 2011a). Results indicated that the global crop production (from 1996 to 2005) water footprint was 7404 Gm^3 yr^{-1} (Mekonnen and Hoekstra 2011a). The majority of the water footprint was associated with green water (78%), while blue and grey water accounted for 12% and 10%, respectively (Mekonnen and Hoekstra 2011a). The authors noted that the water footprint of primary crops differed significantly among crop type and region. However, higher-yielding crops or crops with more significant biomass had a smaller water footprint than low yielding or small biomass crops. Crop categories such as cereals (0.51 l $kcal^{-1}$), oilseeds (0.81 l $kcal^{-1}$), and pulses (1.19 l $kcal^{-1}$) were evaluated with wheat, barley and corn requiring 1827, 1423 and 1222 m^3 ton^{-1}, respectively (Mekonnen and Hoekstra 2011a). Irrigated crops generally had higher yields than non-irrigated crops, and therefore the total water footprint of the crop was lower (2230 Gm^3 yr^{-1}; 48% green, 40% blue, 12% grey) than non-irrigated crops (5173 Gm^3 yr^{-1}; 91% green, 9% grey; Mekonnen and Hoekstra 2011a). Canada's average water footprint of crop production during the same period was estimated as 140 Gm^3 yr^{-1} (86% green, 1% blue, 13% grey; Mekonnen and Hoekstra 2011a).

Legesse et al. (2018a) examined changes in the water use intensity (WUI) of Canadian beef production over a 30-yr period. The authors estimated water requirements associated with animal consumption, feed production, and beef processing. The total water use, and water use $animal^{-1}$ yr^{-1} was 30% and 14% higher in 2011 compared to 1981, however, when expressed on an intensity basis (water use kg boneless beef^{-1}), there was a decrease of 17% in 2011 compared to 1981 (Legesse et al. 2018a). Bluewater use (in L kg LW^{-1}) decreased by 20% from 1981 to 2011 (Legesse et al. 2018a). Reduced water requirements on an intensity basis were attributed to improvements in reproductive efficiency, ADG, slaughter weight, and crop productivity (Legesse et al. 2018a).

2.4.3 Ammonia emissions

Ammonia is a colourless gas with a characteristically pungent scent. The primary source of NH_3 is through the interaction between urea and urease (the nitrogenous compounds) in urine and feces, resulting in the release of gaseous NH_3 (Neumeier and Mitloehner, 2013). Therefore, the waste products associated with animal production are a significant contributor to NH_3 emissions. Furthermore, NH_3 odour from livestock operations can be a nuisance to the public in close proximity to the farms or the waste holding reserves (i.e., lagoons). An abundance of NH_3 can create respiratory issues, reduce livestock performance and is associated with eutrophication in water bodies (Neumeier and Mitloehner, 2013; Sheppard and Bittman 2012). Ammonia can make its way to water bodies via nutrient runoff and leaching into the groundwater. Additionally, there is a loss of net revenue when NH_3 is lost because it is a source of N, a valuable crop input (Sheppard and Bittman 2012).

2.4.3.1 Ammonia intensity associated with Canadian beef production

Determining NH_3 emissions from cattle operations is complex, with several factors regulating NH_3 volatilization, including manure management, composition, and pH, as well as ambient conditions including temperature and wind speed (Hristov et al. 2011). A large portion of the emissions arising from cattle manure are directly linked to nutritional aspects, including the utilization of feed N in the rumen, overfeeding dietary N due to an inaccurate prediction of the degradable and undegradable protein requirements, and an underestimation of the role of urea recycling to the rumen (Hristov et al. 2011). Urea, the main source of manure NH_3, is the primary N constituent in ruminant urine, with the proportion increasing as the crude protein (CP) of the diet increases (Hristov et al. 2011).

Hristov et al. (2011) assessed the NH_3 emissions from beef feedlots and found that feedlot beef cattle excreted 80 to 90% of the total N consumed, therefore only retaining 10 to 20%. Atmospheric NH_3 can take multiple forms (gaseous, fine particulate, and liquid), and each phase is highly dependent on other atmospheric compounds present. The typical form of NH_3 from feedlots is gaseous and fine particulate (Hristov et al. 2011). Hristov et al. (2011) reported a broad range (3.6 to 88 $\mu g\ m^{-2}\ s^{-1}$) of NH_3 emissions arising from feedlot pen surfaces. Retention ponds and lagoons are additional sources of NH_3 emissions; however, the levels are inconsistent, and the range is unclear due to differences in system types (Hristov et al. 2011).

Legesse et al. (2018b) compared NH_3 losses from various sources of Canadian beef cattle production in 1981 and 2011. The authors used a mass balance approach based on the total ammoniacal N (i.e., TAN; the potentially highly volatile portion of the total N contained in animal manure) produced by cattle under confined vs. grazing conditions. Manure storage and application, and temperature were among the factors included in the analysis. Total N excreted was 6.17×10^5 ($Mg\ yr^{-1}$) in 2011 and 4.74×10^5 ($Mg\ yr^{-1}$) in 1981 and the excretion rate was 89% and 90%, respectively (Legesse et al. 2018b). Mature cows accounted for approximately half of the total N excreted, whereas backgrounding and finishing cattle were responsible for approximately 25%, and calves, bulls, and replacement heifers contributed the remainder (Legesse et al. 2018b). Total NH_3 emissions in 1981 and 2011 were estimated to be 1.42×10^5 and 1.79×10^5 Mg $NH_3\ yr^{-1}$, respectively (Legesse et al. 2018b). Approximately 40%, 28%, 21%, and 12% of NH_3 emissions in both study years were associated with manure in confinement, manure storage, land spreading of manure, and grazing, respectively (Legesse et al. 2018b). The total NH_3 emissions were less in 1981 than 2011 (16.0 vs. 18.4 kg NH_3 hd^{-1} yr^{-1}, respectively), however emissions expressed on an intensity basis (kg NH_3 kg $beef^{-1}$) decreased by 20% from 1981 to 2011 in Canada. The authors attributed the observed reduction to improvements in production efficiencies and noted that these improvements might be negated with the increased levels of substituting grains in finishing and backgrounding diets with protein-dense by-products such as dried distillers grains (DDGS; Legesse et al. 2018b).

In Alberta, McGinn and Flesch (2018) evaluated feedlot environmental impacts and measured NH_3 using inverse dispersion methods. Two sites at a commercial feedlot in Alberta were used for the analysis, and the average daily NH_3 emissions hd^{-1} were between 100 and 117 g (McGinn and Flesch 2018). These authors indicated that the emission rates represented 39 to 37% of the fed N in the feedlot system.

2.5 Environmental impact associated with growth-enhancing technology removal from cattle production systems

As described above, advancements in genetics, nutrition, and crop production have led to improved productivity and environmental sustainability (Capper and Bauman 2013; Legesse et al. 2016, 2018a, 2018b; Neumeier and Mitloehner 2013). Researchers in Canada and the United States have examined the impact of GET-use on the environmental footprint of beef in animal trials. Several studies have used a LCA approach garnering data from published literature to estimate the impact of GET's including ionophores, HGP's, and β-AA's, on the environmental sustainability of beef cattle in North America, and have found that their use reduces GHG emissions anywhere from 1 to 40% (Capper 2012; Basarab et al. 2012; Capper and Hayes 2012; Stackhouse et al. 2012;

Webb 2018), land use from 8 to 45% (Capper 2012; Basarab et al. 2012; Capper and Hayes 2012), water use from 1 to 75% and energy use from 1 to 29% (Capper 2012; Capper and Hayes 2012; Webb 2018). Similarly, animal production trials with and without the use of GET's have demonstrated a 10 to 16% reduction in CH_4 emissions (Stackhouse-Lawson et al. 2012) and an 8 to 30% reduction in NH_3 emissions (Stackhouse-Lawson et al. 2012; Stackhouse et al. 2012). As discussed by Capper and Hayes (2012), increased growth rates due to GET-use reduced fixed costs and had a "dilution of maintenance" effect (Capper and Hayes 2012). Additionally, N and phosphorus excretion were reduced by 9.8% and 10.6% with GET-use, respectively.

2.6 Conclusion

Canadian consumers and producers share a common value: access to a safe, healthy, nutrient-rich diet that is produced sustainably. While the removal of GET's may allow for additional markets to be explored domestically and internationally, environmental consequences may arise with the removal or reduction of their use. An assessment of the environmental implications of removing these products is necessary to provide science-based information that will help consumers make informed decisions about the food they eat.

3 HYPOTHESES AND OBJECTIVES

3.1 Hypothesis

The environmental footprint of backgrounding and finishing beef cattle in a feedlot production system will be reduced with the use of growth-enhancing technologies (GET's). Environmental footprint indices include: i) greenhouse gas emissions (carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O)), ii) land use, iii) water use, and iv) ammonia (NH_3) emissions.

3.2 Objectives

Estimate and compare the environmental impact of backgrounding and finishing cattle managed with or without the use of GET's using a 4-trial data set comprised of animal productivity (i.e., animal average daily gain, dry matter intake, feed efficiency, carcass outcomes), diet composition, and housing garnered from a study conducted in Lethbridge, Alberta in addition to region specific crop yields and climatic data to assess:

i. Greenhouse gas emissions associated with animal and crop production;

ii. Land use associated with the production of the feed required to supply the nutritional needs of cattle;

iii. Water use associated with crop production, animal servicing, and processing plant operations, and;

iv. Ammonia emissions associated with animal housing, manure storage, and manure land application.

4 MANUSCRIPT

ENVIRONMENTAL IMPACTS ASSOCIATED WITH THE REMOVAL OF GROWTH-ENHANCING TECHNOLOGIES FROM CANADIAN FEEDLOTS: A CASE STUDY

4.1 ABSTRACT

Greenhouse gas (GHG) and ammonia (NH_3) emissions, as well as land and water requirements from feedlot cattle backgrounded and finished with or without the use of growth-enhancing technologies (GET's), were estimated using several models, including Holos, a whole-farm GHG emission model (www.agr.gc.ca/holos-ghg). Model inputs were obtained from a multi-year study (n= 4 trials) which evaluated the performance of feedlot heifers (H) and steers (S) with six management scenarios/treatments (n = 40 hd treatment^{-1} trial^{-1}): 1) H control (HCON); 2) H implanted (HTBA); 3) H supplemented with melengestrol acetate (HMGA); 4) S control (SCON); 5) S implanted (STBA); and 6) S implanted and supplemented with ractopamine hydrochloride (SRAC; conducted in the last two years). All cattle were finished to achieve a consistent number of days on feed (DOF; n = 233 ± 8). Lighter finish weights were observed for HCON and SCON, therefore, DOF of HCON and SCON were adjusted (-1 to 65-d) to achieve the same final weight as GET cattle, resulting in four additional treatments (HCON_AdjTBA, HCON_AdjMGA, SCON_AdjTBA, and SCON_AdjRAC). The GHG emissions (kg CO_2e kg boneless beef^{-1}) from HTBA, HMGA, STBA, and SRAC were 4%, 3%, 6%, and 7% lower than the respective weight-adjusted control cattle, respectively. Similarly, the land required (ha kg boneless beef^{-1}) was reduced by 7%, 5%, 10%, and 10% for HTBA, HMGA, STBA, and SRAC, respectively, compared to their respective weight-adjusted control cattle. Water requirements (m^3 H_2O kg boneless beef^{-1}) were reduced by 6%, 5%, 11%, and 12% for HTBA, HMGA, STBA, and SRAC compared to the respective weight-adjusted control cattle. Furthermore, NH_3 emissions (kg NH_3 kg boneless beef^{-1}) from GET cattle (HTBA (1.53×10^{-1}), HMGA (1.50×10^{-1}), STBA (1.40×10^{-1}), and SRAC (1.41×10^{-1})) were lower than HCON_AdjTBA (1.60×10^{-1}), HCON_AdjMGA (1.54×10^{-1}),

SCON_AdjTBA (1.52×10^{-1}), and SCON_AdjRAC (1.52×10^{-1}). This study demonstrates that conventional beef production systems have a lower environmental footprint than "natural" beef production systems – information which is paramount for consumers, producers and policy makers who endeavour to reduce emissions and meet targets necessary to realize a net zero carbon economy.

4.2 INTRODUCTION

The Food and Agriculture Organization of the United Nations (FAO) estimates the global population to increase to 9.1 billion people by 2050 (FAO, 2009). Avenues by which food security can be achieved in concert with the growing global population include: i) investments in sustainable agricultural production and rural development, ii) technology and productivity growth, and iii) support to farmers, trade, and markets (Rockström et al. 2017).

Evaluating the environmental footprint (i.e., greenhouse gas and ammonia emissions, land use, water use, and nutrient excretion) of cattle production systems is a necessary step to assess impact and identify strategies to improve environmental sustainability of Canadian agro-ecosystems. Of the total Canadian greenhouse gas (GHG) emissions in 2019, livestock and crop production accounted for approximately 8.1%, of which enteric methane (CH_4) emissions were responsible for 41% (Environment and Climate Change Canada 2021). Additional environmental implications associated with cattle production include natural resource use (i.e., land and water) and volatilization and leaching of nitrogenous products (i.e., ammonia; NH_3) from cattle waste. Previous modeling studies which estimated the environmental impact associated with the cattle industry in several countries have been conducted (Beckett and Oltjen 1993; Hristov et al. 2011; Capper 2011; Stackhouse-Lawson et al. 2012; White and Hall 2017), including a number of Canadian studies (Ominski et al. 2007; Beauchemin et al. 2010; Legesse et al. 2016; Alemu et al. 2017a; Legesse et al. 2018a, 2018b).

Potential avenues to reduce the environmental footprint associated with beef production, observed through modeling, include genetic improvements (Basarab et al. 2013), variation in management strategies (i.e., conventional, natural, and grass fed; Capper 2012), and diet modification (Cayuela et al. 2014; Guyader et al. 2016). Additional mitigation strategies examined through modeling include a reduced backgrounding period, dietary changes (i.e., inclusion of oilseeds, higher quality forages, and dried distillers grain; DDGS), and alterations in herd and farm management (i.e., increased breeding stock longevity, increased number and weight of weaned calves, and converting crop land to pasture; Beauchemin et al. 2011).

Improvements in animal production efficiency (i.e., average daily gain (ADG) and feed efficiency; FE) have occurred via genetic advancements and improved management systems, including the use of growth-enhancing technologies (GET's; Brameld and Parr 2016). Despite the demonstrated value of GET's, there has been a shift in demand towards "free-from" products (i.e., free from GET's or antibiotics) in domestic and global markets (Webb et al. 2017; Yang et al. 2020; Garmyn 2020). However, consumers are mainly unaware of the implications of their food choices regarding GET's on environmental sustainability and future food security. Efficiency lost from removing GET's (Ribeiro et al., 2020) may make it increasingly challenging to feed a growing human population, with negative consequences for the environment. Several existing studies from the United States and Canada used published data from single production studies or input data from a wide range of studies to model or measure the effects of the removal of GET's on the environmental footprint of beef production. Aboagye et al. (2021) summarized the environmental impacts associated with GET's in beef production systems and reported a 1 to 40% reduction in GHG emissions, a 7.8 to 44.7% reduction in land, a 1 to 75% reduction in water, and 8 to 30% reduction in NH_3 emissions. The objective of the current study was to compare the environmental footprint (i.e., GHG and NH_3 emissions, land use, and water use) of Western Canadian feedlot cattle raised using conventional production strategies with or without GET's using data collected over a 4-yr period with the same feeding strategy and trial design in each year.

4.3 MATERIALS AND METHODS

4.3.1 Trial design

4.3.1.1 Description of experimental site and climate

The trials from which the data were garnered were conducted at the Agriculture and Agri-food Canada (AAFC) Research Development Centre located in Lethbridge, AB. The site description has previously been described by Ribeiro et al. (2020). Additional site information including soil characteristics, soil correlation area, ecodistrict, agroclimate, and subregion was obtained (Alberta Soil File Information Center 2016) and characterized as 3 (Dark brown soil zone of South-Western Alberta), 793 (moist mixed grassland), 2A, and mixed-grass, respectively. Long term (1991 to 2020) average monthly temperatures and growing season precipitation (May to October; 260 mm) were obtained from the Lethbridge weather station (Table 4.1).

Table 4.1 Monthly mean and long-term average (1991 to 2020) temperature and growing season (May to October) precipitation (2015 to 2018) in Lethbridge, AB[a]

Month	Mean temperature (°C)					Mean growing season precipitation (mm)				
	2015	2016	2017	2018	Long-term Ave.	2015	2016	2017	2018	Long-term Ave.
January	-2.4	-6.2	-5.0	-2.3	-6.0	–	–	–	–	–
February	-3.2	2.6	-2.5	-13.0	-5.0	–	–	–	–	–
March	3.6	3.9	0.5	-4.6	-0.4	–	–	–	–	–
April	5.9	8.0	6.1	2.2	5.4	–	–	–	–	–
May	9.4	10.8	12.7	14.2	10.7	29.3	67.5	41.1	25.1	54.8
June	16.9	16.4	16.1	15.8	14.8	13.4	12.8	28.3	45.8	81.9
July	18.3	18.3	20.4	18.1	18.1	39.3	32.4	7.3	13.6	36.7
August	18.2	17.6	18.7	18.0	17.6	16.1	30.1	10.8	21.5	32.2
September	12.2	13.5	13.8	10.3	13.0	39.5	19.4	0.0	19.1	32.3
October	8.7	6.1	6.2	5.7	6.2	7.1	14.2	38.7	14.6	22.3
November	-1.5	5.5	0.5	0.4	-0.8	–	–	–	–	–
December	-5.2	-11.9	-3.5	-2.8	-5.9	–	–	–	–	–
Total	–	–	–	–	–	144.7	176.4	126.2	139.7	260.2

[a] Values obtained from Environment and Climate Change Canada (2020) for the Lethbridge weather station, [Online] Available: https://climate.weather.gc.ca/prods_servs/cdn_climate_summary_e.html

4.3.1.2 Description of the animal production system

The four backgrounding and finishing trials were conducted over four consecutive years from 2015 to 2018 with 120 heifers and 80 steers included each year for the first two trials and an additional 40 steers included in the final two years. The production cycle in each of the four trials began in the fall and ended the following summer with timelines as follows: trial 1: November 19, 2015, to August 5, 2016; trial 2: October 27, 2016 to July 2, 2017; trial 3: October 31, 2017 to July 15, 2018, and trial 4: December 11, 2018 to September 7, 2019 (Table 4.2). Cattle were allocated into pens (n=10 hd pen^{-1}), separated randomly by weight and sex, backgrounded for the first 84-d, followed by a 28-d transition period, and finished during the remaining 148 ± 5-d of the production cycle.

