Impact of bread waste inclusion in feedlot diets on the environmental footprint of growing and finishing beef cattle

By

Janine Elizabeth Ingrid Hansen

A thesis submitted to the Faculty of Graduate Studies of

The University of Manitoba

In Partial Fulfillment of the Requirements of the Degree of

MASTER OF SCIENCE

Department of Animal Science

University of Manitoba

Winnipeg

ABSTRACT

The objective of this research was to examine the impact of including bread waste in feedlot diets on the environmental footprint of beef cattle. Existing data including diet composition and animal performance metrics (body weight, average daily gain, dry matter intake, feed:gain ratio) were obtained from two previous feeding trials in which steers were fed bread by-product (BBy) at rates of 40% (growing steers) and 55% DM (finishing steers). Environmental footprint metrics were estimated through modeling approaches and included land use requirements, greenhouse gas (GHG) emissions, ammonia (NH_3) emissions, and water use requirements. Including bread waste in the diets of growing steers resulted in a 45% decrease in land use (ha hd^{-1}) for feed crops, a 14% decrease in GHG emission intensity (kg CO_2e hd^{-1}), a 4% decrease in NH_3 emission intensity (kg NH_3 hd^{-1}), and a 37% decrease in water use intensity (L kg^{-1} live weight) compared to steers fed a conventional corn-based diet. Finishing steers fed a BBy-based diet had a 63% reduction in land use (ha hd^{-1}) for feed production, a 19% reduction in GHG emission intensity (kg CO_2e hd^{-1}), 1% reduction in NH_3 emissions (kg NH_3 hd^{-1}), and a 61% reduction in water use intensity (L kg^{-1} live weight). Furthermore, GHG emissions associated with BBy from production to waste management were 24% and 53% lower when diverting BBy from landfill to growing and finishing diets, respectively. Utilizing bread waste in feedlot diets not only reduces the environmental footprint of growing and finishing cattle but makes use of land, water and fertilizer resources that have already been expended. Furthermore, as bread waste is priced lower than conventional feedstuffs, its inclusion in feedlot diets is expected to reduce the cost of production for growing and finishing cattle. Despite the benefits,

current challenges that must be considered include availability and proximity of bread waste to feedlots, short shelf life, and regulatory restrictions.

FOREWORD

This thesis was written in accordance with the University of Manitoba Faculty of Graduate thesis guidelines and follows the Canadian Journal of Animal Science manuscript style format. It consists of an abstract, introduction, materials and methods, results, discussion, and conclusion. At this time, it has not been submitted for publication.

CONTRIBUTION OF AUTHORS

J.E.I. Hansen: Conducted modeling, result interpretation, writing of thesis document and subsequent editing; K.H. Ominski: Supervision, conceptualization, reviewing, and editing; T. A. McAllister, K. Stanford, S.A. Terry, and M.R.C. Cordeiro: Reviewing and editing; G. Legesse, G. Mengistu, I. Aboagye: Spreadsheet development; D. Fulawka: Review of model input and output data.

ACKNOWLEDGEMENTS

I would like to express my deepest gratitude to Dr. Kim Ominski, whose guidance, support and encouragement have been paramount in the completion of this thesis. Truly, there is no possibility in which I would be where I am today without her patience, tenacity and inspiration. I have never had a mentor, academic or otherwise who has demonstrated such confidence in my ability to succeed. Through her leadership and passion, I will forever be reminded to “take my seat at the table” and make my voice heard. I am profoundly grateful for the opportunity to learn from her. Thank you for making this journey a rewarding and memorable experience.

I would like to extend my heartfelt thanks to my committee members, Tim McAllister, Kim Stanford, Stephanie Terry, and Marcos Cordeiro, for all their support and encouragement throughout this program. Their insightful feedback has been invaluable and has greatly enhanced the clarity and depth of my research. Thank you to the Beef Cattle Research Council and the Mitacs Accelerate Internship for providing the financial support to conduct this research. To Deanne Fulawka, Dr. Genet Mengistu, Dr. Isaac Aboagye, and Mireille Krul, thank you for your help and support with Holos, spreadsheets, and tracking down articles.