Table 4.2 Number of days in each month that steers and heifers were on feed during the backgrounding and finishing phases of the four production trials conducted in Lethbridge, AB

	Trial 1	Trial 2	Trial 3	Trial 4
Month	Days			
January	31[a]	17[a]	22[a]	31[a]
February	10[a]	15[b]	10[b]	28[a]
March	24[b]	31[b]	31[b]	4[a]
April	30[b]	30[b]	30[b]	30[b]
May	31[b]	31[b]	31[b]	31[b]
June	30[b]	30[b]	30[b]	30[b]
July	31[b]	2[b]	15[b]	31[b]
August	5[b]	–	–	31[b]
September	–	–	–	7[b]
October	–	6[a]	1[a]	–
November	12[a]	30[a]	30[a]	–
December	31[a]	31[a]	31[a]	21[a]
Total	84[a]	84[a]	84[a]	84[a]
	151[b]	139[b]	147[b]	160[b]

[a] Days occurring in the backgrounding phase

[b] Days occurring in the finishing phase

Six treatments were examined (40 hd treatment^{-1} yr^{-1}): 1) control heifers (HCON); 2) implanted heifers (HTBA); 3) heifers receiving melengestrol acetate (HMGA); 4) control steers (SCON); 5) implanted steers (STBA); and 6) steers receiving implants + ractopamine hydrochloride (SRAC; conducted in the last two years; Ribeiro et al. 2020). The implanting protocol for HTBA, STBA, and SRAC consisted of three implants (Component TE-100; Elanco Animal Health, Mississauga, ON; 100 mg trenbolone acetate + 10 mg estradiol USP + 29 mg of tylosin tartrate) at two 90-d intervals, followed by an additional implant during finishing

(Component TE-200, Elanco Animal Health, Mississauga, ON; 200 mg trenbolone acetate + 20 mg estradiol USP + 29 mg of tylosin tartrate). Melengestrol acetate (MGA 100 Premix; Zoetis Canada Inc., Kirkland, QC) was included in HMGA diets at a rate of 0.40 mg heifer^{-1} d^{-1}. Ractopamine (Optaflexx; Elanco Animal Health, Mississauga, ON) was included in addition to the implant protocol for SRAC at a rate of 30 mg RAC kg of total diet^{-1} during the last 42-d before slaughter. A summary of the treatments is described in Table 8.1 of the Appendix.

Cattle were fed with slick bunk management and diets consisted of corn silage, barley grain, corn DDGS, and a mineral supplement, with phase-specific inclusion rates (Table 4.3). Feed samples were collected daily and analyzed for each period. Crude protein (CP; kg kg^{-1} diet) and total digestible nutrients (TDN; %) values were pooled by phase and reported (Table 4.4). Daily feed intake (kg pen^{-1} d^{-1}) was recorded as total dry matter intake (DMI; kg dry matter phase^{-1}) and as individual feed ingredients, pooled within treatment (Appendix Table 8.2 and 8.3). An ionophore (Rumensin; Elanco Animal Health, Mississauga, ON; 33 mg kg dry matter (DM)$^{-1}$) was included in all diets, including HCON and SCON.

Table 4.3 Backgrounding and finishing dietary inclusion rates (% DM) during the beef cattle production trials conducted in Lethbridge, AB[a]

	Backgrounding diet	Finishing diet
Ingredient	Inclusion rate (DM, %)	
Corn silage	60	9
Barley grain	20	80
Corn DDGS	15	6
Mineral	5	5

[a] Ribeiro et al., (2020).

Table 4.4 Crude protein (CP) and total digestible nutrient (TDN) analysis of backgrounding and finishing diets (DM) with and without melengestrol acetate (MGA) delivered to cattle in the four production trials conducted in Lethbridge, AB[a]

	CP, $kg\ kg^{-1}$				TDN, %			
Item	Trial 1	Trial 2	Trial 3	Trial 4	Trial 1	Trial 2	Trial 3	Trial 4
Backgrounding diet								
No-MGA	0.133	0.133	0.140	0.122	69	68	66	73
With MGA[b]	0.132	0.136	0.136	0.132	74	74	73	79
Finishing diet								
No-MGA	0.137	0.132	0.132	0.121	69	68	68	72
With MGA	0.136	0.136	0.136	0.134	75	74	73	79

[a] Ribeiro et al., (2020)

[b] Diets containing MGA were only fed to heifers.

As described by Ribeiro et al. (2020), body weight (BW) was recorded at the end of each period in the backgrounding (n=4) and finishing (n=5) phases, and ADG was estimated for each phase (Appendix Table 8.2 and 8.3). The HCON and SCON treatments had lower final weights than the GET-treated cattle, as all groups were fed for 233 ± 8 DOF. The final weights of HCON and SCON cattle were lower than GET cattle. To compare control and GET-treatments on the same basis, the number of additional DOF required to achieve the same finished weight as the GET cattle were estimated and added to the finishing period within each trial year. More specifically, HCON was adjusted to HTBA and HMGA finishing weights, resulting in HCON_AdjTBA and HCON_AdjMGA, respectively, and SCON was adjusted to STBA and SRAC finishing weights, resulting in SCON_AdjTBA and SCON_AdjRAC, respectively. Control cattle dressing percentage (DP) was applied to the control-adjusted cattle values, as there was no means to adjust the DP without speculation. The resulting difference in DOF varied from -1 to 65-d (Appendix Table 8.4).

At the end of each finishing period, cattle were processed at Cargill Ltd., located in High River, AB. Carcass characteristics, including hot carcass weight (HCW) and DP, were summarized by treatment and year (Appendix Table 8.5 and Table 8.6). Carcass weight was used to estimate boneless beef (kg) as described in Eq. 1 (Mekonnen and Hoekstra, 2010; AAFC 2013).

$$\textit{Boneless beef} = DP \times \textit{slaughter weight} \times 0.71 \qquad (1)$$

Where:

Boneless beef = the lean meat associated with a carcass, kg;

DP = average dressing percentage, %;

0.71 = used to convert from carcass weight to boneless beef.

The average carcass weight and DP of the cattle in each treatment within each year were employed in the equation.

4.3.1.3 Description of agronomic inputs and land use required for feed production of backgrounded and finished cattle

Total land required (ha treatment^{-1}) to produce feed for cattle in the production trials was calculated using the ingredient inclusion rate (% DM), total DMI (kg hd^{-1} d^{-1}), the average DM of the feed ingredient (%; Table 4.5), as well as storage and feeding losses (Rotz and Muck 1994; Table 4.5). Estimates of crop yields were based on a 4-yr average (2015 to 2018) and were expressed as kg ha^{-1}, as fed (Table 4.5). Finally, DOF and number of cattle pen^{-1} (n= 10) were used to calculate the land requirements over the duration of the trial for each treatment (Appendix Table 8.7 and 8.8). Any necessary conversions were implemented to ensure that comparative values had equivalent units.

Table 4.5 Yield, dry matter (DM, %), harvest and storage loss, and fertilizer inputs associated with production of the feed ingredients included in backgrounding and finishing cattle diets

Items	Yield ($kg\ ha^{-1}$ as fed)	DM (%)[a]	Storage loss (%)[b]	Land applied nitrogen ($kg\ N\ ha^{-1}$)	Land applied phosphorus ($kg\ P_2O_5\ ha^{-1}$)
Corn silage	56338.8[c]	35[c]	5	205.3[c]	125.3[c]
Barley grain	5743.1[d]	87[a]	0	72.9[e]	33.6[e]
Corn grain	2452.7[f]	89[a]	0	168.1[e]	67.3[e]

[a]Feedipedia: https://www.feedipedia.org/

[b]Rotz and Muck (1994)

[c]Gideon Stoutjesdyk, personal communication

[d]Area-specific barley yields were 5165 $kg\ ha^{-1}$, 5219 $kg\ ha^{-1}$, 6187 $kg\ ha^{-1}$, and 6402 $kg\ ha^{-1}$ for trial 1, trial 2, trial 3, and trial 4, respectively, resulting in a mean yield of 5743 $kg^{-1}\ ha^{-1}$ (Agriculture Financial Services Corporation 2019)

[e]Agri-Facts (2004)

[f]Area-specific grain corn yield was not available; therefore, provincial data was sourced (Trial 1: 2054 $kg\ ha^{-1}$, Trial 2: 2452 $kg\ ha^{-1}$, Trial 3: 2549 $kg\ ha^{-1}$, and Trial: 2755 $kg\ ha^{-1}$), with an average yield of 2453 $kg\ ha^{-1}$ (Statistics Canada. Table 32-10-0359-01) which was converted to corn DDGS (Blaschek et al. 2016; Mackenzie Zimmerman, 2020)

Annual fertilizer requirements for local silage production were met by dairy manure (93540 $l\ ha^{-1}\ yr^{-1}$) sourced from a local dairy operation with an average nutrient analysis of 180-100-280 for N, P, and K, respectively (Gideon Stoutjesdyk, personal communication). Additional corn silage nutrient requirements were met using Alpine G22 (6-22-2; https://www.alpinepfl.com/product/alpine-g22/;), a liquid fertilizer, at a rate of 47 litres ha^{-1} (Gideon Stoutjesdyk, personal communication). There was no specified fertilizer application method or source for corn grain and barley grain, however it was assumed that rates that met the N and P requirements were applied (Table 4.5).

Corn grain was not a direct component of the diet, but the diet did contain corn DDGS. It was assumed that the corn grain needed to produce DDGS was grown and harvested on the farm and transported to a local ethanol plant, with 1 tonne of corn producing 303.5 kg corn DDGS

(Blaschek et al. 2016; Mackenzie Zimmerman, 2020). Thereafter, DDGS were transported to the farm to be used as a feed source. Co-product allocation and transportation was not considered as it was deemed to be beyond the scope of this study. Therefore, co-allocation to ethanol production was not considered.

Finally, it was assumed that all crops were irrigated and grown conventionally with herbicide, however specific rates and application methods were not included in the description.

4.3.2 Estimating greenhouse gas emissions and land use

4.3.2.1 Description of the model used: Holos (Version 3.0.6)

Holos, a whole-farm emissions software developed by AAFC (www.agr.gc.ca/holos-ghg), which employs IPPC Tier II algorithms modified to account for Canadian conditions and agricultural practices, was used to estimate on-farm GHG emissions (Little et al. 2008). The model has previously been used to assess the carbon footprint associated with agricultural management practices in Canada (Beauchemin et al. 2010; Guyader et al. 2017; Alemu et al. 2017b; Little et al. 2017). Further, the algorithms in the model have been used to examine the environmental impact of changes in management practices over time on GHG emissions (Legesse et al. 2016).

In the current study, GHG emissions estimated from the Holos model included on-farm emissions of i) CH_4 arising from enteric fermentation and manure decomposition, ii) nitrous oxide (N_2O; direct) from cropping, and iii) carbon dioxide (CO_2) from energy use. Crop input manufacturing, such as fertilizers and pesticides, and indirect emissions of N_2O from N leaching and volatilization were also included in the model.

Methane from enteric fermentation was calculated based on the CH_4 conversion factor of the diet (unadjusted), TDN, and CP content of the diet. Manure CH_4 emissions were calculated based on the manure handling system and the associated conversion factor.

Direct N_2O emissions from soil and cropping were estimated by multiplying total N inputs in the cropping system and the direct N_2O EF from crops and soils for each specific ecodistrict (EF eco). Adjustments within the model accounted for climatic variables (growing season precipitation and potential evapotranspiration), soil variables (i.e., soil type and texture), tillage intensity, and topography. Monthly variables considered included: i) average temperature, and ii) soil N_2O conversion; the annual soil N_2O emissions allocated to each month (%). Indirect N_2O emissions included N leaching and volatilization fractions (i.e., amount of N lost to runoff, leaching, and volatilization). Direct N_2O emissions from manure were estimated using the emission factor (EF; kg N_2O-N (kg N)$^{-1}$) associated with the manure handling system.

Fossil fuels and energy required for crop production (i.e., machinery use), animal feeding, and manure handling were the primary sources of on-farm energy used to calculate CO_2 emissions. The secondary on-farm energy use included CO_2 emissions associated with crop inputs manufacturing (i.e., herbicides, fertilizers, etc.).

Estimated GHG's, expressed in carbon dioxide equivalents (CO_2e), were calculated as described in the 2006 Intergovernmental Panel on Climate Change (IPCC) Guidelines for National Greenhouse Gas Inventories (IPCC, 2006); the global warming potential (GWP^{100}) of the gas is multiplied by the total emissions of the gas. The GWP^{100} for each GHG are as follows: CO_2: 1, CH_4: 28, and N_2O: 265 to 298 (IPCC, 2021). Values of 23 and 296 were used for CH_4 and N_2O, respectively, as described in Holos model Version 3.0.6. System boundary and scope

A partial lifecycle assessment (LCA) was conducted to quantify whole-farm GHG emissions (Figure 4.1) encompassing the feedlot (backgrounding and finishing) and associated inputs/outputs. When direct data from the yearly experiments were unavailable, default values within Holos and information from published literature were used.

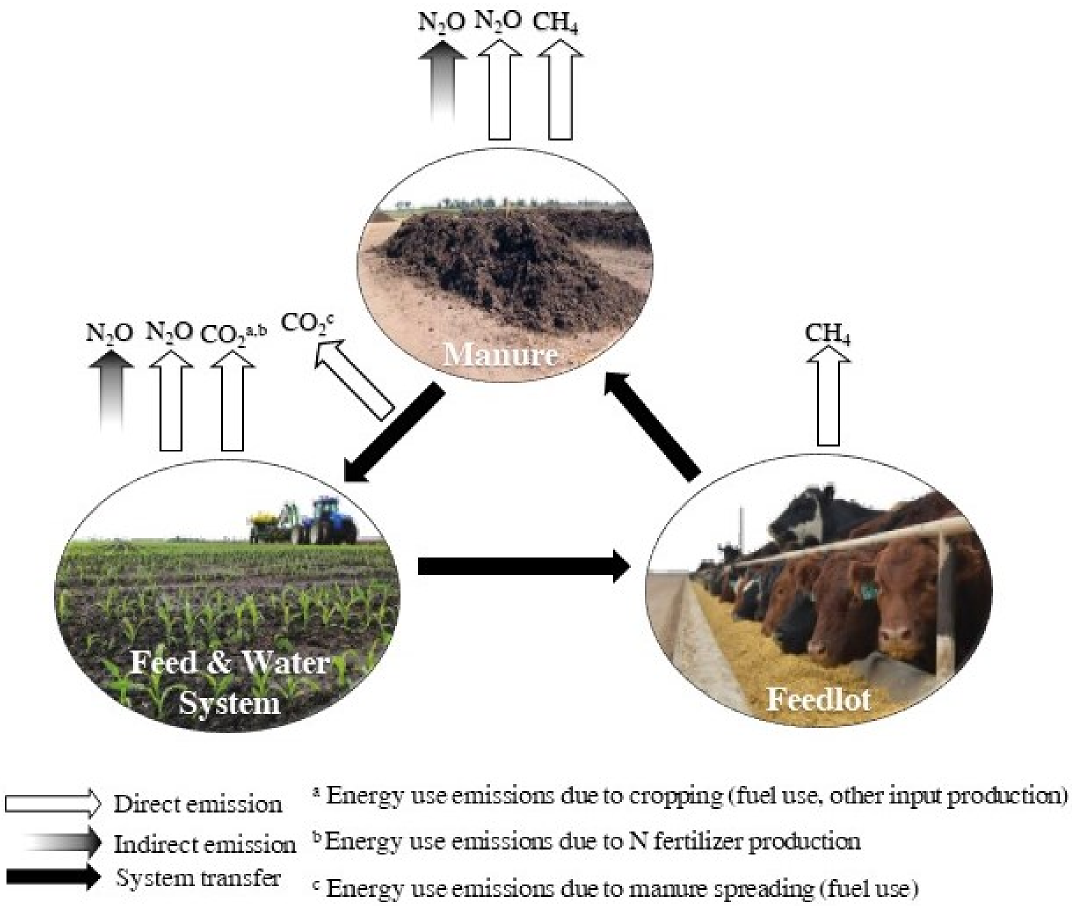

Figure 4.1 System boundaries of greenhouse gas emissions associated with feedlot cattle, adapted from Beauchemin et al. (2010).

Corn silage, barley, and DDGS were considered to originate from the farm and were produced using low-till management, while minerals and supplements were purchased. Cattle were housed in feedlot pens bedded with barley straw, and manure was managed using passive composting or stockpiling. Emissions associated with transportation (i.e., animals and feed), other capital goods, and processing and manufacturing were not included in the analysis.

Greenhouse gases associated with the production system were expressed in CO_2e to account for differences in the GWP^{100} of each gas and on an intensity basis based on the sum of all GHG sources and the unit of output (kg CO_2e, kg of boneless beef^{-1}).

4.3.3 Water use

4.3.3.1 Estimating water use: Drinking water

Animal category, average BW (Ribeiro et al. 2020), and ambient temperature over the feeding period (Environment and Climate Change Canada 2020) were used to estimate daily water consumption (Table 4.6), as described by Legesse et al. (2018a; Eq. 2):

$$\text{Total drinking water use} = WU_{coeff} \times n_d \times n_{hd} \qquad (2)$$

Where:

WU_{coeff} = the water use coefficient (litres hd^{-1} d^{-1}; National Academies of Sciences, Engineering, and Medicine, 2016);

n_d = the number of days in the feeding period;

n_{hd} = the number of animals (hd treatment^{-1} yr^{-1}).

Water intake was assumed to remain constant at temperatures ≤ 4.4 °C but was assumed to increase as temperature increased beyond this point (Table 4.6). Water used for cleaning cattle and the facilities was considered to be negligible (Beaulieu, 2007; Legesse et al., 2018a) and was excluded from the analysis.

4.3.3.2 Estimating water use: Processing plants

Water use associated with processing beef in Canadian processing plants (16.5 L kg boneless beef^{-1} Legesse et al. (2018a), considered and included differences in water use efficiency among processing plants.

Table 4.6 Estimated total daily water intake (litres hd^{-1} d^{-1}) of growing and finishing cattle[a]						
	Temperature*, °C					
Weight, kg	4.4	10.0	14.4	21.1	26.6	32.2
Growing heifers and steers						
182	15.1	16.3	18.9	22.0	25.4	36.0
273	20.1	22.0	25.0	29.5	33.7	48.1
364	23.0	25.7	29.9	34.8	40.1	56.8
Finishing cattle						
273	22.7	24.6	28.0	32.9	37.9	54.1
364	27.6	29.9	34.4	40.5	46.6	65.9
454	32.9	35.6	40.9	47.7	54.9	78.0

[a]Adapted from the National Academies of Sciences, Engineering, and Medicine (2016)

4.3.3.3 Estimating water use: Feed crop production

Water use associated with feed production was estimated based on water demand for each crop and the consumption (kg DM) of each feed ingredient within the diets, as described by Legesse et al. (2018a). Crop water demand (Table 4.7) was calculated as follows (Eq. 3):

Table 4.7 Crop water demand (L kg DM^{-1}) associated with the feed crops used in cattle diets during the four production trials

	Crop water demand, L kg DM^{-1}		
Crop	Green water use	Blue water use	Total water use
Barley grain	321	579	900
Corn grain	600	1208	1808
Corn silage	62	131	193

$$Crop\ water\ demand = PET \times K_c \tag{3}$$

Where:

PET = potential evapotranspiration;

K_c = the respective crop coefficient (ASCE 1996).

Green water that originates from precipitation was estimated by Eq. 4:

$$Green\ water = AET \times K_c \tag{4}$$

Where:

AET = actual evapotranspiration, estimated using data garnered from the National Drought Model (NDM);

K_c = the respective crop coefficient (ASCE 1996).

Blue water (water supplied via irrigation) was estimated using Eq. 5:

$$Blue\ water = PET \times K_c - Precipitation \tag{5}$$

Where:

PET = potential evapotranspiration;

K_c = the respective crop coefficient (ASCE 1996);

Precipitation = the amount of water generated from precipitation (mm).