Thank you to my friends and family who have been patient and supportive throughout the challenges and enthusiastic to celebrate the successes. I am beyond grateful for Talyia who has been there for me during the most challenging moments, calmed me down, made me laugh, told me I could overcome whatever I was facing, and made this experience undeniably more worthwhile.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

ABBREVIATIONS

AAFC	Agriculture and Agri-Food Canada
ADF	Acid detergent fibre
ADG	Average daily gain
BBy	Bread by-product
BSE	Bovine spongiform encephalopathy
CF	Crude fat
CFIA	Canadian Food Inspection Agency
CH_4	Methane
CO_2	Carbon dioxide
CO_2e	Carbon dioxide equivalents
CP	Crude protein
DM	Dry matter
DMI	Dry matter intake
DOF	Days on feed
ECCC	Environment and Climate Change Canada
ET	Evapotranspiration
EU	European Union
FAO	Food and Agriculture Organization of the United Nations
FBBy	Finishing steers (all weights) fed the bread by-product-based diet
FC	Finishing steers (all weights) fed the corn-based diet
FLW	Food loss and waste
FPCM	Fat-and-protein corrected milk
GBBy	Growing steers fed the diet including bread by-product
GBP	Grocery by-product
GC	Growing steers fed the diet including corn
GHG	Greenhouse gas
GWP	Global warming potential
H	Heavy weight

HBBy	Heavy finishing steers fed the bread by-product-based diet
HC	Heavy finishing steers fed the corn-based diet
ha	Hectare
HCW	Hot carcass weight
IPCC	Intergovernmental Panel on Climate Change
L	Light weight
LBBy	Light finishing steers fed the bread by-product-based diet
LC	Light finishing steers fed the corn-based diet
LCA	Life cycle assessment
LW	Live weight
M	Medium weight
MBBy	Medium finishing steers fed the bread by-product-based diet
MC	Medium finishing steers fed the corn-based diet
MG	Metabolizable glucose
MP	Metabolizable protein
N	Nitrogen
NDF	Neutral detergent fibre
NH_3	Ammonia
N_2O	Nitrous oxide
NRC	National Research Council Canada
NZWC	National Zero Waste Council
SBM	Soybean meal
SD	Standard deviation
TDN	Total digestible nutrients
UNEP	United Nations Environment Programme
US EPA	U.S. Environmental Protection Agency
WRAP	Waste and Resources Action Programme
WUI	Water use intensity

1 GENERAL INTRODUCTION

Despite significant advancements in agricultural productivity, the number of people in the world that experience hunger or food insecurity has been increasing since 2014 (FAO, 2020). However, there is no global food shortage but rather an exorbitant amount of food that is lost or wasted. Globally, 2.5 billion tonnes of edible food are discarded annually (World Food Program USA, 2021) with only 30-50% of the food produced being eaten (Babbitt et al., 2022). This loss of food from production through to waste management not only exacerbates world hunger but also further impacts global climate change by emitting over 9.3 billion tonnes of carbon dioxide equivalents (CO_2e), annually (Zhu et al., 2023). Indeed, food loss and waste (FLW) accounts for almost 60% of the Canadian food industry's environmental footprint (Nikkel et al., 2019). In addition, $940 billion is spent globally on food that is ultimately wasted, annually (US EPA, 2021). As the global population is estimated to grow to 9.7 billion people by 2050 (United Nations, 2022), food systems must not only provide sufficient nutrition but do so in an environmentally sustainable fashion with efficient use of resources. While animal agriculture is estimated to produce 14.0-17.3% of global greenhouse gas (GHG) emissions (Xu et al., 2021), it is also the source of products with highly bioavailable protein and many critical micronutrients important for human health (Randolph et al., 2007). Feeding food waste to livestock presents a possible multi-faceted mitigation strategy by transforming waste into high quality nutrients from animal products while reducing the environmental burden of both animal agriculture and food waste. Addressing these interlinked challenges requires an approach that includes sustainable agricultural practices, efficient food production and distribution systems, and robust programs and policies to mitigate climate change and reduce FLW.

2 LITERATURE REVIEW

2.1 Food loss and waste

Food produced for the human population that is not consumed can be primarily categorized as either food loss or food waste, often differentiated by where in the food chain it has been discarded. Food loss refers to any food or commodity that was damaged, lost or destroyed before it could be consumed by humans (World Food Program USA, 2021), thus being discarded during production, processing, or manufacturing (Gooch et al., 2019). Some losses are planned and are limited in value for human consumption such as screenings, hulls, and peels (Gooch et al., 2019), while unplanned losses consist of fresh produce left in fields, mishandled food that cannot proceed to distribution, and commodities that cannot continue to retail as they do not meet consumer standards in physical appearance (Nikkel et al., 2019) or in their chemical profile (i.e. malt barley). Food waste is defined as human edible food that is deliberately discarded before or after it spoils at the consumption stage by retailers, household consumers or service providers such as restaurants. These definitions can vary between sources. For example, a Canadian report by Second Harvest defined produce deemed unacceptable for distribution for cosmetic reasons as food loss (Nikkel et al., 2019) whereas the World Food Program (2021) considers it food waste. Due to these differences in definitions, this study will refer to them collectively as FLW.

2.1.1 Regional differences in FLW

In 2011, the FAO released a report that quantified the per capita FLW at pre-consumption (production up to but not including retail) and consumption (retail onwards) stages for the following regions: Europe, North America and Oceania, Industrialized Asia, Sub-Saharan African, North Africa and West and Central Asia, South and Southeast Asia, and Latin America (FAO, 2011). Six of the seven regions, excluding South and Southeast Asia, had pre-consumption FLW above 150 kg person^{-1} year^{-1}. The North American and Oceania regions reported consumption FLW at 115 kg person^{-1} year^{-1} and ranked highest in consumption FLW for six of seven commodity groups, including cereals, roots and tubers, fruits and vegetables, meat, fish, and dairy. Together, these two regions ranked third for consumption FLW of oilseeds and pulses. Higher consumption FLW was reported in mid- to high-income regions such as Europe, North America and Oceania, and industrialized Asia compared to low-income regions (FAO, 2011). The dominant portion of FLW in low-income regions occurs pre-consumption and is due to issues in post-harvest handling such as pests as well as storage availability, transportation (including infrastructure) and a lack of preservation technologies (FAO, 2011). Lipinski et al. (2013) examined FLW across geographic regions in calories rather than on a weight basis, as food types vary in their water and caloric content per kilogram. North America and Oceania ranked first at 1520 kcal in terms of wasted food (person^{-1} d^{-1}) followed by Europe and industrialized Asia at 748 and 746 kcal (person^{-1} d^{-1}), respectively. It is apparent that there are significant differences between countries with opportunities for reduction, particularly in North America and Oceania, with a two-fold higher caloric loss compared to Europe and Asia.

2.1.2 Food loss and waste in Canada

Despite Canada’s classification as a high-income country, 4 million Canadians experience food insecurity, while 58% of all food produced nationally is destined for landfills (Nikkel et al., 2019). Gooch et al. (2010) identified the primary contributors to FLW in Canada, as overproduction, defects in products as well as unnecessary inventory (especially in households) with the latter accounting for 51% of the total FLW observed along the agri-food value chain – the single biggest contributor to food waste. In addition, the National Zero Waste Council (NZWC; 2022) has suggested that 63% of the food discarded by Canadians was fit for consumption at the time of disposal. These losses are largely due to retail marketing tools and consumer behaviour. Canadian retailers often use price-reduction promotions when food is bought in bulk, which encourages consumers to buy quantities beyond their needs (Gooch et al., 2010). This often leads to preparation of surplus amounts, resulting in leftovers and/or not utilizing products before their quality has declined to a point where they are perceived as unappealing or unsafe for consumption. Use of "best before" and even expiry dates are infrequently based on food safety but instead are used to protect brands and manage quality perception (Nikkel et al., 2019). This encourages consumers to discard food and beverages that are still safe for human consumption. Furthermore, retailers often have "sell by" dates and items that have reached that date will be removed from shelves and discarded, even though this food would be safe to consume, further contributing to 1.31 million tonnes of annual retail FLW in Canada (Gooch et al., 2019).

Strategies to improve the environmental sustainability of food systems often focus on reducing transportation emissions and plastic packaging. However, Gooch et al. (2010) found that transportation or "food miles" accounted for only 3% of food waste in Canada. Further, the Waste and Resource Action Programme (WRAP, 2022) estimated that more GHGs are produced from wasted food than from plastic packaging. Despite the negative effect plastic pollution has on the environment (UNEP, 2024a), plastic packaging does offer some benefits by extending shelf life of food and reducing spoilage (Helmcke et al., 2022).