Precipitation data from Environment and Climate Change Canada for the Lethbridge weather station (ID 3033875) were used. Where data were missing, they were garnered from two of the next closest weather stations (ID 3033897 and ID 3033890) to simulate evapotranspiration (PET and AET; Figure 4.2; Environment and Climate Change Canada 2020).

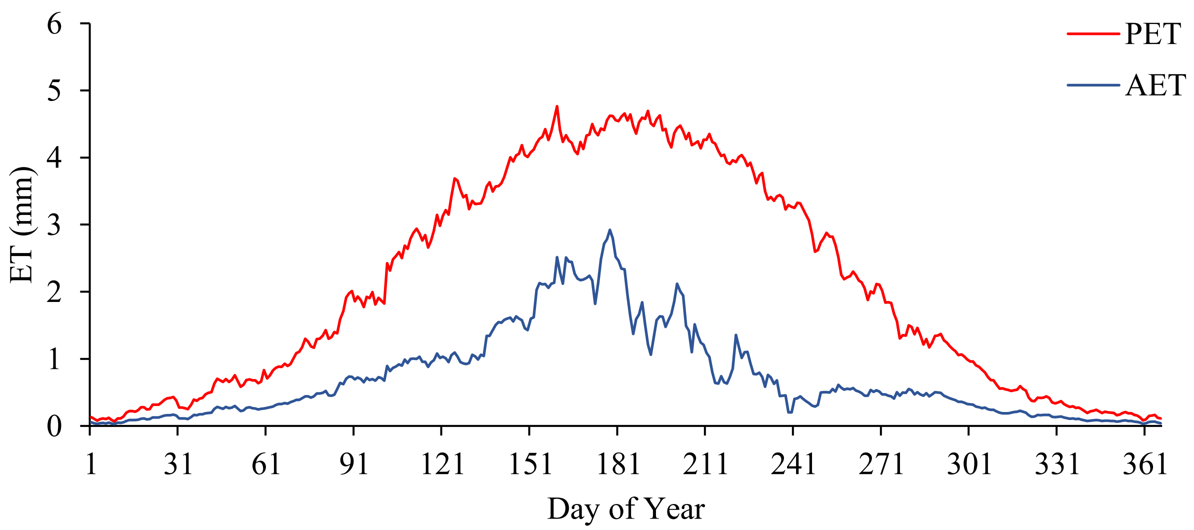

Figure 4.2 Average potential evapotranspiration (PET, mm) and actual evapotranspiration (AET, mm) over the 4-yr period (2015 to 2018) measured at the Lethbridge weather station, Lethbridge, Alberta.

Crop coefficients (K_c) were derived from literature (Allen et al., 2007, 1998; ASCE, 1996), and a K_c curve (Figure 4.3) was developed for each crop, which considered the duration and crop development stage ($K_{c\ init}$, $K_{c\ mid}$, and $K_{c\ end}$; Appendix Table 8.9). Crop-specific development stages (Allen et al., 1998) were based on growing conditions and management practices in the area, and each stage was associated with an appropriate K_c. The four stages of development were: i) initial development (L_{init}), with correspondence to $K_{c\ init}$, ii) maturity (L_{mid}), corresponding to $K_{c\ mid}$, iii) rapid development (L_{dev}), which was between L_{init} and L_{mid}, with a corresponding K_c of the rate from $K_{c\ init}$ to $K_{c\ mid}$, and iv) late-season period (L_{late}) with a corresponding K_c from the rate of $K_{c\ mid}$ to $K_{c\ end}$. An additional consideration in constructing the K_c curve was the planting date, which was sourced from Huffman et al., (2015) and signified the onset of the growing season. The length of the growing season (d) was calculated as the sum of the crop development stage (L_{init}, L_{dev}, L_{mid}, L_{late}) lengths.

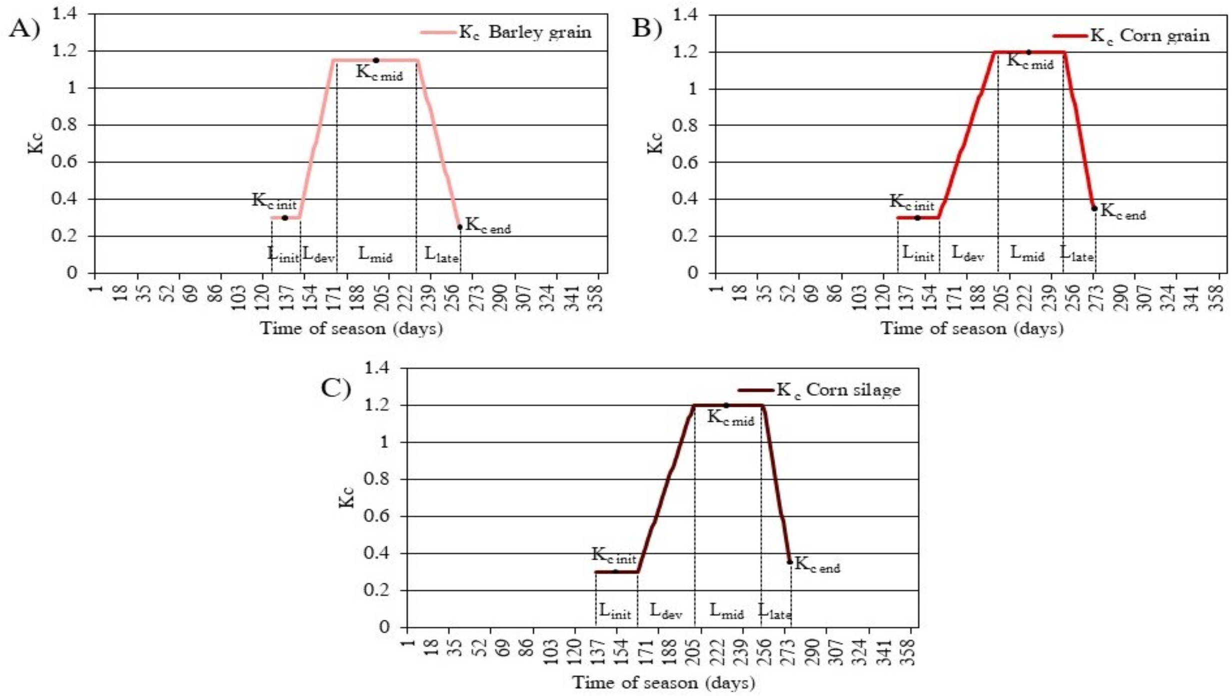

Figure 4.3 Crop coefficient (K_c) curves for A) barley grain, B) corn grain, and C) corn silage grown near Lethbridge, AB (Allen et al., 2007, 1998; ASCE, 1996; Huffman et al., 2015).

The growing season crop evapotranspiration (ET_c; Eq. 6) was the amount of water required during the growing season for the crop to develop and reach maturity.

$$ET_c = Crop\ water\ demand \times yield \quad (6)$$

Where:

Yield = the amount of crop produced (kg DM ha^{-1}).

Crop-specific ET_c for barley and corn grain and corn silage grown in Lethbridge, AB, are depicted in Figure 4.4.

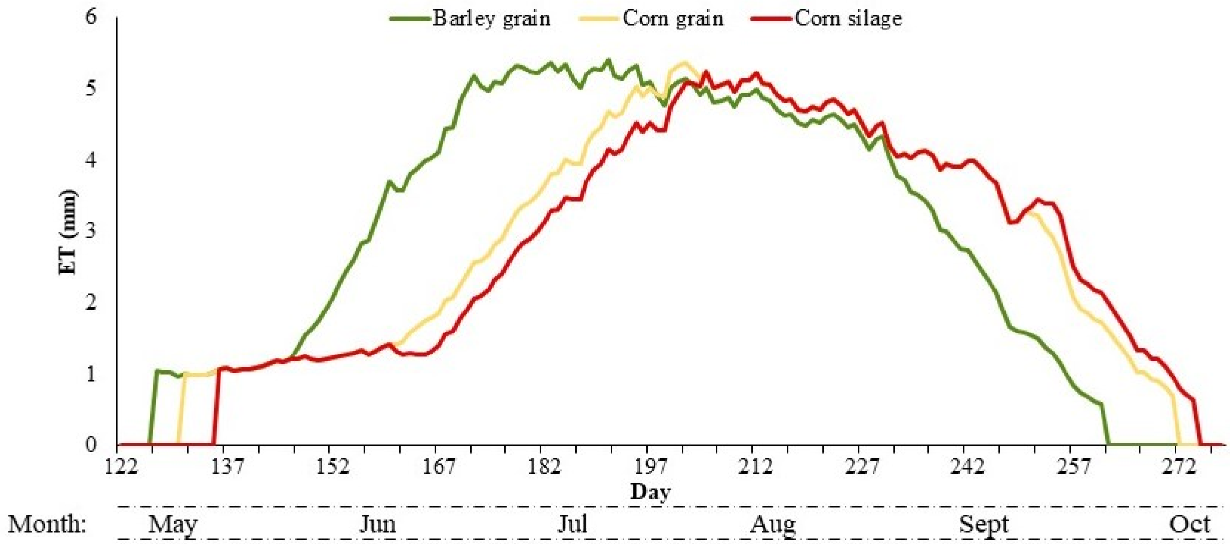

Figure 4.4 Mean growing season crop evapotranspiration (ET_c) for barley grain, corn grain, and corn silage in Lethbridge, AB.

4.3.4 Ammonia emissions

4.3.4.1 Animal N intake and total ammoniacal nitrogen excretion

Excreted total ammoniacal N (TAN) was assumed to be comprised of NH_3 and other hydrolyzable sources of N, including urea in urine, as described by Legesse et al. (2018b). The daily N excreted by cattle ($N_{excretion\ rate}$) was estimated as the difference between daily N intake and daily N retention. The estimation of daily N intake (IPCC, 2006; Eq. 7) included: i) protein intake (PI), ii) gross energy intake (GE), and iii) net energy required for maintenance ($NE_{maintenance}$).

$$N\ intake = PI / 6.25 \qquad \text{(IPCC, 2006; 7)}$$

Where:

N intake = Daily N intake, kg N hd^{-1} d^{-1};

PI = the protein intake, kg hd^{-1} d^{-1} (Eq. 8);

6.25 = the coefficient for the conversion from dietary protein to dietary N.

$$PI = (GE / 18.45) \times CP \qquad \text{(IPCC, 2006; 8)}$$

Where:

GE = gross energy intake, MJ hd^{-1} d^{-1} (Eq. 9);

18.45 = the conversion factor for gross energy kg^{-1} DM, MJ kg^{-1};

CP = the crude protein of the feed, kg kg^{-1}.

$$GE = (((NE_{maintenance} + NE_{activity}) / REM) + (NE_{gain} / REG)) / (DE / 100) \qquad \text{(IPCC, 2006; 9)}$$

Where:

$NE_{maintenance}$ = the net energy required for maintenance, MJ hd^{-1} d^{-1} (Eq. 10);

$NE_{activity}$ = the net energy required for feeding, MJ hd^{-1} d^{-1}. Trial $NE_{activity}$ was assumed to be negligible as a consequence of confinement during the feeding period (IPCC, 2006);

REM = the ratio of the net energy available in the diet for maintenance to the digestible energy consumed, MJ hd^{-1} d^{-1} (Eq. 11);

NE_{gain} = the net energy for gain, MJ hd^{-1} d^{-1} (Eq. 12);

REG = the ratio of net energy available in the diet for gain to the digestible energy consumed, MJ hd^{-1} d^{-1} (Eq. 13);

DE = the digestible energy of the feed, % TDN.

$$NE_{maintenance} = Cf_{adjusted} \times BW_{average}^{\ 0.75} \qquad \text{(IPCC, 2006; 10)}$$

Where:

$Cf_{adjusted}$ = The temperature adjusted baseline maintenance coefficient (IPCC, 2006);

$BW_{average}$ = the average body weight, kg.

$$REM = 1.123 - (4.092\times10^{-3} \times DE) + (1.126\times10^{-5} \times DE^2) - (25.4 / DE) \quad \text{(IPCC, 2006; 11)}$$

$$NE_{gain} = 22.02 \times (BW_{average} / (C_d \times BW_{final}))^{0.75} \times ADG^{1.097} \quad \text{(IPCC, 2006; 12)}$$

Where:

C_d = cattle description coefficient (IPCC, 2006);

BW_{final} = final body weight, kg;

ADG = average daily gain, kg d^{-1}.

$$REG = 1.164 - (5.160\times10^{-3} \times DE) + (1.308\times10^{-5} \times DE^2) - (37.4 / DE) \quad \text{(IPCC, 2006; 13)}$$

Daily N retention included protein retained for growth (PR_{gain}; Eq. 14)

$$PR_{gain} = ADG \times ((268 - (29.4 \times (RE / ADG)) / 1000) \quad \text{(NRC, 2000; 14)}$$

Where:

PR_{gain} = Protein retained for growth, kg hd^{-1} d^{-1};

RE = retained energy, Mcal hd^{-1} d^{-1} (NRC, 2000; Eq. 15).

$$RE = 0.0635 \times EBW^{0.75} \times EBG^{1.097} \quad (15)$$

Where:

EBW = empty body weight, kg hd^{-1} (NRC, 2000; Eq. (16));

EBG = empty body gain, kg hd^{-1} d^{-1} (NRC, 2000; Eq. (17)).

$$EBW = BW_{average} \times 0.891 \quad (16)$$

$$EBG = ADG \times 0.956 \tag{17}$$

The N excreted in urine ($TAN_{excreted}$) and the remaining N, excreted N in the feces ($FecalN_{excreted}$) were estimated with Eq. 18 and Eq. 19, respectively (Dämmgen and Hutchings 2008).

$$TAN_{excreted,} = N_{excretion\ rate} \times Fraction_{Urinary\text{-}N} \tag{18}$$

Where:

$TAN_{excreted}$ = the excreted N in urine, kg TAN hd^{-1} d^{-1};

$Fraction_{Urinary\text{-}N}$ = The fraction of excreted N in urine. As the CP of the diets was between 9% and 15%, the fraction of urinary-N was assumed to be 0.57 for all cattle groups (Chai et al. 2014).

$$FecalN_{excreted} = N_{excretion\ rate} \times (1 - Fraction_{Urinary\text{-}N}) \tag{19}$$

Where:

$FecalN_{excreted}$ = the N excreted in the fecal matter, kg fecal N hd^{-1} d^{-1}.

4.3.4.2 Quantification of NH_3 emissions

4.3.4.2.1 Source: Animal housing

Cattle were held in confinement in all experiments, thus, there were no emissions associated with grazing. Daily NH_3 emissions from confined animal housing ($NH_{3emissions,\ h}$; Legesse et al., 2018b) were estimated with Eq. 20.

$$NH_{3emissions,\ h} = TAN_{excreted} \times EF_{Feedlot\ adjusted} \times 17/14 \qquad (20)$$

Where:

$NH_{3emissions,\ h}$ = NH_3 emissions from confined animal housing, kg NH_3 hd^{-1} d^{-1}

$EF_{Feedlot\ adjusted}$ = the temperature-adjusted emission factor associated with feedlot housing systems, kg (ammoniacal nitrogen; NH_3-N) kg^{-1} (Eq. 21);

17/14 = the coefficient for the conversion of NH_3-N to NH_3.

$$EF_{Feedlot\ adjusted} = feedlot\ adjustment \times EF_{feedlot} \qquad (21)$$

Where:

feedlot adjustment = the adjusted temperature when cattle were housed in the feedlot, kg (NH_3-N) kg^{-1} (Eq. 22);

$EF_{feedlot}$ = 0.90, kg (NH_3-N) kg^{-1}.

$$Feedlot\ adjustment = 1.041^{average\ temperature,\ ℃}) \div 1.041^{17.7} \qquad (22)$$

Periodic TAN excreted (Mg TAN hd^{-1} $period^{-1}$) and periodic NH_3 emissions (Mg NH_3 hd^{-1} $period^{-1}$) from confined animals ($PTAN_{excreted,h}$, and $PNH_{3emission,h}$ respectively) were estimated by converting the kg of TAN and kg of NH_3 into Mg, by dividing by 1000, and multiplying by the days in the period.

4.3.4.2.2 Source: Manure storage

The periodic TAN mass flow from feedlot cattle to the manure storage system ($PTAN_{storage}$) were estimated by subtracting periodic NH_3 volatilization from the periodic TAN excreted during confinement (Eq. 23). Any contribution of NH_3 emissions from waste feeds or bedding materials were assumed to be negligible and were excluded from the analysis (Legesse et al. 2018b).

$$PTAN_{storage} = PTAN_{excreted,h} - (PNH3_{emission,h} \times 14/17) \quad (23)$$

Where:

$PTAN_{storage,}$ = the periodic TAN of stored manure, Mg TAN hd^{-1} $period^{-1}$;

$PTAN_{excreted,\ h}$ = the periodic TAN in manure excreted by housed beef cattle (Mg TAN hd^{-1} feeding $period^{-1}$);

$PNH3_{emissions,\ h}$ = the periodic emission rate of NH_3, temperature corrected, from housed beef cattle (Mg NH_3 hd^{-1} feeding $period^{-1}$);

14/17 = conversion of NH_3-N to NH_3.

Total ammoniacal nitrogen and a specific EF were used to estimate the NH_3 volatilization from stockpiled manure. The variation in TAN during manure storage was estimated by Eq. 24 in which the TAN during storage was combined with TAN mineralized from organic N by subtracting nitrified N from the total.

$$PTAN_{stroage2} = PTAN_{storage} \times (1 - F_{immob})(1 - F_{nitrifiy}) + (PON_{storage} \times F_{mineralize}) \quad (24)$$

Where:

$PTAN_{storage2}$ = the adjusted TAN in manure storage used to calculate NH_3 emissions hd^{-1} during manure storage, Mg TAN hd^{-1} $period^{-1}$;

F_{immob} = the TAN fraction which is immobilized to organic N during manure storage;

$F_{nitrify}$ = the TAN fraction nitrified during manure storage;

$PON_{storage}$ = the periodic organic N mass flow from the feedlot (confined feeding system) to the manure storage system (excreted fecal N) per animal, Mg organic N hd^{-1} $period^{-1}$;

$F_{mineralize}$ = the organic N fraction which is mineralized as TAN during manure storage.

It was assumed that no TAN was immobilized to organic N during manure storage (F_{immob} = 0), while the fraction of organic N mineralized as TAN was 0.28 ($F_{mineralize}$ = 0.28) and the TAN fraction nitrified from stockpiled manure was 0.14 ($F_{nitrify}$ = 0.14; Chai et al., 2014).

4.3.4.2.3 Source: Manure land application

Using the methodology described by Legesse et al., (2018b), 57% and 43% of the manure was spread on tilled and untilled land, respectively based on management practices used on Canadian beef farms. Further, as the majority of stockpiled manure from beef farms is applied in the spring or fall (Sheppard, 2015), average temperatures from April to May and September to November were used to estimate NH_3 emissions. Volatized NH_3 during land application of manure was estimated based on the available TAN and the EF associated with the application practice, land, tillage, and month of application The quantity of manure applied and remaining NH_3 after storage was used to estimate the monthly TAN transfer from manure storage to land application (Eq. 25).

$$PTAN_{land,tillage} = F_{till/untill} \times (PTAN_{storage2} - MNH_{3emissions,s} \times 14/17 \quad (25)$$

Where:

$PTAN_{land,tillage}$ = the periodic TAN applied on tilled or untilled land during a specific period, Mg NH_3 hd^{-1} $month^{-1}$;

$MNH_{3emissions,s}$ = NH_3 emission rate during the manure storage period, Mg NH_3 hd^{-1} $month^{-1}$;

$F_{till/until}$ = the fraction of manure which is applied on tilled or untilled land.