2.1.3 Economic impact of FLW

The negative economic impact associated with food loss affects producers, processors and consumers. Producers are not compensated for their investment in production resources (seed, water, fertilizer etc.) when food becomes lost along the value chain, regardless of if there are significant price increases to consumers (Lipinski et al., 2013). For example, the mishandling of delicate produce, such as raspberries, can reduce the amount of sellable product by as much as 75%, with the remainder sold at a higher price to ensure profitability for retailers (Gooch et al., 2010). At the processing stage, inaccurate market demand forecasts can also contribute to food loss. If processors are unable to fill orders due to shortages, they may be financially penalized or otherwise. To avoid this scenario, surplus quantities are ordered from producers which may leave processors with excess product that cannot be sold before spoilage (Nikkel et al., 2019).

Nikkel et al. (2019) estimated that $50 billion could be saved along the Canadian food supply chain by eliminating avoidable food loss and waste. However, these losses could be closer to $107 billion per year when the cost of waste inputs, such as labour and transport are included (Gooch and Felfel, 2014). In Canada, household waste alone has been estimated at $20 billion across the country, as the average household wastes $1,300 of food each year (NZWC, 2022).

2.1.4 Environmental impact of FLW

Food that is directed to landfills and large-scale or household compost represents a loss of resources beyond the food itself. The land, energy, water, and other inputs such as fertilizer that were used to produce the food have also been wasted, while at the same time additional and unnecessary carbon dioxide (CO_2) and methane (CH_4) are produced and emitted into the atmosphere (FAO, 2011). Many consumers believe discarded food (especially when composted) decomposes and the nutrients within are returned to the soil. However, decomposing food releases large amounts of CH_4 into the atmosphere due to the presence of methanogens that live in anaerobic environments such as landfills (Mitloehner, 2020). Methane is a GHG that is 28 times more potent than CO_2 over a 100-year period, despite its relatively short lifespan. Methane only remains in the atmosphere for 12-15 years whereas CO_2 can remain in the atmosphere for thousands of years (US EPA, 2024). Greenhouse gas emissions from food waste account for 8-10% of the global emissions annually (UNEP, 2021). In Canada, 56.5 million tonnes of CO_2 equivalent emissions are created annually by food waste (Gooch et al., 2019). Improved utilization of the current food supply by decreasing FLW would reduce GHG emissions from the agricultural sector, as well as the need for fertilizers, processing, and transport (Lipinski et al., 2013).

In addition to the reduction in GHG emissions, decreasing food waste can also have a land and water-use sparing effect. The land associated with producing FLW accounts for 28% of global agricultural area (FAO, 2013). Furthermore, approximately 173 billion m^3 of water are used annually to produce food that is wasted or lost, accounting for close to one quarter of all water used for global agriculture (Lipinski et al., 2013). For every tonne of FLW occurring in Canada, 128 tonnes of water are wasted, accounting for close to 60% of the food industry’s water footprint (Gooch et al., 2019). Therefore, more efficient utilization of the food already produced would reduce the need to draw more water from the earth’s reserves (Lipinski et al., 2013).

2.2 Livestock feed and the role of livestock in mitigating FLW

The Organization for Economic Cooperation and Development, in collaboration with the FAO (2020), determined that the agriculture industry uses 40% of the earth’s land, 70% of which is pasture. Globally, 6 billion tonnes of dry matter (DM) are fed to livestock each year; however, 86% of this consists of human inedible sources and 57% of the land is used to produce forage and is unsuitable for cultivation of crops for consumption by humans (Mottet et al., 2017). Historically, human-inedible food, such as processing by-products, have not been considered as FLW (MacRae et al., 2016). However, Gooch et al. (2019) included and categorized by-products as “planned losses” to more accurately identify inefficiencies and opportunities in the agri-food system. It is important to note that without an end use for these residues, the processing of human food would further present an environmental burden (Mottet et al., 2017). It has been estimated that 30% of global livestock feed already consists of residues, co-products or by-products from food processed for human consumption, accounting for close to 1.6 billion tonnes of DM (Mottet et al., 2017). As swine, poultry and beef feedlot diets rely heavily on concentrate feed (Puente-Rodriguez et al., 2022), these operations can use processing residues to meet nutrient requirements of diets including fibre, protein, and energy. Utilizing FLW as animal feed not only mitigates waste but also reduces the need to grow conventional feed that relies on costly resources (Rajeh et al., 2020).