4.4 RESULTS

4.4.1 *Effect of GET removal on greenhouse gas emissions*

GHG emissions profile

On average, 44.9%, 35.7%, 13.1%, 5.8%, and 0.5% associated with enteric CH_4, direct N_2O, energy CO_2, indirect N_2O, and manure CH_4, respectively (Figure 4.5).

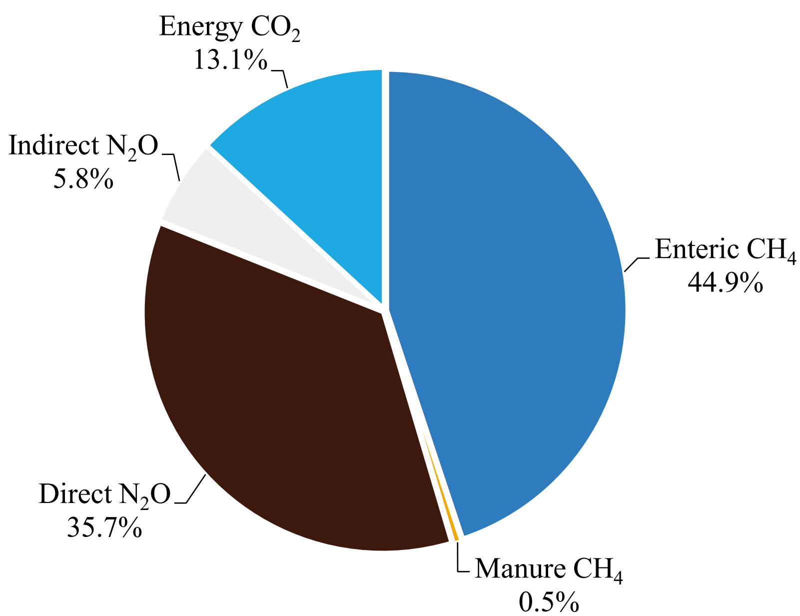

Figure 4.5 Sources of average annual greenhouse gas emissions (% of total) from all treatments during backgrounding and finishing of heifers and steers in a western Canadian feedlot.

Impact of GET's on total GHG emissions

Average total GHG emissions (n= 4 trials) were 70 to 235 kg CO_2e hd^{-1} and 388 to 488 kg CO_2e hd^{-1} lower for HCON and SCON, respectively than for GET cattle, representing a 3 to 9% and 14 to 17% reduction in GHG emissions for heifers and steers, respectively (Figure 4.6). However, when adjusted for DOF, total mean GHG emissions were 3% and 2% lower for HTBA and HMGA cattle compared to HCON_AdjTBA and HCON_AdjMGA cattle, respectively. Similarly, total emissions were 6% and 5% lower for STBA and SRAC compared to SCON_AdjTBA and SCON_AdjRAC steers, respectively.

More specifically, HTBA had lower energy CO_2 (5%), indirect N_2O (2%), direct N_2O (4%), manure CH_4 (7%), and enteric CH_4 (1%) emissions compared to HCON_AdjTBA (Figure 4.6). Heifers receiving MGA also had lower energy CO_2 (4%), indirect N_2O (1%), direct N_2O (3%) and manure CH_4 (8%) emissions than HCON_AdjMGA heifers, with no difference in enteric CH_4 emissions. Steers implanted with TBA had lower energy CO_2 (9%), indirect N_2O (7%), direct N_2O (7%), manure CH_4 (7%), and enteric CH_4 (4%) emissions as compared to SCON_AdjTBA steers. Furthermore, SRAC had 9%, 6%, 6%, 7% and 3% lower energy CO_2, indirect N_2O, direct N_2O, manure CH_4, and enteric CH_4 emissions, respectively, than SCON_AdjRAC.

Source-specific GHG emissions

Average enteric CH_4 emissions from all treatments during the backgrounding phase were higher than the finishing phase (84% vs. 75%), while average direct and indirect N_2O emissions were higher (17% vs. 11% and 8% vs. 5%, respectively) during the finishing phase (Figure 4.7). Average direct N_2O emissions were responsible for the majority of the crop emissions. Average indirect N_2O emissions represented 8%, 5%, 4%, and 4% of the total emissions from land applied manure, corn grain, barley grain, and corn silage, respectively. Energy CO_2 represented more than a third of all crop emissions for corn grain (45%), barley grain (41%), and a quarter of the crop emissions for corn silage, with a relatively minor amount associated with land applied manure (2%; Figure 4.7).

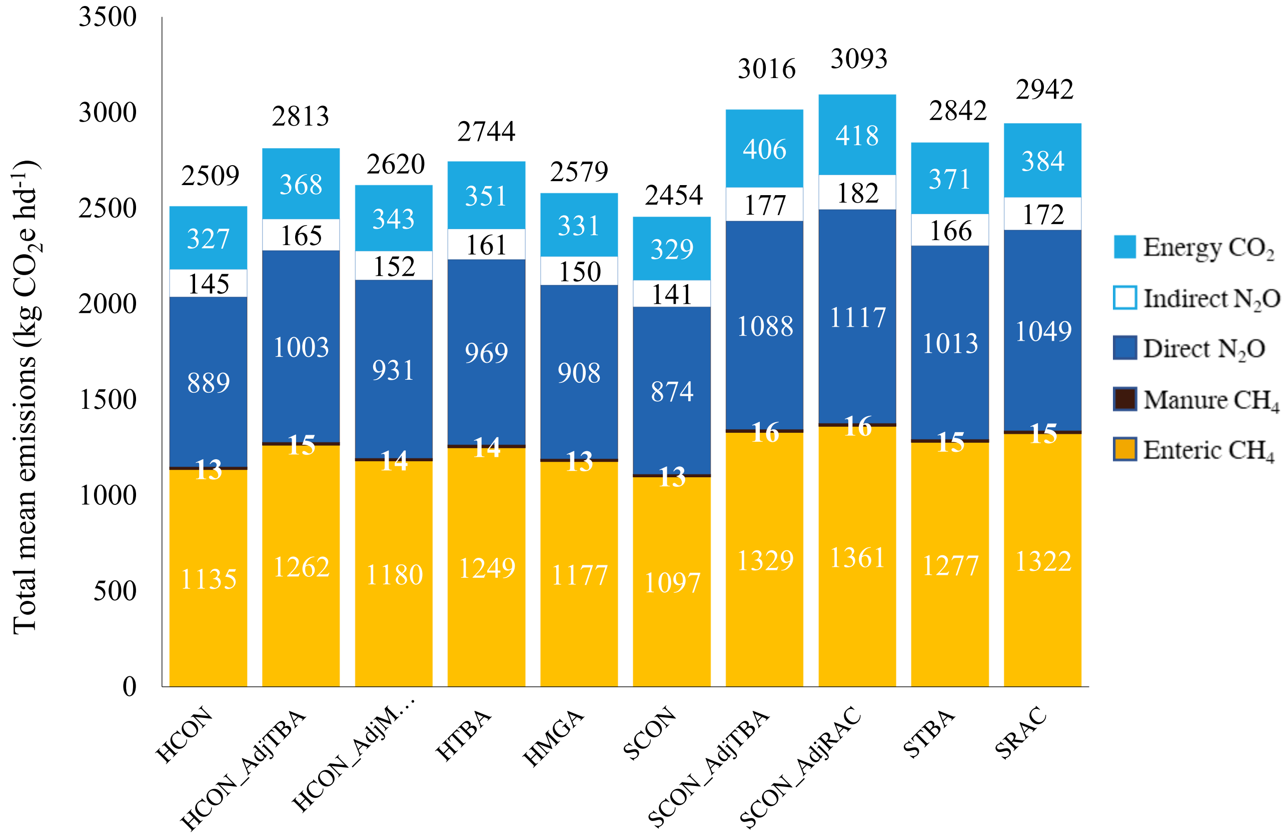

Figure 4.6 Total mean greenhouse gas emissions and emission profile (kg CO_2e hd^{-1}) for heifers and steers backgrounded and finished with or without the use of growth-enhancing technologies. Treatments were **HCON** = control heifers; **HTBA** = implanted heifers; **HMGA** = heifers receiving MGA + implants; **HCON_AdjTBA** = control heifers adjusted to reach the same finished body weight (BW) as HTBA; **HCON_AdjMGA** = control heifers adjusted to reach the same finished BW as HMGA; **SCON** = control steers; **STBA** = implanted steers; **SRAC** = steers received a β-AA + implants; **SCON_AdjTBA** = control steers adjusted to reach the same finished BW as STBA; **SCON_AdjRAC** = control steers adjusted to reach the same finished BW as SRAC.

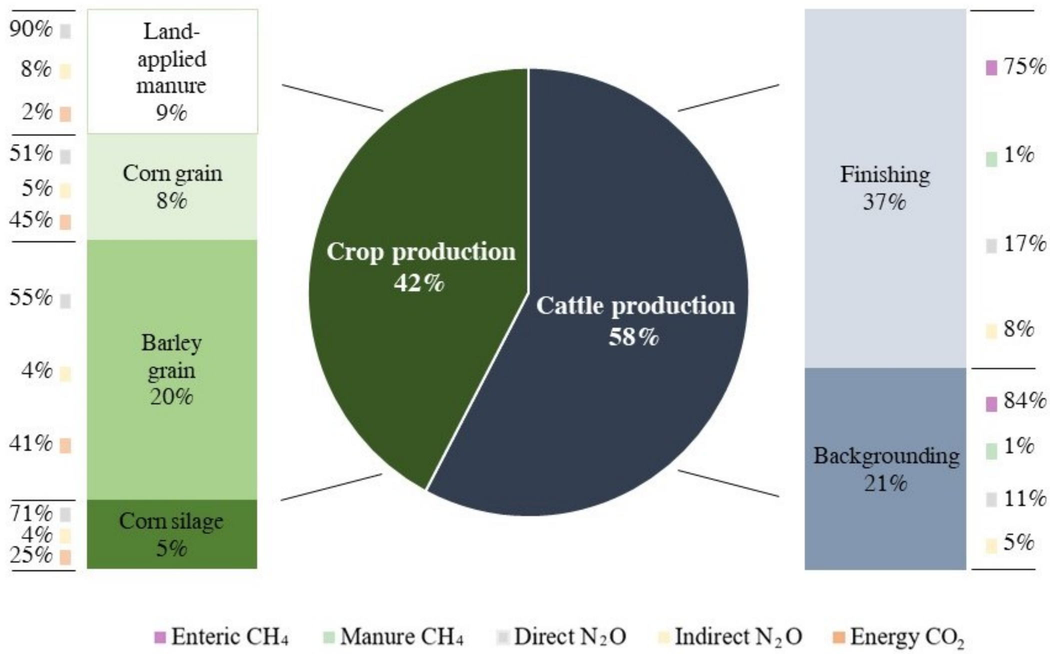

Figure 4.7 Sources of average annual crop and cattle-associated greenhouse gas emissions (% of total) associated with feedlot heifers and steers.

Greenhouse gas emissions from crops (barley grain, corn silage, and corn grain) and land applied manure were lower for GET cattle than control cattle (Figure 4.8). Direct and indirect N_2O and energy CO_2 emissions associated with crops and land applied manure were 4% and 5% lower for HTBA than HCON_AdjTBA, respectively. A similar trend was observed for HMGA, as N_2O (direct and indirect) and energy CO_2 emissions from crops and land applied manure were 3% and 4% lower respectively, than HCON_AdjMGA. Similarly, the direct and indirect N_2O, as well as energy CO_2 emissions associated with crops and land applied manure of STBA and SRAC were 8%, 7%, and 9% lower than the respective control-adjusted cattle.

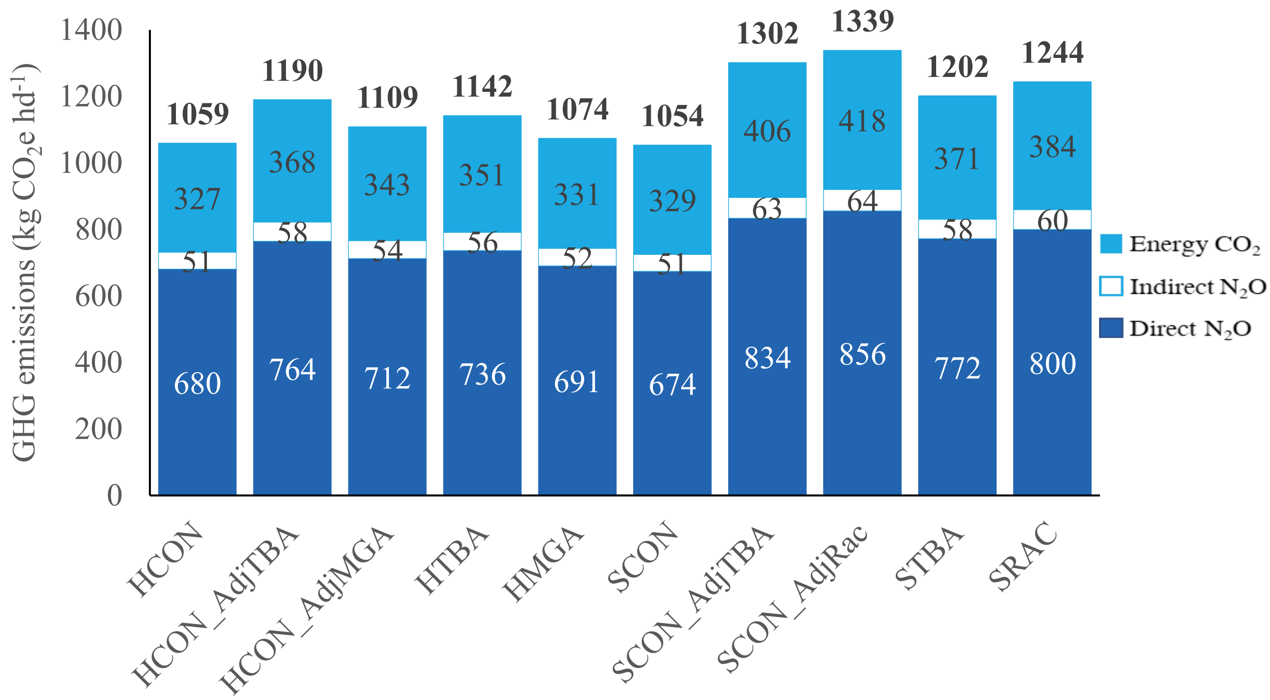

Figure 4.8 Crop and land applied manure greenhouse gas emission profile of heifers and steers finished with or without growth-enhancing technologies. Treatments were **HCON** = control heifers; **HTBA** = implanted heifers; **HMGA** = heifers receiving MGA + implants; **HCON_AdjTBA** = control heifers adjusted to reach the same finished body weight (BW) as HTBA; **HCON_AdjMGA** = control heifers adjusted to reach the same finished BW as HMGA; **SCON** = control steers; **STBA** = implanted steers; **SRAC** = steers received a β-AA + implants; **SCON_AdjTBA** = control steers adjusted to reach the same finished BW as STBA; **SCON_AdjRAC** = control steers adjusted to reach the same finished BW as SRAC.

Total emissions on an intensity basis:

Emissions expressed on a kilogram carcass weight (Table 4.8), or a kilogram boneless beef basis (Figure 4.9) were greater for HCON_AdjTBA and HCON_AdjMGA than HTBA (7.3 vs 7.0 kg CO_2e kg carcass weight^{-1}; 10.3 vs 9.9 kg CO_2e kg boneless beef^{-1}) and HMGA (7.0 vs 6.8 kg CO_2e kg carcass weight$^{-1;}$ 9.9 vs 9.6 kg CO_2e kg boneless beef^{-1}). Similarly, weight-adjusted control steers had greater emissions than STBA (7.1 vs 6.7 kg CO_2e kg carcass weight^{-1}; 10.0 vs

9.4 kg CO_2e kg boneless beef^{-1}) and SRAC (7.2 vs 6.7 kg CO_2e kg carcass weight$^{-1;}$ 10.1 vs 9.4 kg CO_2e kg boneless beef^{-1}). As a result, intensity-based emissions were 3 to 4% and 6 to 7% greater for control heifers and steers, respectively, as compared to GET heifers and steers.

Table 4.8 Intensity of greenhouse gases (kg CO_2e kg carcass weight^{-1}) associated with backgrounding and finishing heifers and steers with or without the use of growth-enhancing technologies

Treatment[a]	Emissions intensity
	kg CO_2e kg carcass weight^{-1}
HCON	6.9
HCON_AdjTBA	7.3
HCON_AdjMGA	7.0
HTBA	7.0
HMGA	6.8
SCON	6.5
SCON_AdjTBA	7.1
SCON_AdjRAC	7.2
STBA	6.7
SRAC	6.7

[a]Treatments were **HCON** = control heifers; **HTBA** = implanted heifers; **HMGA** = heifers receiving MGA + implants; **HCON_AdjTBA** = control heifers adjusted to reach the same finished body weight (BW) as HTBA; **HCON_AdjMGA** = control heifers adjusted to reach the same finished BW as HMGA; **SCON** = control steers; **STBA** = implanted steers; **SRAC** = steers received a β-AA + implants; **SCON_AdjTBA** = control steers adjusted to reach the same finished BW as STBA; **SCON_AdjRAC** = control steers adjusted to reach the same finished BW as SRAC.

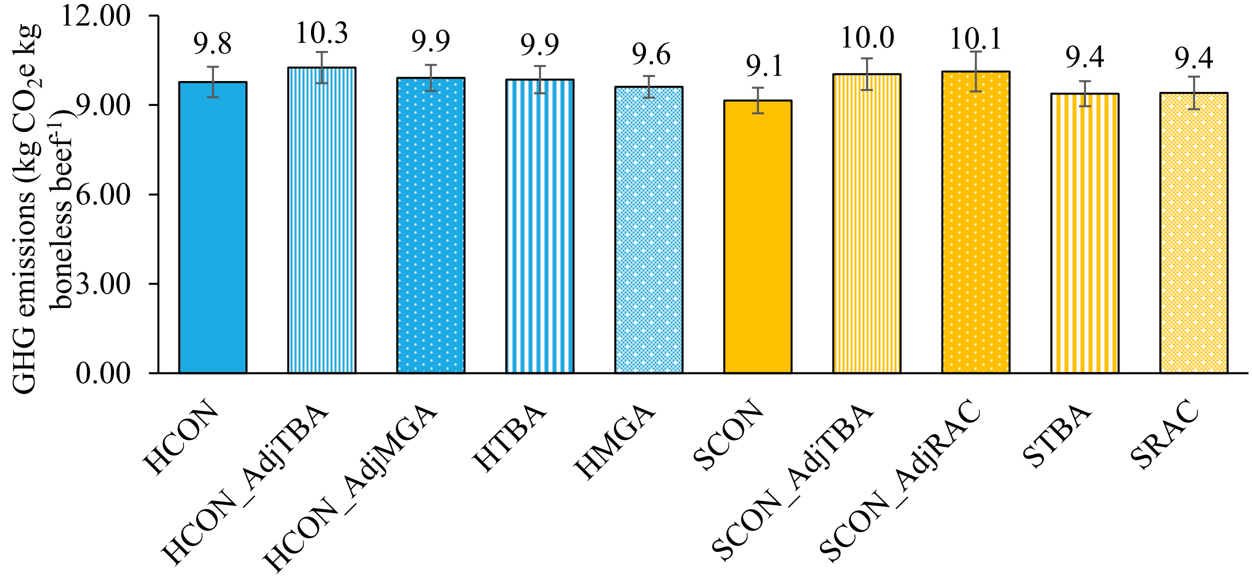

Figure 4.9 Total mean greenhouse gas emissions, expressed on an intensity basis (kg CO_2e kg boneless beef^{-1}) from heifers and steers backgrounded and finished with or without the use of growth-enhancing technologies (n=4 trials). Treatments were **HCON** = control heifers; **HTBA** = implanted heifers; **HMGA** = heifers receiving MGA + implants; **HCON_AdjTBA** = control heifers adjusted to reach the same finished body weight (BW) as HTBA; **HCON_AdjMGA** = control heifers adjusted to reach the same finished BW as HMGA; **SCON** = control steers; **STBA** = implanted steers; **SRAC** = steers received a β-AA + implants; **SCON_AdjTBA** = control steers adjusted to reach the same finished BW as STBA; **SCON_AdjRAC** = control steers adjusted to reach the same finished BW as SRAC.