A circular food economy aims to close the loop of products as well as reduce both resource consumption and waste disposal into the environment (Jurgilevich et al., 2016). Although reducing the surplus of food produced is the ideal solution, alternative strategies include donating excess food (that is still human-edible) to shelters and food banks, and feeding it to livestock (Figure 2.1). Including surplus food in livestock diets provides another opportunity for food that has used land, water, and fertilizer resources in its production to be utilized to generate high quality products for human consumption in the form of meat, milk, and egg products. Feeding FLW to livestock creates an opportunity for products that may no longer meet quality standards for humans to remain in the circular food economy.

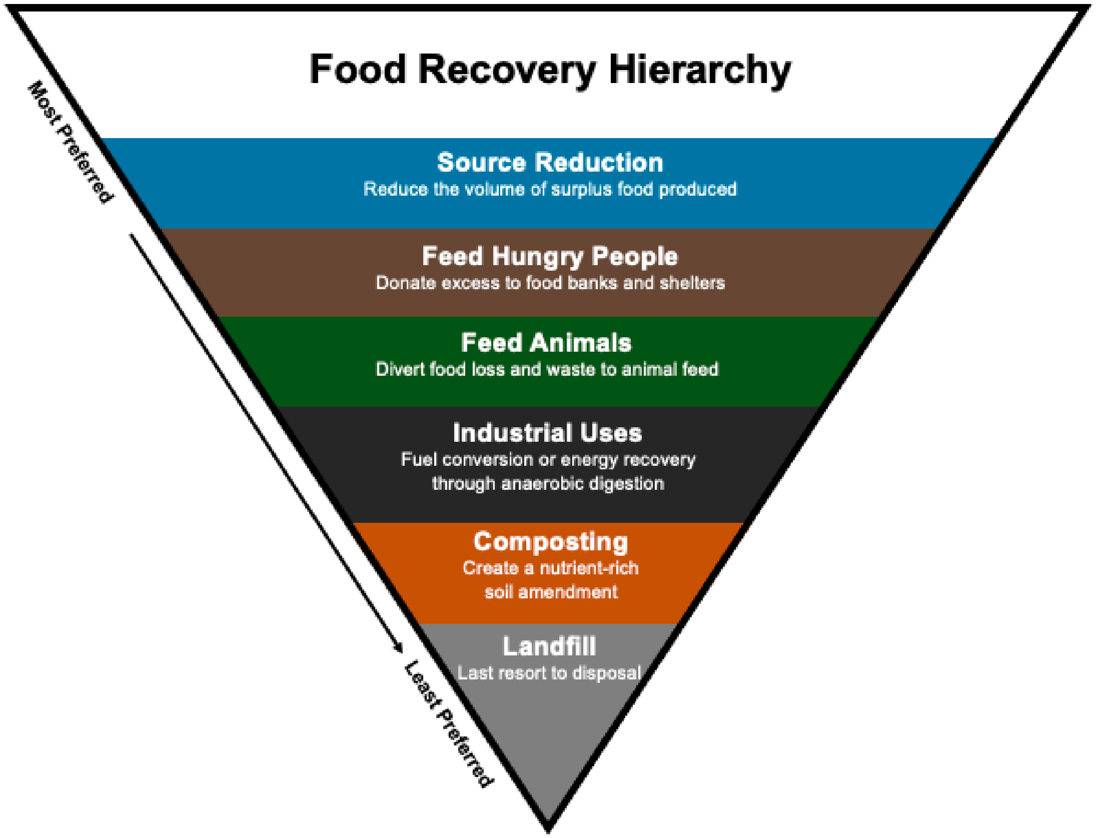

Figure 2.1 Food waste reduction solution hierarchy (Adapted from US EPA, 2021)

Most fruits and vegetables must meet high cosmetic standards to be accepted by retailers and consumers, which results in produce that is deemed undesirable and rejected post-harvest (MacRae et al., 2016). In many cases, this produce can be fed to livestock, providing crop farmers with various benefits including financial compensation, commodity exchange (i.e. manure) or no cost for removal and disposal. In exchange, livestock producers receive valuable low-cost feed.

2.2.1 Impact of FLW on environmental sustainability of ruminant production systems

Ruminant-based food production, particularly beef, has been identified as a significant source of GHG emissions due to production of CH_4 via enteric fermentation as well as the production of feedstuffs, such as cereal grains (Desjardins et al., 2012). Not only does the production of feed crops have an environmental impact, but it is also the subject of feed vs. food debates. Utilizing FLW in ruminant diets allows for the production of high-quality animal-based protein while reducing the number of harvested crops required to feed animals; thereby further supporting production of human-edible food as the highest priority.