4.4.2 Effect of GET removal on land requirements

Average land requirements ranged from 0.36 to 0.41 and 0.36 to 0.47 ha hd^{-1} for heifers and steers, respectively (Table 4.9). Weight-adjusted control heifers required 5% and 3% more land than HTBA and HMGA heifers, respectively. This was a result of HTBA heifers requiring 6% and 3% less land for barley and corn grain production, respectively, than HCON_AdjTBA heifers (Figure 4.10). Heifers receiving MGA required 2%, 4%, and 3% less land for the production of corn silage, barley, and corn grain, respectively, than HCON_AdjMGA heifers. Control steers required 12% and 16% less land than STBA and SRAC, respectively. However, when adjusted to comparable finish weights, SCON_AdjTBA and SCON_AdjRAC required 10%

and 9 % more land than STBA and SRAC, respectively. The TBA-implanted steers required 2%, 11%, and 0.5% less land for corn silage, barley, and corn grain production, respectively, than SCON_AdjTBA steers. Similarly, SRAC steers required 1%, 11%, and 5% less land for the production of corn silage, barley, and corn grain, respectively, than SCON_AdjRAC steers (Figure 4.10).

Table 4.9 Total mean land use (ha hd^{-1}) required to produce feeds included in backgrounding and finishing diets of heifers and steers raised with or without the use of growth-enhancing technologies

	Total land use
Treatment[a]	ha hd^{-1}
HCON	0.36
HCON_AdjTBA	0.41
HCON_AdjMGA	0.38
HTBA	0.39
HMGA	0.37
SCON	0.36
SCON_AdjTBA	0.45
SCON_AdjRAC	0.47
STBA	0.41
SRAC	0.43

[a]Treatments were **HCON** = control heifers; **HTBA** = implanted heifers; **HMGA** = heifers receiving MGA + implants; **HCON_AdjTBA** = control heifers adjusted to reach the same finished body weight (BW) as HTBA; **HCON_AdjMGA** = control heifers adjusted to reach the same finished BW as HMGA; **SCON** = control steers; **STBA** = implanted steers; **SRAC** = steers received a β-AA + implants; **SCON_AdjTBA** = control steers adjusted to reach the same finished BW as STBA; **SCON_AdjRAC** = control steers adjusted to reach the same finished BW as SRAC.

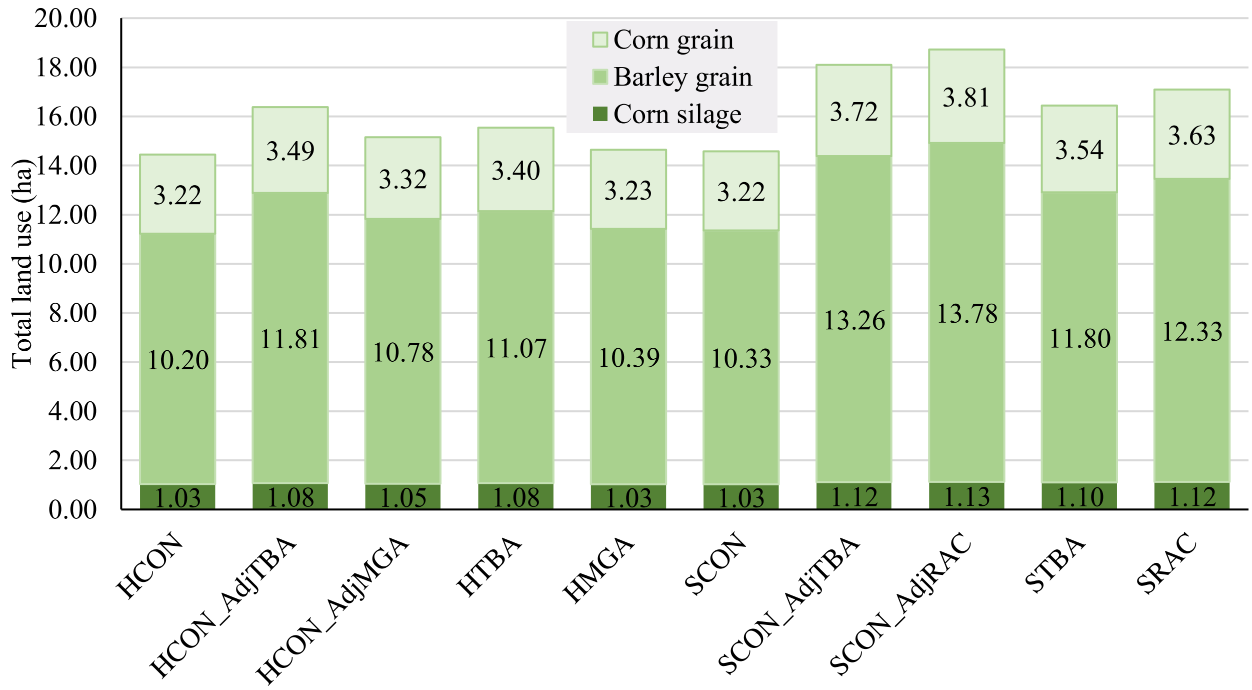

Figure 4.10 Land (ha treatment^{-1}) required to produce each feedstuff included in backgrounding and finishing diets of heifers and steers raised with or without the use of growth-enhancing technologies (n=40-hd). Treatments were **HCON** = control heifers; **HTBA** = implanted heifers; **HMGA** = heifers receiving MGA + implants; **HCON_AdjTBA** = control heifers adjusted to reach the same finished body weight (BW) as HTBA; **HCON_AdjMGA** = control heifers adjusted to reach the same finished BW as HMGA; **SCON** = control steers; **STBA** = implanted steers; **SRAC** = steers received a β-AA + implants; **SCON_AdjTBA** = control steers adjusted to reach the same finished BW as STBA; **SCON_AdjRAC** = control steers adjusted to reach the same finished BW as SRAC.

Land use requirement was greater for weight-adjusted control cattle than the respective GET cattle, while the unadjusted control groups were comparable to the GET cattle (Table 4.10). The weight-adjusted control heifers required an additional 0.005 to 0.007 ha kg carcass weight^{-1} than the associated GET heifers, and the weight-adjusted control steers required an additional 0.011 to 0.012 ha kg carcass weight^{-1} than GET steers. Similarly, when expressed on a boneless beef basis, HCON had higher land use requirements than HTBA and HMGA, while SCON were comparable to STBA and SRAC. Further, weight-adjusted control cattle had higher land requirements than the associated GET cattle. The implanted heifers had 0.007 ha kg boneless beef-

^{1} less land required than HCON_AdjTBA and HMGA required 0.10 ha kg boneless beef^{-1} less land than HCON_HMGA. Implanted steers required 0.015 fewer ha kg boneless beef^{-1} SCON_AdjTBA, and SRAC required 0.016 fewer ha kg boneless beef^{-1} than SCON_AdjRAC. As a result, on an intensity basis, HTBA, HMGA, STBA, and SRAC required 6.7%, 4.9%, 9.9%, and 10.5% less land than their weight-adjusted counterparts, respectively.

Table 4.10 Intensity of land use (ha kg carcass weight^{-1} and ha kg boneless beef^{-1}) associated with heifers and steers backgrounded and finished with or without the use of growth-enhancing technologies

	Land use intensity	
Treatment[a]	ha kg carcass weight^{-1}	ha kg boneless beef^{-1}
HCON	0.100	0.141
HCON_AdjTBA	0.106	0.149
HCON_AdjMGA	0.102	0.143
HTBA	0.099	0.139
HMGA	0.097	0.136
SCON	0.096	0.136
SCON_AdjTBA	0.107	0.151
SCON_AdjRAC	0.109	0.153
STBA	0.096	0.136
SRAC	0.097	0.137

[a]Treatments were **HCON** = control heifers; **HTBA** = implanted heifers; **HMGA** = heifers receiving MGA + implants; **HCON_AdjTBA** = control heifers adjusted to reach the same finished body weight (BW) as HTBA; **HCON_AdjMGA** = control heifers adjusted to reach the same finished BW as HMGA; **SCON** = control steers; **STBA** = implanted steers; **SRAC** = steers received a β-AA + implants; **SCON_AdjTBA** = control steers adjusted to reach the same finished BW as STBA; **SCON_AdjRAC** = control steers adjusted to reach the same finished BW as SRAC.

4.4.3 Effect of GET removal on water requirements

Sources of water use

Feed production was the primary source of water use (99.3%), while drinking and processing plant water accounted for 0.4% and 0.3%, respectively (Figure 4.11). Barley grain required nearly three-quarters of the total crop production water use (74%), while corn silage and corn grain accounted for 6% and 20%, respectively. Blue water accounted for approximately 66% of the water use associated with crop production, with green water accounting for 33%.

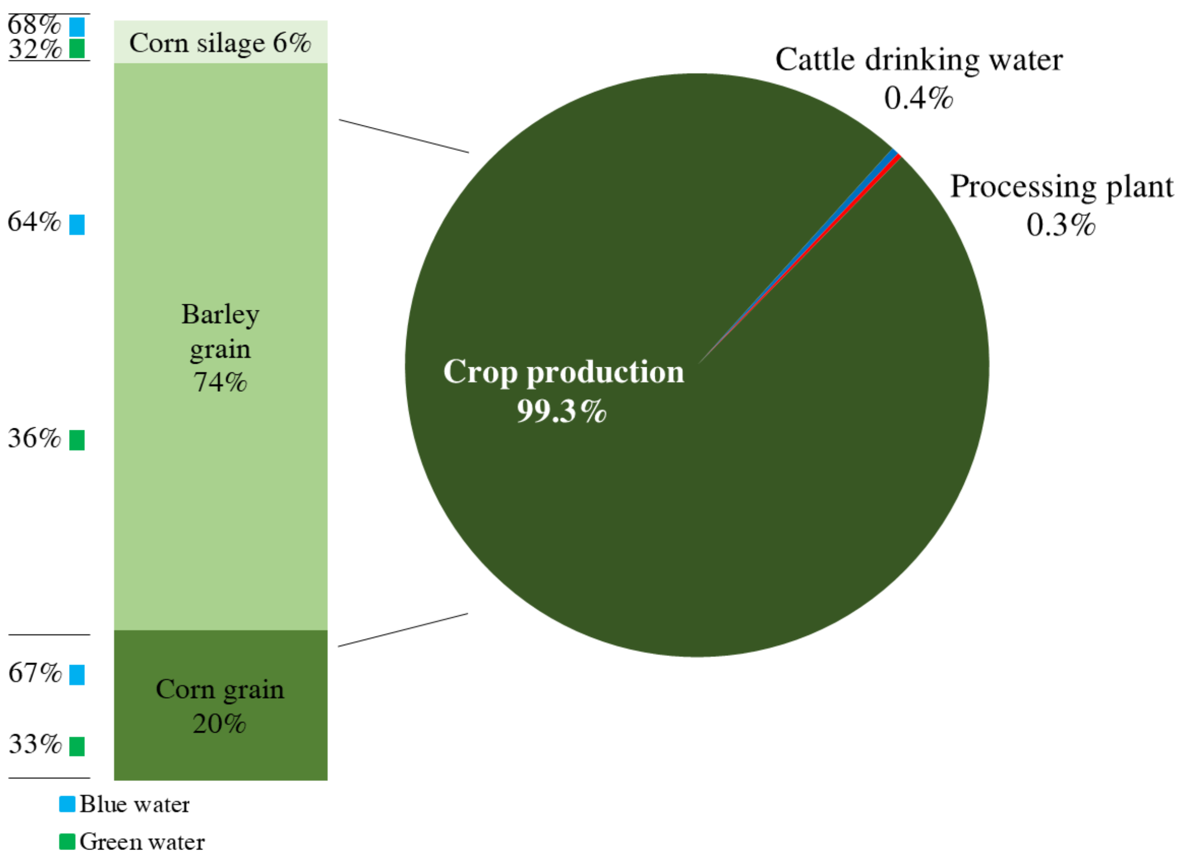

Figure 4.11 Summary of the sources of annual average water use as a portion of total water use for crop production, drinking and processing of backgrounded and finished heifers and steers (n=4 trials).

Drinking water (m^3 hd^{-1}) was 14%, 5%, 27%, and 26% lower for HTBA, HMGA, STBA, and SRAC, compared to their respective weight-adjusted control groups (Table 4.11). Water use associated with crop production was 6%, 3%, 10%, and 10% lower for HTBA, HMGA, STBA, and SRAC compared to their respective weight-adjusted control groups. Water requirement for corn grain was 3% and 5% less for GET heifers and steers, respectively as compared to weight-adjusted controls, while water required for corn silage was 1%, 2%, 1%, and 1% less for HTBA, HMGA, STBA, and SCON compared to respective weight-adjusted controls. Water requirements for barley grain associated with HTBA, HMGA, STBA, and SRAC were 7%, 4%, 12%, and 12% lower than HCON_AdjTBA, HCON_AdjMGA, SCON_AdjTBA, and SCON_AdjRAC, respectively. Further, there was no change in processing plant water use for HMGA vs. HCON_AdjMGA and STBA vs. SCON_AdjTBA, while HTBA and SRAC required 2% and 4% more water for processing, respectively compared to HCON_AdjTBA and SCON_AdjRAC.

Total water use

Implanted heifers and HMGA heifers required 8% and 1% more total water (m^3 hd^{-1}) than HCON and STBA and SRAC required 13% and 17% more water than SCON (Table 4.11). When compared to weight-adjusted control cattle, there was a 6% and 3% reduction in total mean water use (m^3 water hd^{-1}) associated with HTBA and HMGA heifers, respectively and 9% less water was required for STBA and SRAC than the SCON_AdjTBA and SCON_AdjRAC.

Table 4.11 Water use associated with heifers and steers backgrounded and finished with or without the use of growth-enhancing technologies

Water use, m^3 hd^{-1}	Treatment[a]									
	HCON	HCON_ AdjTBA	HCON_ AdjMGA	HTBA	HMGA	SCON	SCON_ AdjTBA	SCON_ AdjRAC	STBA	SRAC
Drinking water	6.4	7.2	6.7	6.3	6.4	6.4	8.0	8.2	6.3	6.5
Crop production										
Green water										
Corn grain	105.3	114.3	108.5	111.2	105.6	105.4	121.9	124.9	115.8	119.0
Corn silage	30.1	31.5	30.6	31.3	29.9	29.9	32.5	33.0	32.2	32.7
Barley grain	408.5	472.9	431.9	443.3	416.1	413.9	531.2	552.0	472.8	494.0
Total green water	543.9	618.7	571.0	585.9	551.7	549.2	685.6	709.8	620.8	645.7
Blue water										
Corn grain	212.0	230.2	218.6	224.0	212.7	212.3	245.5	251.5	233.3	239.6
Corn silage	63.3	66.3	64.4	65.9	63.0	63.0	68.4	69.4	67.7	68.8
Barley grain	737.6	854.0	779.8	800.5	751.4	747.3	959.2	996.7	853.7	892.0
Total blue water	1,012.9	1,150.4	1,062.8	1,090.4	1,027.1	1,022.6	1,273.0	1,317.6	1,154.7	1,200.4
Beef processing	4.2	4.5	4.4	4.6	4.4	4.4	5.0	5.0	5.0	5.2
Total water use	1,567.4	1,780.9	1,644.8	1,687.2	1,589.7	1,582.7	1,971.6	2,040.7	1,786.9	1,857.7

[a]Treatments were **HCON** = control heifers; **HTBA** = implanted heifers; **HMGA** = heifers receiving MGA + implants; **HCON_AdjTBA** = control heifers adjusted to reach the same finished body weight (BW) as HTBA; **HCON_AdjMGA** = control heifers adjusted to reach the same finished BW as HMGA; **SCON** = control steers; **STBA** = implanted steers; **SRAC** = steers received a β-AA + implants; **SCON_AdjTBA** = control steers adjusted to reach the same finished BW as STBA; **SCON_AdjRAC** = control steers adjusted to reach the same finished BW as SRAC.

Water use intensity

There was no difference in water requirements between HCON and HTBA (Table 4.12), however, HTBA required 7% less water than HCON_AdjTBA, on a carcass weight intensity basis (m^3 H_2O kg carcass weight^{-1}). Further, HMGA required 5% less water than HCON and 5% less water than HCON_AdjMGA, on a carcass weight intensity basis (m^3 H_2O kg carcass weight^{-1}). Expressed on a boneless beef intensity basis (m^3 H_2O kg boneless beef^{-1}), HTBA and HMGA required 6% and 9% less water than HCON_AdjTBA and HCON_AdjMGA, respectively.

Similarly, on a carcass weight intensity basis (m^3 H_2O kg carcass weight^{-1}), STBA and SRAC required the same amount of water as SCON but required 11% less water than the weight-adjusted control steers. On a boneless beef intensity basis (m^3 H_2O kg boneless beef^{-1}), STBA and SRAC steers required the same amount of water as SCON, however, TBA and RAC-treated steers required 11 to 12% less water than the respective weight-adjusted control steers.

Table 4.12 Intensity of water use (m^3 H_2O kg carcass weight^{-1} and m^3 H_2O kg boneless beef^{-1}) associated with heifers and steers backgrounded and finished with or without the use of growth-enhancing technologies

	Drinking Water	Processing Plant	Feed	Total	Total
Treatment[a]	m^3 H_2O kg boneless beef^{-1}				m^3 H_2O kg carcass weight^{-1}
HCON	0.0248	0.0165	6.06	6.1	4.3
HCON_AdjTBA	0.0267	0.0165	6.45	6.5	4.6
HCON_AdjMGA	0.0253	0.0165	6.18	6.2	4.4
HTBA	0.0227	0.0165	6.02	6.1	4.3
HMGA	0.0237	0.0165	5.88	5.9	4.2
SCON	0.0239	0.0165	5.86	5.9	4.2
SCON_AdjTBA	0.0267	0.0165	6.52	6.6	4.7
SCON_AdgRAC	0.0268	0.0165	6.63	6.7	4.7
STBA	0.0209	0.0165	5.86	5.9	4.2
SRAC	0.0207	0.0165	5.90	5.9	4.2

[a]Treatments were **HCON** = control heifers; **HTBA** = implanted heifers; **HMGA** = heifers receiving MGA + implants; **HCON_AdjTBA** = control heifers adjusted to reach the same finished body weight (BW) as HTBA; **HCON_AdjMGA** = control heifers adjusted to reach the same finished BW as HMGA; **SCON** = control steers; **STBA** = implanted steers; **SRAC** = steers received a β-AA + implants; **SCON_AdjTBA** = control steers adjusted to reach the same finished BW as STBA; **SCON_AdjRAC** = control steers adjusted to reach the same finished BW as SRAC.

4.4.4 Effect of GET removal on ammonia emissions

Nitrogen excretion

Total average annual N excretion (n=4 trials) was estimated as 2.58 Mg N, 2.89 Mg N, 2.69 Mg N, 2.82 Mg N, and 2.65 Mg N for HCON, HCON_AdjTBA, HCON_AdjMGA, HTBA, and HMGA, respectively. Control steers, weight-adjusted control steers, and GET steers excreted, 2.44 Mg N, 2.99 to 3.05 Mg N, and 2.81 to 2.88 Mg N, respectively (Figure 4.13).

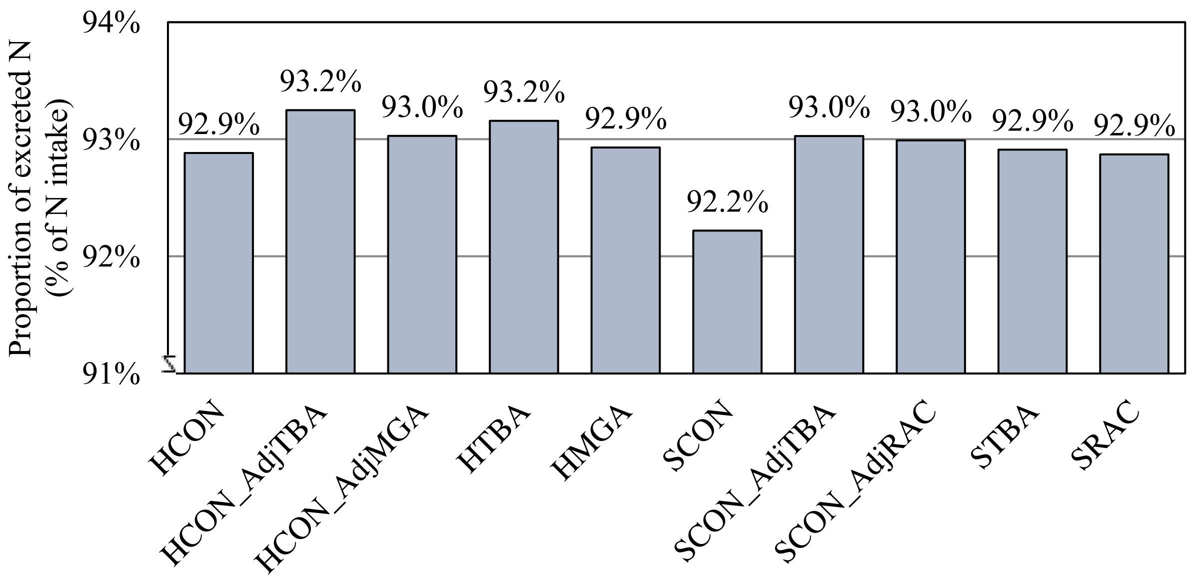

Figure 4.12 Total mean N excreted as a percentage of N intake for feedlot cattle raised with or with the use of growth-enhancing technologies (n=4 trials). Treatments were **HCON** = control heifers; **HTBA** = implanted heifers; **HMGA** = heifers receiving MGA + implants; **HCON_AdjTBA** = control heifers adjusted to reach the same finished body weight (BW) as HTBA; **HCON_AdjMGA** = control heifers adjusted to reach the same finished BW as HMGA; **SCON** = control steers; **STBA** = implanted steers; **SRAC** = steers received a β-AA + implants; **SCON_AdjTBA** = control steers adjusted to reach the same finished BW as STBA; **SCON_AdjRAC** = control steers adjusted to reach the same finished BW as SRAC.

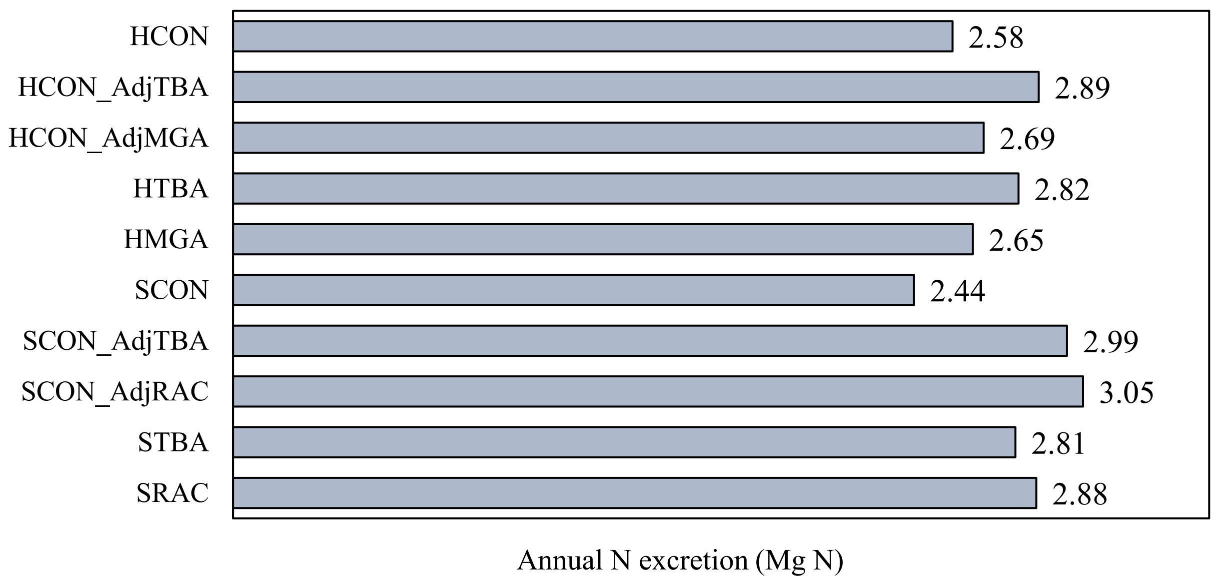

Figure 4.13 Total annual N excreted by feedlot cattle raised without or without the use of growth-enhancing technologies (n=4 trials). Treatments were **HCON** = control heifers; **HTBA** = implanted heifers; **HMGA** = heifers receiving MGA + implants; **HCON_AdjTBA** = control heifers adjusted to reach the same finished body weight (BW) as HTBA; **HCON_AdjMGA** = control heifers adjusted to reach the same finished BW as HMGA; **SCON** = control steers; **STBA** = implanted steers; **SRAC** = steers received a β-AA + implants; **SCON_AdjTBA** = control steers adjusted to reach the same finished BW as STBA; **SCON_AdjRAC** = control steers adjusted to reach the same finished BW as SRAC.

Ammonia emissions

On average, 46% of the N intake for all treatments was lost as NH_3-N; with greater loss during the finishing phase (49%) than during the backgrounding phase (40%). Sources of ammonia emissions were relatively consistent among all treatments with an estimated 66% ± 1%, 17% ±0%, and 17% ± 0% of emissions associated with animal housing, manure storage, and land spreading, respectively (Figure 4.14).

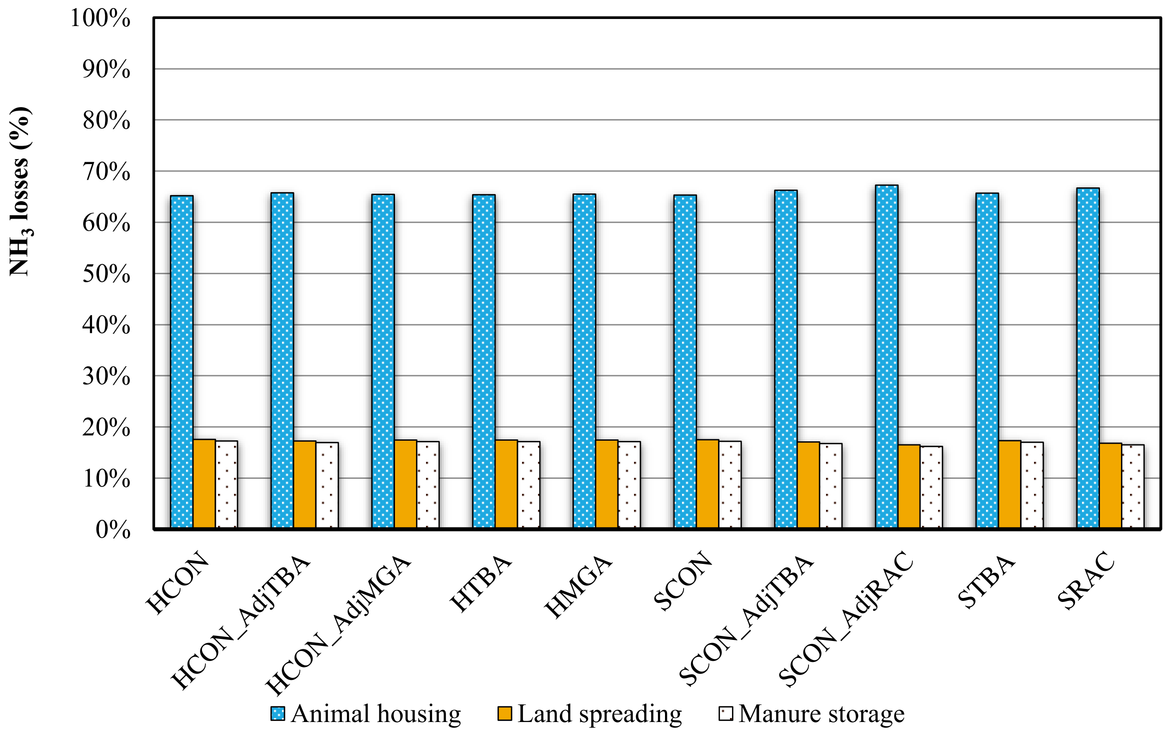

Figure 4.14 Breakdown of sources of NH_3 emissions (%) by source and treatment from feedlot cattle backgrounded and finished with or without the use of growth-enhancing technologies (n= 4 trials). Treatments were **HCON** = control heifers; **HTBA** = implanted heifers; **HMGA** = heifers receiving MGA + implants; **HCON_AdjTBA** = control heifers adjusted to reach the same finished body weight (BW) as HTBA; **HCON_AdjMGA** = control heifers adjusted to reach the same finished BW as HMGA; **SCON** = control steers; **STBA** = implanted steers; **SRAC** = steers received a β-AA + implants; **SCON_AdjTBA** = control steers adjusted to reach the same finished BW as STBA; **SCON_AdjRAC** = control steers adjusted to reach the same finished BW as SRAC.

Total average NH_3 emissions (n= 4 trials) were lower for HCON (3.90×10^{-2} Mg NH_3 hd^{-1}) than the GET-treated heifers (4.02×10^{-2} to 4.26×10^{-2} Mg NH_3 hd^{-1}; Figure 4.15). As a result, HCON had 8 to 3% lower NH_3 emissions than HTBA and HMGA, respectively. However, HTBA and HMGA had 3% and 1% lower NH_3 emissions than HCON_AdjTBA and HCON_AdjMGA, respectively. Control steers (3.69×10^{-2} Mg NH_3 hd^{-1}) had lower total annual NH_3 emissions (n= 4 trials) than GET steers (4.26×10^{-2} to 4.41×10^{-2} Mg NH_3 hd^{-1}). However, GET-treated steers had 7% and 6% lower emissions than the respective weight-adjusted control steers.

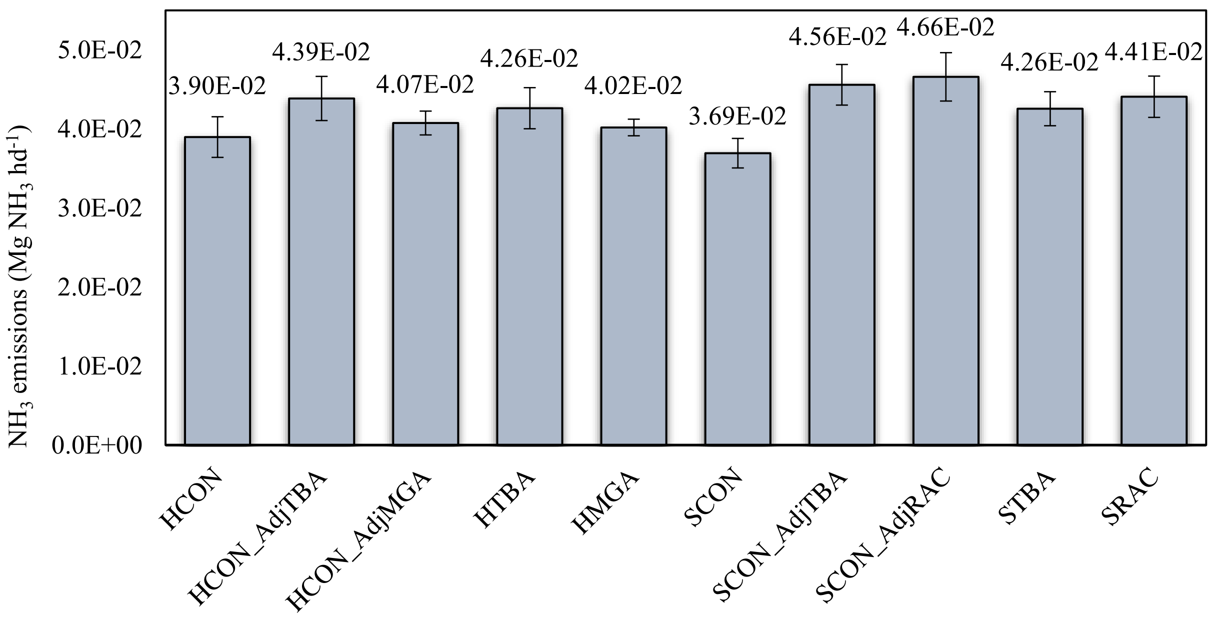

Figure 4.15 Total average NH_3 emissions (Mg hd^{-1}) associated with feedlot cattle backgrounded and finished with or without the use of growth-enhancing technologies (n=4 trials). Treatments were **HCON** = control heifers; **HTBA** = implanted heifers; **HMGA** = heifers receiving MGA + implants; **HCON_AdjTBA** = control heifers adjusted to reach the same finished body weight (BW) as HTBA; **HCON_AdjMGA** = control heifers adjusted to reach the same finished BW as HMGA; **SCON** = control steers; **STBA** = implanted steers; **SRAC** = steers received a β-AA + implants; **SCON_AdjTBA** = control steers adjusted to reach the same finished BW as STBA; **SCON_AdjRAC** = control steers adjusted to reach the same finished BW as SRAC.

Ammonia emissions: intensity basis

On an intensity basis, estimated NH_3 emissions (Figure 4.16) were 4% and 3% lower for HTBA and HMGA, respectively, compared to the respective weight-adjusted control heifers. Ammonia emissions (kg NH_3 kg boneless $beef^{-1}$) were also 8% and 7% lower for STBA and SRAC, respectively, than SCON_AdjTBA and SCON_AdjRAC.

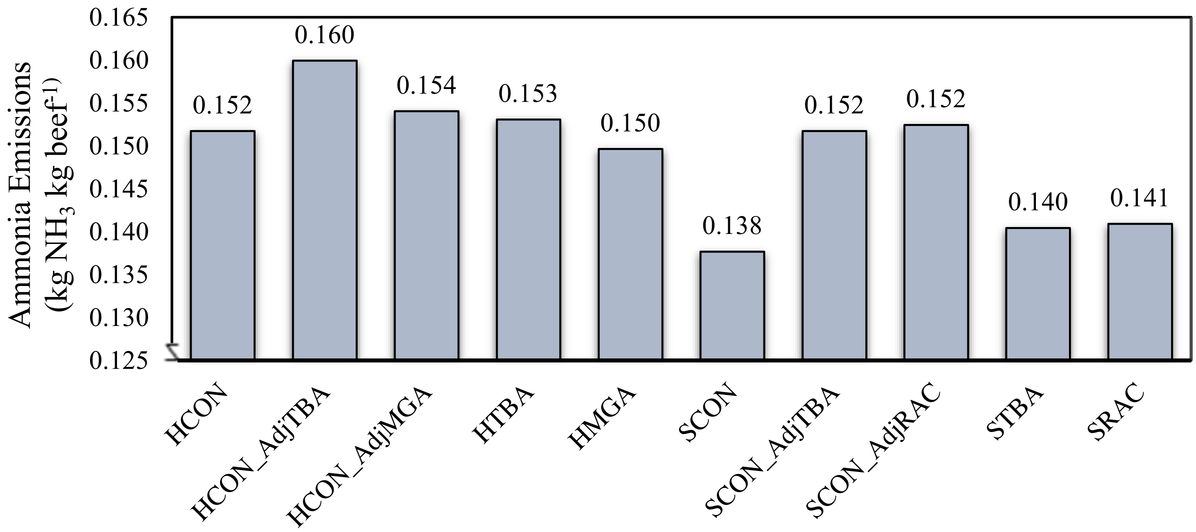

Figure 4.16 Mean ammonia emissions expressed on an intensity basis (kg NH_3 kg beef^{-1}) for feedlot cattle backgrounded and finished with or without the use of growth-enhancing technologies (n=4 trials). Treatments were **HCON** = control heifers; **HTBA** = implanted heifers; **HMGA** = heifers receiving MGA + implants; **HCON_AdjTBA** = control heifers adjusted to reach the same finished body weight (BW) as HTBA; **HCON_AdjMGA** = control heifers adjusted to reach the same finished BW as HMGA; **SCON** = control steers; **STBA** = implanted steers; **SRAC** = steers received a β-AA + implants; **SCON_AdjTBA** = control steers adjusted to reach the same finished BW as STBA; **SCON_AdjRAC** = control steers adjusted to reach the same finished BW as SRAC.

4.5 DISCUSSION

Profile of average GHG emissions

The GHG emissions profile (i.e., the mean value of all treatments) observed in this trial aligns with previously published values, as the observed enteric CH_4 emissions (44.9% of total emissions; Figure 4.5) accounted for the largest proportion of emissions from backgrounding and finishing cattle production systems. For example, Beauchemin et al. (2010) reported enteric CH_4 emissions were responsible for the greatest proportion (63%) of total GHG emissions from a Canadian beef cattle production system. The difference between the CH_4 emissions, as a proportion of the total emissions reported in the two studies, can likely be attributed to the class of cattle and type of diets examined. Beauchemin et al. (2010) included the cow-calf sector in their LCA, which accounted for 79% of the enteric CH_4 emissions and 61% of total GHG emissions. Cows are primarily fed forages, resulting in greater enteric CH_4 emissions kilogram DMI^{-1} than high concentrate diets fed in the present study.