Plant-based protein and dairy alternatives are growing in popularity due to increased concern about the effects of animal agriculture on climate change and human health (World Animal Protection, 2021). However, processing of plant-based commodities can produce substantial quantities of by-products/co-products that are not consumed by humans. Roquette in Portage la Prairie, Manitoba operates the world’s largest pea protein processing plant and relies on the consumption of their co-products by livestock for economic viability. Pea cream, pulp and screenings are human-inedible products from the refining of peas for human consumption (Roquette, 2022). Directing these co-products to landfills would present a significant environmental burden while eliminating a valuable feed resource. Van Zanten et al. (2018) reported that human consumption of products from animals that were fed diets comprised of FLW would reduce the amount of required arable land by 25%, compared to if only plant-based food was consumed. Additionally, complete removal of livestock from the agri-food landscape would not only reduce the upcycling of by-products/co-products but may also compromise the ability to meet the nutrient requirements of humans (Mottet et al., 2017). Although there are many plant-based proteins that provide protein and a variety of micro-nutrients, it is difficult to consume adequate quantities of all required micro-nutrients on a plant-based diet (Mottet et al., 2017).

2.2.2 Challenges of utilizing FLW for livestock feed

Barriers to further incorporating FLW into livestock diets include availability, cost and logistics of transport, storage, as well as unique challenges including federal approval. Although there are some relatively new organizations coordinating the redistribution of FLW for use in livestock diets, increased resources and effective communication are needed to mitigate challenges associated with FLW.

The Canadian Food Inspection Agency requires all feedstuffs to be reviewed and assessed for them to be approved as feed for livestock (CFIA, 2024a). Many processing by-products and cull crops are currently approved, but as consumer trends change, some new by-products or crops with suitable nutrition profiles for animal feed may not be approved (Ominski et al., 2021). As previously stated, the largest contribution to food waste in Canada is from households, but the Canadian Federal Feeds Act and Regulations bans the use of plate-waste as livestock feed. This is a precaution to avoid the recurrence of Bovine Spongiform Encephalopathy (BSE) as well as reduce the risk of spreading zoonotic diseases (CFIA, 2024b). The European Union (EU) also banned this practice in livestock to avoid the transmission of BSE, foot and mouth disease and African swine fever (EC regulation No. 1774/2002). However, as of August 2022, the feeding of processed animal protein to non-ruminant livestock was allowed in the EU (EC regulation No. 999/2001). If other countries follow suit, feeding of plate-waste could replace 8.8 million tons of human-edible grain that are currently used globally in swine diets (zu Ermgassen et al., 2016). Although there is tremendous potential to utilize plate-waste as a livestock feed, there are several hurdles which must be overcome, including collection of household waste, the heterogeneous nature of the product, and risk of contaminants.

Furthermore, availability of several products including fruit and vegetables grown in Canada is seasonal, which may require frequent modification of livestock diets to ensure that nutrient requirements are met throughout the year (Halmemies-Beauchet-Filleau et al., 2018). Consistent streams of homogenous feedstuffs are desirable to ensure nutrient requirements are met with the least amount of effort. Froetschel et al. (2014) found that the nutrient profile of grocery by-product (GBP), composed of vegetables, fruit, and fresh bakery items varied monthly. Therefore, these authors recommended that each batch of GBP be analyzed individually to accurately quantify nutrient content to support diet formulation (Froetschel et al., 2014). Other products, such as bread, may be more consistent in nutrient profile and available throughout the year. Guiroy et al., (2000) found that a mixture of surplus bakery items and their by-products collected from the same source were consistent in their nutrient composition across lots.

The cost of including FLW in livestock diets is an important consideration for producers. Not only must the price of the FLW product be equal to or lower than conventional commodities but the cost and ease of coordination for shipping and storage must also meet or be superior to the standard feedstuff (Lardy et al., 2015). To ensure economical transport, surplus or damaged food must be within a reasonable distance to livestock or distributors (Lipinski et al., 2013). If the cost of transport is based on weight, products that contain higher moisture, such as fruit and vegetables, will cost more per unit of DM. Furthermore, wet products, such as pea cream, require different distribution and storage infrastructure than that required for dry products, which may require significant capital investment.