Direct N_2O emissions associated with the manure handling system, N inputs to the cropping system, as well as soil type and texture, and indirect N_2O emissions (leaching and volatilization) in the current study accounted for 35.7% and 5.8% of total GHG emissions, respectively, (Figure 4.5) while Beauchemin et al. (2010) reported that N_2O emissions from manure and soil accounted for 23% and 4% of total emissions, respectively. These differences exist because of i) diet type; diets fed to feedlot cattle typically have a greater CP level than cow-calf diets, and ii) the manure handling system in the LCA. Similarly, energy CO_2 emissions of the feedlot system has a greater reliance on machinery for field activities to grow and harvest crops, to feed and bed cattle, and to remove manure from pens, increasing the fossil fuel use, and therefore these emissions (14.7% : Figure 4.5) as compared to the cow-calf sector (5%; Beauchemin et al., 2010). Cow-calf systems rely more on grazing, where lower energy inputs are required. The backgrounding phase had lower direct and indirect N_2O emissions than the finishing phase (8% vs. 17%) as precipitation and temperature were less conducive to promoting N_2O emissions than the finishing phase (i.e., fall and winter vs spring and summer).

The relatively low manure CH_4 emissions reported in the current study may be attributed to the type of manure handling systems (passive compost), which has a lower coefficient (0.005) compared to other manure storage types, including deep bedding (0.17) and solid storage (0.02). Further, the highly digestible feedstuffs in the backgrounding and finishing diets lead to lower manure CH_4 emissions as compared to the lower quality, forage-based diets associated with cow-calf production systems (Beauchemin et al., 2010).

The observed enteric CH_4 emissions (kg CO_2e hd^{-1}), expressed as an average of all treatments, were greater in the backgrounding than the finishing phase (Figure 4.5) due to the higher inclusion of corn silage in the backgrounding diet (60% vs. 9%) which is designed to achieve a slow rate of gain utilizing inexpensive fibrous feed. The inclusion rates are significant because lower enteric CH_4 emissions are associated with highly digestible grains, like barley, as compared to fibrous feeds, like corn silage (kg DMI^{-1}; Beauchemin et al. 2010; Knapp et al. 2014).

The GHG emissions associated with corn grain, barley grain, and corn silage accounted for 42% of total emissions (Figure 4.7) and reflected the amount of land required to produce each of the ingredients (Vergé et al. 2008). Barley grain, corn grain, and corn silage were included at 20%, 15%, and 60% of diet DM in the backgrounding diet, respectively, and 80%, 6%, and 9%, respectively, in the finishing diet. As a result, GHG emissions associated with barley grain was the highest of the three feed crops (42% of total crop emissions), based on the quantity used during both phases of the trial. Direct N_2O was responsible for the largest proportion of emissions for the crop production components (land applied manure, corn grain, barley grain, and corn silage), followed by energy CO_2 and indirect N_2O.

Total GHG emissions and GHG emission intensity

The 3 to 7% (Figure 4.9) range in the reduction of total GHG emissions (kg CO_2e kg boneless beef^{-1}) associated with GET-use as compared to weight-adjusted control cattle may be attributed to several factors, including sex, DOF and type of GET administered. More specifically, greater GHG emission reductions were more apparent for steers compared to heifers and GET cattle vs. weight-adjusted control cattle. As described by Ribeiro et al. (2020), the magnitude of response of the GET's was greater for steers than heifers (e.g., total body weight gain from TBA implants increased by 7.4 to 11.3% for heifers and steers, respectively), an outcome attributed to the higher DMI of steers than heifers. Similarly, in most cases, greater reductions were evident for HTBA vs. HMGA and SRAC vs. STBA, which is supported by the cattle performance (Ribeiro et al., 2020). The emission reductions estimated in the current study are comparable to that reported in published literature (1.1 to 28%; Basarab et al., 2012; Cooprider et al., 2011; Stackhouse et al., 2012; Webb, 2018).

Differences in the magnitude of emissions reductions between the current study and published literature can be attributed to the class of cattle included in the LCA, diet, as well as type, timing, and duration of the GET administered, and the method used to estimate emissions (measured vs modelled, as well as the model types and inputs). A recent review by Aboagye et al. (2021) summarized the environmental impacts, including GHG emissions, associated with the use of GET's in several studies conducted in North America. In Canada, Basarab et al. (2012) estimated a 4.9 to 5.1% reduction in GHG emissions for GET cattle using Holos, the same model used in our study. The primary difference between the two studies was the type and timing of

GET's administered. Basarab et al. (2012) estimated GHG emissions from heifers and steers, which were either: i) implanted with 200 mg progesterone and 20 mg estradiol benzoate at weaning and re-implanted with 120 mg TBA and 24 mg estradiol (90 to 100-d before slaughter), or ii) implanted with 200 mg progesterone and 20 mg estradiol benzoate at weaning and then four more times (80 to 90-d intervals), followed by a 120 mg TBA and 24 mg estradiol implant (90 to 100-d before slaughter). In the current study, all cattle received ionophores, heifers received three TBA/estradiol implants, or MGA, and steers received the same implant regime as the heifers, with or without the addition of RAC. In addition to type of implant, class of cattle for which emissions were estimated also differed between the two studies. In the current study, emissions were measured during the background and finishing phases, whereas Basarab et al. (2012) estimated GHG emissions from birth to finish. As far as we are aware, our study is the first that run multi-year animal experiments specifically to assess the impact of GET's on the carbon footprint of feedlot cattle.

In the United States, Stackhouse-Lawson et al. (2013) reported a considerably greater reduction in GHG emissions (kg beef^{-1}; 22%) from GET-treated feedlot compared to GET-free cattle with direct measurement of CH_4, N_2O, CO_2, and NH_3 emissions, among other parameters. Although the GET's used in the Stackhouse-Lawson et al. (2013) study and the current study were similar; i) ionophore only, ii) ionophore plus implant, and iii) ionophore plus implant plus beta-adrenergic agonists, differences in the duration of feeding were evident (365-d vs. 245 ± 18-d) differed. Further, the inclusion of ionophores as a treatment independent of the control allowed for a closer investigation of their impact on GHG emissions than the current study, including an 11% reduction in total CH_4 and a 6% reduction in CO_2 (Stackhouse-Lawson et al. 2013). In the current study, including a control group without ionophores may have resulted in a larger reduction in emissions associated with GET-use. However, as evidenced by the reduction in GHG emission intensity as a result of improved efficiency and productivity, it is clear that GET's can lower the environmental footprint of beef production per kg of beef.

Land requirements

Average crop yield and fertilizer rates (2015 to 2018), temperature (1991 to 2020), precipitation (2015 to 2018), were consistent between treatments and trial years, eliminating any potential annual variation that these parameters may have had on estimated land required to produce the necessary feedstuffs for the experiments. Therefore, variation in total land use and land requirements for feed production (i.e., corn grain, barley grain and corn silage) may be attributed to differences in cattle DMI and DOF (Table 4.9). For example, decreased land requirements observed for GET cattle were due to the increased DOF required for control cattle to reach the same finish weight (i.e., the adjusted control treatments). Similarly, Basarab et al. (2012) observed a 7.8% reduction in land use associated with GET cattle and attributed that reduction to improved feed efficiency and therefore, a reduction in the amount of feed to raise GET cattle as compared to non-treated cattle. However, it is important to note that the magnitude of land use in the current study for all GET-treatments (Table 4.10) was lower than that reported by Basarab et al. (2012; 318.5, 319.7, 403.4, and 407.3 total ha for non implanted "calf-fed," implanted "calf-fed," not implanted "yearling-fed," and implanted "yearling-fed," respectively), as the latter examined the production cycle from cow-calf to finish.

In the current study, the observed improvement in land use efficiency (ha kg boneless beef^{-1}) associated with GET heifers and steers compared to weight-adjusted control cattle can be attributed to the increased carcass weight of GET cattle (Figure 4.10; Appendix Table 8.5). Similarly, Basarab et al. (2012) observed that improved land-use efficiency was due to a 9%

increase in the carcass weight of implanted cattle. Further, Capper and Hayes (2012) reported a 9.1% increase in the amount of land required to produce feed when GET's were eliminated, which was directly related to lower animal productivity (i.e., ADG) of naturally produced cattle. As with GHG emissions, the magnitude of land use reductions was also greater for steers than heifers. It is clear that removal of GET's from cattle production systems would have a negative influence on land use.

Water requirements

Average water intake for all treatments as a proportion of total water use was relatively small (0.4%; Figure 4.13) and aligned with the results of Legesse et al. (2018a), who recently estimated water use efficiency for the Canadian cattle industry (cow-calf to finishing) and reported that drinking water accounted for <1% of the total water used in the beef production system. Average water requirements for crop production, with green water included, represented the largest share of water use (99.3%; Figure 4.11) which is also comparable to the estimated crop use of 99% reported by Legesse et al. (2018). As a percentage of the total water use for feed production, barley grain (74%) and corn grain (20%) had a much higher water footprint compared to corn silage (6%; Figure 4.10). The magnitude of these values is related to the inclusion rates of the feed ingredients in the diets.

Impact of GET's on water use

The observed reduction in total water use (m^3 water hd^{-1}), including drinking water, crop production water, and processing plant water, for HTBA (5%), HMGA (3%), STBA (10%), and SRAC (10%), compared to their respective weight-adjusted control treatments were due to the cumulative reduction in water use. The observed reduction in drinking water (m^3 water hd^{-1}; 5 to 27%) and total water for crop production (m^3 water hd^{-1}; 3 to 10%) was due to the improved production efficiency (ADG and FE) associated with GET-use (Table 4.11). The loss of productivity from GET-removal led to an increase in DOF and therefore increased water use for drinking and to produce the feed needed for this extended feeding period. When the control cattle DOF was not adjusted, GET-treated cattle had a higher water footprint than HCON and SCON.

Further, water use (m^3 water hd^{-1}) for GET-treated cattle was reduced by 4 to 12%, 3 to 5%, and 1 to 6% for barley grain, corn grain, and corn silage, respectively, which can be attributed to differences in DOF, feed required, as well as the ingredient inclusion rate. Water use in processing plants (m^3 water boneless $beef^{-1}$) was equal for all treatments, as the requirements were based on a kg of boneless beef (Table 4.12) and align with the estimates of values published by Legesse et al. (2018a).

The reduction in total water use (m^3 water hd^{-1}) observed in the study was similar to values reported by Capper (2012) who estimated an 18% reduction in water requirements for GET cattle (485,689 litres x 10^6 to produce 1.0 x 10^9 kg of beef) than GET-free cattle (572,477 x 10^6 to produce 1.0 x 10^9 kg of beef) of an entire beef production system. Capper (2012) attributed the reduction in water use for GET-treated cattle to an increase in growth rate and slaughter weight. In our study, the reduced water use for GET-treated cattle was attributed to the increased ADG, resulting in fewer DOF. In the United States, Webb (2018) estimated that the use of GET's in a cow-calf to finish production system resulted in a 1%, 5.8%, and 4.4% reduction in water use (kg H_2O kg HCW^{-1}) with ionophores, implants plus ionophore, and ionophores plus implants plus beta-adrenergic agonists (β-AA), respectively. Differences between our results and those reported by Webb (2018) may be attributed to differences in the model used to estimate water use and the location of the trial (Alberta vs. South Dakota and Nebraska), as soil type, climate, precipitation level, and crop yields differ regionally, and/or seasonally, directly impacting water use. Further, Capper and Hayes (2012) evaluated the effects of GET's (βAA + implant + ionophore + MGA) in backgrounding and finishing cattle and estimated a 7.1% reduction in the amount of water required to produce 454×10^6 kg of beef.

Nitrogen excretion

The observed decrease (2 to 6%; Figure 4.15) in nitrogen excretion (annual Mg N), and therefore potential run-off and NH_3 emissions from GET-treated heifers and steers compared to their respective control treatments, may be attributed to greater N use efficiency (Stackhouse et al., 2012) as a result of improved FE and ADG (Capper, 2012) associated with the use of the GET's in conventional production systems. Capper (2012) estimated a 17.8% reduction in N excretion with the use of GET's (399,789 t N kg beef^{-1} and 486,683 t N kg beef^{-1}, for conventional and natural treatments, respectively), which differs from the reduction in N excretion observed in the current study (2 to 6%) because Capper (2012) estimated emissions from the entire beef production system, including those from dairy calves entering the system. An additional consideration may be the feed type used (pasture, grass hay, straw, flaked corn and soybean meal, corn silage, corn grain, and alfalfa hay), as the level of CP in feedstuffs will affect N excretion levels.

Total NH_3 emissions and NH_3 emissions intensity

The observed reduction in total NH_3 emissions (Figure 4.20; HTBA vs. HCON_AdjTBA = 3% reduction, HMGA vs. HCON_AdjMGA = 1% reduction, STBA vs. SCON_AdjTBA = 7% reduction, and SRAC vs. SCON_AdjRAC = 6% reduction) and NH_3 emissions intensity (Figure 4.21; 4%, 3%, 8%, and 7% less NH_3 kg beef^{-1} associated with HTBA, HMGA, STBA, and SRAC compared to the respective control treatments) was generally lower than that reported by Stackhouse et al. (2012; 7.7 to 13.5% reduction). This greater reduction in NH_3 emissions may reflect the longer backgrounding period (84 vs.182-d) used by Stackhouse et al., (2012) as compared to the current study.

In the current study, NH_3 values reported did not include emissions associated with crop production and therefore do not include emissions resulting from fertilizer application to meet the crop nutrient requirements. However, the GHG emissions, land use, and water use associated with crop production were all a function of the cattle diets. Therefore, we can assume that the trend would be similar for NH_3 emissions as well.

Stackhouse et al. (2012) demonstrated a 13% reduction in NH_3 emissions from cattle that received β-AA compared to control cattle and a 6% reduction with β-AA compared to implanted cattle which received an estrogen and TBA implant in the stocker phase and an ionophore, tylosin, and an implant during the finishing phase. The authors suggested that despite the short duration of β-AA in the diets (20-d), the increased muscle mass (by increased protein synthesis and decreased protein degradation) resulted in lower urea N, reduced urea excretion and NH_3 volatilization (Stackhouse et al. 2012).

Subsequently, Stackhouse-Lawson et al. (2013) measured NH_3 emissions and observed a 20%, 37%, and 43% reduction in NH_3 emissions (g of NH_3 kg HCW^{-1} d^{-1}) in β-AA treated steers compared to GET-free steers receiving an ionophore only, and implanted steers, respectively. The authors noted that the β-AA (zilpaterol) in the diet impacted rumen degradation of dietary protein, increased protein deposition, reduced urinary urea excretion, and thereby decreased NH_3. The greater magnitude of response with β-AA compared to other GET treatments aligns with the results of our study.

Although the reduction in NH_3 emissions is similar to the current study, Stackhouse-Lawson et al. (2013) noted that it may be challenging to make direct comparisons to other studies due to differences in sample collection technique and climatic variation among studies. However, it is evident by current and previous studies that cattle raised with GET's have lower NH_3 emissions than cattle raised without GET's.

Study attributes and considerations

The study design offered a unique opportunity to model differences in GET-treated and control cattle resulting in a data set replicated over time. Although mean values of the environmental and crop productivity data were used in each analysis to eliminate any potential year effects, the animal production study included four replicates of six treatments with 40-hd treatment^{-1}, over a 4-yr period, providing a unique data set with 3840 observations. Further, the analysis of heifers and steers as separate treatments offered a unique opportunity to assess gender effects. As described above, the magnitude of reductions for all environmental indices was greater for steers than heifers and may be attributed to increased production efficiency of steers compared to heifers (8.1 to 16.3% increase in ADG from GET use in heifers vs. 18.7% increase in ADG from GET use in steers; Ribeiro et al. 2020). However, in commercial feedlots, heifers are typically finished at a lower BW than steers due to the increased fat to lean meat ratio (Government of Alberta n.d.), resulting in lower DP and lean yield than steers. In addition to observed differences between gender, clear differences were evident in the magnitude of response between GET's, as the impact of HTBA > HMGA and SRAC > STBA for all environmental indices, reflecting the performance outcomes (ADG, FE, carcass weights) reported by Ribeiro et al. (2020).

Emissions associated with corn DDGS were estimated by assuming that corn grain was grown on-farm, transported to a processing facility, and the resulting by-product, DDG, was transported back to the farm. The estimated emissions did not include emissions generated as a result of the fermentation of the corn to ethanol as these would be allocated to ethanol production

(Beauchemin et al. 2011). As a consequence, the environmental impact may have been overestimated compared to other strategies used to estimate use of by-product emissions, including applications such as GHG Genius (version 5.01a; http://www.ghgenius.ca) as described by (Hünerberg et al. 2014) in which co-product allocation is accounted for using an assigned emission value.

Finally, as the adoption of GET's in feedlots is much greater than other cattle management systems (i.e., cow-calf), this partial LCA allowed us to estimate the environmental benefits associated with use of these technologies during a phase of the cattle production system that consumers typically view as the most harmful to the environment.

4.6 Conclusion

Results from this study confirm that the use of GET's lower GHG and NH_3 emissions, as well as land and water use, thereby decreasing the environmental footprint associated with the backgrounding and finishing phases of beef cattle production. Improvements in ADG, FE, and slaughter weight as a result of GET-use were among the main factors that contributed to the decrease in each of the environmental indices. This study adds to the growing body of evidence that the use of GET's results in a significant reduction in the environmental footprint of beef. Although recent trade agreements and consumer demand have signalled opportunities regarding the adoption of GET-free production, the economic benefit to producers remains to be elucidated. It is clear from the research presented here and elsewhere that the environmental cost of GET removal or reduction will be significant.

5 GENERAL DISCUSSION

As indicated in this thesis, GET-use in beef production systems has positive environmental outcomes, including reduced GHG and NH_3 emissions, as well as land and water use compared to GET-free beef. Therefore, GET's are a strategy to reduce the environmental sustainability of beef production.

Gaps in Knowledge:

Gaps in knowledge regarding further benefits (i.e., habitat conservation, carbon sequestration, and biodiversity promotion) and consequences (i.e., potential environmental contamination, erosion, regional air quality) should also be included in environmental assessments that examine the effects of GET-use. However, these metrics can be challenging to define and quantify. For example, studies examining the presence of GET's in nearby ecosystems, including waterways (i.e., surface or groundwater), and the effects on living organisms are limited. However, studies to research to assess the occurrences and fate of GET residues in highly concentrated cattle areas are emerging. A recent study by Challis et al. (2021) was among the first to examine the occurrence and fate of GET's in cattle feedlots and adjacent environments. Concentrations of RAC in fresh fecal matter and pen floor manure were 3 to 4-fold greater than the concentrations of TBA and MGA. Further, concentrations of RAC in catchwater basins exceeded the levels found to cause behavioural effects in zebra fish by 5 to 32-fold.

The complexity of agroecological systems presents further challenges in the quantification of overall sustainability associated with the removal of GET's, as numerous trade-offs exist. There is a need to develop tools to quantify the trade-offs using a complete system approach, which should consider environmental, economic and social sustainability factors. Few trade-off analyses exist in the realm of Canadian beef production; however, an example of an existing study includes the comparative economic and environmental trade-off analysis of cow-calf production in Manitoba by Possberg and Kulshreshtha (2018), who reported a positive correlation between production efficiency, environmental sustainability and economic profitability. Published literature in this area is limited and therefore requires further research.

More specifically, the exclusion of GET's in cattle production systems as a result of local and international consumer demand should consider a cost-benefit analysis. It has been estimated that with full implementation, the Canada-European Union Comprehensive Economic and Trade Agreement (CETA) will result in an additional $600 million from GET-free beef export from Canada to the European Union (CAFTA, https://cafta.org/trade-agreements/canada-and-the-european-union/). Although the market for GET-free beef exports to the European Union (EU) appeared favourable at the onset of CETA, a 2019 news article in the Western Producer stated that Canada was "on pace to run a $150-million trade deficit in red meat with the European Union," as the EU has been slow to abide by its commitments, and therefore this market remains "elusive" for Canadian agri-food exporters (Arnason, 2020). On a more positive note, recent information garnered from Canada Beef indicated an increase in the value of products exported from Canada to countries that only accept "natural" beef, with margins of 6.5 to 235.9% and significant increases in volume (i.e., a 775.30% increase in MT in United Kingdom) from 2020 to 2021 (Table 5.1). However, this only represents a combined 0.7% of the total annual value of Canadian exports ($10,585,869 of $1,512,267,000) and 1.5% of the total volume of Canadian exports (2,929 MT of 195,253 MT; Canada Beef, 2021). Further, the price per kg decreased by almost two thirds (61.6% reduction in 2021 from 2020) for Canadian beef in the United Kingdom. Market fluctuation is expected, however, 2020 and 2021 were both exceptional years due to Covid-19, which has undoubtedly contributed to price volatility.

Table 5.1 Canadian exports of beef and veal products to countries only accepting natural beef as of May 2021

	Value ('000 CAD)			Volume (MT)			Price ($/kg)		
Country	2020	2021	Year Change	2020	2021	Year Change	2020	2021	Year Change
United Kingdom	2138	7182	235.90%	319	2790	775.30%	$6.71	$2.57	-61.60%
Netherlands	2023	2155	6.50%	82	95	15.70%	$24.52	$22.58	-7.90%
Italy	1501	1761	17.30%	87	107	22.90%	$17.15	$16.38	-4.50%

Source: Canada Beef, 2021

It is evident that the current and future outcomes of CETA and Canadian beef exports are complicated. However, since the CETA agreement was implemented in 2017, Canadian producers have not seen a benefit from the trade agreement (Arnason, 2020). Further, with the removal of GET's, Canadian producers will incur extra feed costs, due to the added days on feed associated with GET-free beef production. Results from this study indicated that an additional 90 to 430 kg of barley per feeding period (DM basis) are required to finish GET-free cattle, adding $35 to $170 to the cost of production (regional feed barley cost at $340 per tonne as of June 7th, 2021; Jean Préjet, personal communication). Furthermore, if the projected increased market capacity of 64,950 tonnes of GET-free beef were realized, assuming it is boneless, Canada would also bear the burden of significant environmental consequences, including an additional annual 19,485 to 46,114.5 tonnes of GHG emissions, 454,650 to 1,039,200 ha of land, 1.9×10^7 to 4.8×10^7 m^3 of water, and 259.8 to 779.4 tonnes of NH_3 compared to conventionally raised beef.

Alternatively, an incentive to produce GET-free beef is the expected premium. However, the current premium Canadian producers receive is unclear. It is estimated that a $150 hd^{-1} premium would be required to recover the costs of producing GET-free beef in Canada (Tim McAllister, personal communication), which does not account for any added costs due to additional administrative tasks required to conform to the program. Information regarding the number of cattle capturing a premium when raised as GET-free beef is not readily available.

From the information presented above, it is not clear if is economically beneficial for producers to raise beef without the use of GET's. However, there is no doubt that there is a growing demand for GET-free beef in high-income nations (i.e., North America and Europe). Aboagye et al., (2021) summarized the impact of GET's on consumer choice and found that some consumers are willing to pay large premiums for GET-free beef over conventionally produced beef (GET-free beef prices are up to 47% higher in the United States, 29% higher in Germany, 20% higher in Britain, and the premiums in Canada range from $12.13 to $30.07; CAD kg^{-1}). The willingness to purchase GET-free beef is attributed to perceived food safety, environmental, and animal welfare concerns, indicating the need for deeper discussions with consumers making decisions on these bases.

Encouraging science-based conversations and engagement with consumers:

Ensuring consumers have the knowledge and understanding of the products they demand will be critical for the viability and future success of the agriculture sector, as it is becoming increasingly important to have a social licence to operate. Therefore, developing effective strategies to capture the attention of and effectively engage with consumers (Ominski et al. 2020) should be a priority of all stakeholders in the agriculture industry. Several local and national level initiatives exist with this goal in mind. Examples of such initiatives include: Agriculture More than Ever, Agriculture in the Classroom, Open Farm Day, Great Tastes of Manitoba, and Think Beef.

Finally, the overarching goal, beyond the hypothesis, of this research was to provide science-based information for producers and consumers regarding GET-use on environmental sustainability; a critical first step in empowering producers to discuss on-farm management practices that have led to improved sustainability, as well as assisting consumers to make informed decisions about the food they purchase and consume.

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6 GENERAL CONCLUSION

Study conclusions:

- The data presented confirm our hypothesis that the use of growth-enhancing technologies (GET's) reduces the environmental footprint of beef cattle in feedlot production systems.

- Greenhouse gas emissions (kg CO_2e kg boneless beef^{-1}) were 3 to 7% greater in backgrounding and finishing cattle raised without GET's. Use of GET's led to a 5 to 10% reduction in land (ha kg boneless beef^{-1}), a 6 to 12% reduction in water (m^3 H_2O kg boneless beef^{-1}), and a 3 to 8% reduction in ammonia (NH_3; kg NH_3 kg boneless beef^{-1}).

- The range in magnitude of response observed here and in other studies is due to the effects GET's on average daily gain and feed efficiency, as well as gender. In all cases, cattle receiving GET's had improved environmental sustainability than GET-free cattle.

- This study will add to the shortlist of previously published Canadian and North American studies investigating the environmental effects of reducing productivity from GET removal in beef production systems.

- To the author's knowledge, this study was the first in Canada to use animal production data to assess the environmental effects of GET-removal from Canadian cattle production systems, indicating the novelty of, and need for additional research on this topic.

Future considerations:

- Further research to examine additional effects of GET-use on the environmental footprint of beef production, economic analyses, including effects on international and domestic markets, and development of trade-off analyses are required to develop a more comprehensive, whole-systems understanding.

- An important note for the design of future environmental modeling studies is the strategy used to determine the end of the feeding period (i.e., days on feed or final carcass weight). Comparable carcass weight is necessary to make direct comparisons and avoid the need for weight-adjustments to compare environmental outcomes.

- There is a need for a greater understanding of consumer concern regarding GET-use (i.e., environmental, food safety, welfare) and the subsequent development of programs to effectively engage and communicate with target audiences to disseminate information with an emphasis of the shared value between beef producers and consumers.

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8 APPENDIX

Table 8.1 Treatment descriptions

Treatment	Abbrev.	Added product?	Product type	Product name	Product active ingredient	Dosage
Heifer Control	HCON	No	-	-	-	-
Heifer Implant	HTBA	Yes	Implant	Component TE-100	Trenbolone acetate (100 mg) Estradiol (10mg) Tylosin tartrate (29 mg)	Two implants: 90-d interval
			Implant	Component TE-200	Trenbolone acetate (200 mg) Estradiol (20 mg) Tylosin tartrate (29 mg)	One implant: 90-d following the second implant
Heifer MGA	HMGA	Yes	In-feed product	Melengestrol acetate	Melengestrol acetate	0.40 mg $heifer^{-1}$ d^{-1}
Steer Control	SCON	No	-	-	-	-
Steer Implant	STBA	Yes	Implant	--------------------- Same as HTBA -------------		
Steer Implant + Ractopamine	SRAC	Yes	Implant	--------------------- Same as HTBA -------------		
			In-feed product	Optaflexx	Ractopamine hydrochloride	30 mg kg^{-1} diet in the final 42-d before slaughter

Table 8.2 Treatment, phase, and trial specific performance (initial body weight (BW), days on feed (DOF), average daily gain (ADG), and dry matter intake (DMI)) Holos model inputs of hiefers (Ribeiro et al., 2020)

Phase	Parameter	Trial	HCON	HCON _AdjTBA	HCON _AdjMGA	HTBA	HMGA
Backgrounding	Initial BW, kg	1	271.3	271.3	271.3	270.3	271.6
		2	277.8	277.8	277.8	279.5	278.1
		3	283.8	283.8	283.8	285.0	284.6
		4	292.6	292.6	292.6	293.4	293.6
	DOF	1	84.0	84.0	84.0	84.0	84.0
		2	84.0	84.0	84.0	84.0	84.0
		3	84.0	84.0	84.0	84.0	84.0
		4	84.0	84.0	84.0	84.0	84.0
	ADG, kg d^{-1}	1	1.00	1.00	1.00	1.03	1.08
		2	1.10	1.10	1.10	1.26	1.24
		3	1.23	1.23	1.23	1.29	1.21
		4	0.96	0.96	0.96	1.06	0.92
	DMI, kg d^{-1}	1	6.9	6.9	6.9	6.6	7.0
		2	7.1	7.1	7.1	7.4	7.0
		3	7.2	7.2	7.2	7.6	7.1
		4	7.0	7.0	7.0	7.2	6.6
Finishing	Initial BW, kg	1	396.1	396.1	396.1	397.1	402.5
		2	405.1	405.1	405.1	426.5	422.3
		3	408.9	408.9	408.9	418.7	417.0
		4	414.2	414.2	414.2	420.6	408.3
	DOF	1	151.0	168.0	164.0	151.0	151.0
		2	139.0	176.0	157.0	139.0	139.0
		3	147.0	171.0	146.0	147.0	147.0
		4	160.0	197.0	184.0	160.0	160.0
	ADG, kg d^{-1}	1	1.25	1.25	1.25	1.40	1.33
		2	1.33	1.33	1.33	1.59	1.44
		3	1.47	1.47	1.47	1.69	1.46

	4	1.41	1.41	1.41	1.64	1.59
DMI, kg d^{-1}	1	9.1	9.1	9.1	9.7	9.4
	2	9.9	9.9	9.9	10.8	10.3
	3	10.0	10.0	10.0	10.8	9.8
	4	9.8	9.8	9.8	10.9	10.1

Table 8.3 Treatment, phase, and trial specific performance (initial body weight (BW), days on feed (DOF), average daily gain (ADG), and dry matter intake (DMI)) Holos model inputs of steers (Ribeiro et al., 2020)

Phase	Parameter	Trial	SCON	SCON _AdjTBA	SCON _AdjRAC	STBA	SRAC
Backgrounding	Initial BW, kg	1	266.0	266.0		264.9	
		2	286.9	286.9		287.4	
		3	287.8	287.8	287.8	289.6	285.3
		4	266.3	266.3	266.3	265.3	265.3
	DOF	1	84.0	84.0		84.0	
		2	84.0	84.0		84.0	
		3	84.0	84.0	84.0	84.0	84.0
		4	84.0	84.0	84.0	84.0	84.0
	ADG, kg d^{-1}	1	1.18	1.18		1.20	
		2	1.18	1.18		1.42	
		3	1.19	1.19	1.19	1.36	1.40
		4	1.05	1.05	1.05	1.14	1.15
	DMI, kg d^{-1}	1	6.6	6.6		6.9	
		2	7.3	7.3		7.7	
		3	7.4	7.4	7.4	7.9	7.9
		4	6.6	6.6	6.6	6.5	6.7
Finishing	Initial BW, kg	1	411.9	411.9		417.9	
		2	421.5	421.5		455.0	
		3	419.6	419.6	419.6	442.4	439.8
		4	396.1	396.1	396.1	414.5	410.1
	DOF	1	151.0	189.0		151.0	
		2	139.0	204.0		139.0	
		3	147.0	193.0	197.0	147.0	147.0
		4	160.0	168.0	196.0	160.0	160.0
	ADG, kg d^{-1}	1	1.40	1.40		1.74	
		2	1.39	1.39		1.85	
		3	1.53	1.53	1.53	1.92	1.96
		4	1.58	1.58	1.58	1.91	2.02

DMI, kg d^{-1}	1	9.6	9.6		11.1	
	2	9.7	9.7		11.5	
	3	10.3	10.3	10.3	11.4	11.8
	4	9.8	9.8	9.8	11.3	11.3

Table 8.4 Adjustment of control treatments using initial body weights (BW) and average daily gain (ADG) to achieve the same finished BW of growth-enhancing technology-treated cattle in each trial

Treatment	Trial	Finishing period initial BW (kg)	Finishing period ADG (kg d^{-1})	Target adjusted finished BW (kg)	Total DOF	Increased DOF from control
HCON_AdjTBA	1	396.1	1.25	607.0	168	17
HCON_AdjMGA	1	396.1	1.25	601.3	164	13
SCON_AdjTBA	1	411.9	1.4	677.3	189	38
HCON_AdjTBA	2	405.1	1.33	640.3	176	37
HCON_AdjMGA	2	405.1	1.33	615.0	157	18
SCON_AdjTBA	2	421.5	1.39	706.3	204	65
HCON_AdjTBA	3	408.9	1.47	661.7	171	24
HCON_AdjMGA	3	408.9	1.47	623.6	146	-1
SCON_AdjTBA	3	419.6	1.53	713.9	193	46
SCON_AdjRAC	3	419.6	1.53	719.7	197	50
HCON_AdjTBA	4	414.2	1.41	672.6	184	24
HCON_AdjMGA	4	414.2	1.41	650.4	168	8
SCON_AdjTBA	4	396.1	1.58	705.9	196	36
SCON_AdjRAC	4	396.1	1.58	718.3	204	44

Table 8.5 Carcass outcomes (finished body weight (BW), hot carcass weight (HCW), dressing percentage (DP), and boneless beef) of heifers backgrounded and finished with or without the use of growth-enhancing technologies (n = 4 trials)

		Treatment				
Parameter	Trial	HCON	HCON _AdjTBA	HCON _AdjMGA	HTBA	HMGA
Finished BW, kg	1	583.4	607.0	601.3	607.0	601.3
	2	585.7	640.3	615.0	640.3	615.0
	3	618.5	661.7	623.6	661.7	623.6
	4	630.5	672.6	650.4	672.6	650.4
HCW, kg	1	350.3	364.4	361.0	370.3	369.6
	2	353.7	386.7	371.4	388.9	373.9
	3	366.3	391.9	369.3	399.3	373.5
	4	376.7	401.8	388.6	410.1	395.7
DP, %	1	60.0	60.0	60.0	61.0	61.5
	2	60.4	60.4	60.4	60.7	60.8
	3	59.2	59.2	59.2	60.4	59.9
	4	59.7	59.7	59.7	61.0	60.9
Boneless beef, kg	1	248.7	258.8	256.3	262.9	262.4
	2	251.2	274.6	263.7	276.1	265.5
	3	260.1	278.2	262.2	283.5	265.2
	4	267.4	285.3	275.9	291.2	281.0

Table 8.6 Carcass outcomes (finished body weight (BW), hot carcass weight (HCW), dressing percentage (DP), and boneless beef) of steers backgrounded and finished with or without the use of growth-enhancing technologies (n = 4 trials)

		Treatment				
Parameter	Trial	SCON	SCON _AdjTBA	SCON _AdjRAC	STBA	SRAC
Finished BW, kg	1	621.9	677.3		677.3	
	2	611.5	706.3		706.3	
	3	635.0	713.9	719.7	713.9	719.7
	4	634.6	705.9	718.3	705.9	718.3
HCW, kg	1	376.9	410.5		410.7	
	2	374.3	432.2		435.9	
	3	378.7	425.7	429.2	431.1	441.5
	4	381.3	424.2	431.6	429.4	439.6
DP, %	1	60.6	60.6		60.6	
	2	61.2	61.2		61.7	
	3	59.6	59.6	59.6	60.4	61.3
	4	60.1	60.1	60.1	60.9	61.3
Boneless beef, kg	1	267.6	291.4		291.6	
	2	265.7	306.9		309.5	
	3	268.9	302.2	304.7	306.1	313.5
	4	270.7	301.2	306.5	304.9	312.1

Table 8.7 Model inputs for the land area (ha) required to grow the feedstuffs (corn silage, barley grain and corn grain) included in the heifer diets (n= 4 trials; Ribeiro et al., 2020)

		Treatment				
		HCON	HCON _AdjTBA	HCON _AdjMGA	HTBA	HMGA
Ingredient	Trial	Land use, ha				
Corn silage	1	2.91	3.00	2.97	2.89	2.98
	2	2.99	3.20	3.09	3.15	3.00
	3	3.07	3.20	3.06	3.26	3.03
	4	3.06	3.19	3.11	3.23	2.96
Barley grain	1	9.75	10.76	10.50	10.24	9.99
	2	9.77	12.14	10.94	10.64	10.16
	3	10.34	11.90	10.25	11.23	10.15
	4	10.94	12.43	11.44	12.15	11.26
Corn grain	1	3.10	3.27	3.23	3.14	3.17
	2	3.16	3.56	3.36	3.36	3.21
	3	3.27	3.54	3.26	3.51	3.22
	4	3.34	3.59	3.42	3.58	3.30
Total	1	15.76	17.03	16.70	16.27	16.14
	2	15.92	18.91	17.39	17.15	16.36
	3	16.68	18.64	16.57	18.00	16.40
	4	17.34	19.21	17.97	18.96	17.53

Table 8.8 Model inputs for the land area (ha) required to grow the feedstuffs (corn silage, barley grain and corn grain) included in the steer diets (n= 4 trials; Ribeiro et al., 2020)

		Treatment				
		SCON	SCON _AdjTBA	SCON _AdjRAC	STBA	SRAC
Ingredient	Trial	Land use, ha				
Corn silage	1	2.86	3.07		3.09	
	2	3.04	3.40		3.30	
	3	3.14	3.40	3.42	3.42	3.45
	4	2.93	3.13	3.17	3.06	3.09
Barley grain	1	10.17	12.53		11.65	
	2	9.63	13.70		11.25	
	3	10.64	13.65	13.90	11.81	12.18
	4	10.89	13.17	13.66	12.51	12.49
Corn grain	1	3.11	3.52		3.43	
	2	3.18	3.87		3.53	
	3	3.35	3.87	3.91	3.68	3.74
	4	3.24	3.63	3.71	3.51	3.53
Total	1	16.14	19.11		18.17	
	2	15.85	20.97		18.08	
	3	17.14	20.92	21.24	18.90	19.37
	4	17.06	19.93	20.55	19.08	19.11

Table 8.9 Crop data (planting date, stage of development length, and crop coefficients) required for calculating water use intensity (WUI) of the feedstuffs included in the backgrounding and finishing diets

	Crop		
Parameter	Barley grain	Corn grain	Corn silage
Planting date[z]	06-May	10-May	15-May
Stages of development length, d[y]			
L_{init}	20	30	30
L_{dev}	25	40	40
L_{mid}	60	50	50
L_{late}	30	21	19
Crop Coefficients[x]			
$K_{c\ init}$	0.3	0.3	0.3
$K_{c\ mid}$	1.15	1.2	1.2
$K_{c\ end}$	0.25	0.35	0.35
$K_{c\ dev}$	0.03	0.026	0.026
$K_{c\ late}$	-0.03	-0.04	-0.045

[z]Huffman et al. (2015)

[y]Allen et al. (1998)

[x]Allen et al. (2007; 1998); ASCE (1996)