High moisture levels not only increase transportation costs but also increase the potential for spoilage. At distribution or grocery centers, sorting higher moisture items (fruits and vegetables) from drier items (breads) may reduce the spoilage of the drier items and increase their suitability as livestock feed. Unfortunately, this requires time and financial investment to train staff as well as additional infrastructure for storage. This may also discourage retailers from participating in redistribution if directing surplus food to landfill is less expensive and the framework is already in place (Nikkel et al., 2019).

In addition, fungi that thrive in high moisture environments can release toxins that negatively impact livestock. The presence of toxins can decrease feed intake, nutrient digestibility and absorption, negatively impacting animal performance (Xu et al., 2022). The potential presence of anti-nutritive factors must also be considered. For example, due to toxic alkaloids, green or sprouted potatoes are unsuitable as livestock feed (CFIA, 2024a). Some authors have suggested that fruit and vegetable wastes may potentially contain pesticide residues, heavy metals and dioxins (Wadhwa et al., 2015) which may negatively affect livestock and/or preclude their approval for inclusion in livestock diets. Furthermore, feeding these novel feedstuffs may present unique challenges with regards to processing and delivery. For example, potatoes and other tubers such as turnips and beets may cause cattle to choke, particularly if they are frozen. As a result, it is recommended that these feed sources be chopped or ensiled prior to feeding (Lardy et al., 2015).

High inclusion rates of some types of FLW in livestock diets can negatively affect the quality of the desired animal product. For example, inclusion of carrots in the diet at more than 20% DM caused discolouration of fat in lambs and cattle (Lardy et al., 2015). Further, inclusion rates of onions of more than 25% DM in dairy cow diets, produced off-flavoured milk (Lardy et al., 2015).

2.3 Environmental indices to quantify the environmental footprint of beef

Greenhouse gas emissions are an often-used environmental indicator to examine the impact of food waste mitigation strategies (Shurson, 2020). Additional environmental metrics that should be considered include land and water use as well as NH_3 emissions.

2.3.1 Greenhouse gas emissions

Traditionally, the term Global Warming Potential (GWP) has been used to examine the cumulative warming potential of different GHGs compared to CO_2 over a fixed period - usually 100 years (Environment and Climate Change Canada; ECCC, 2024). Carbon dioxide is a naturally occurring gas that is a part of the carbon cycle which has been significantly altered by human activities over the past 200 years (Falkowsi et al., 2000). It accounts for 26% of GHG emissions from the Canadian agriculture sector (Desjardins et al., 2020). In beef operations, CO_2 is produced from the burning of fossil fuels during the production of fertilizers, seeding and harvesting of feed crops, and the transportation of feed and livestock. Carbon dioxide is used as the baseline GHG and has a GWP of 1 (US EPA, 2024).

Methane accounts for 38% of GHG emissions from Canadian agriculture, most of which (87%) is produced by enteric fermentation, with the remainder generated from manure (Desjardins et al., 2020). Diverse strategies have been investigated to reduce enteric CH_4 such as genetic selection, vaccines, feed manipulation and dietary additives (Palangi et al., 2022). The GWP for CH_4 is estimated to be 27-30 over 100 years (US EPA 2024).

The remaining 36% of Canadian GHG emissions arises from nitrous oxide (N_2O), a potent GHG with a GWP 273 times that of CO_2 (Desjardins et al., 2020). These emissions occur as a result of soil management practices such as fertilizer application, biomass decomposition, cultivation and tillage as well as the handling, storing and spreading of manure (Fouli et al., 2021).

Measuring GHG emissions directly is a costly and time-consuming endeavor. Models offer an inexpensive and more efficient alternative to field experimentation (Bryant and Snow, 2008), enabling the agricultural industry to identify management strategies that may lead to reduced GHG emissions (Jose et al., 2016). For example, modeling can assess the impact of on-farm management strategies, including the use of food waste in diets on the carbon footprint of the beef industry. However, it is important to note that scenario-specific model coefficients which are garnered from field experimentation are required to obtain meaningful output data.

Holos (https://agriculture.canada.ca/en/agricultural-production/holos-software-program) is a whole-farm model software program that was developed by Agriculture and Agri-food Canada (AAFC) as a free resource to help producers identify strategies that reduce GHG emissions from their operations. Input data includes livestock species, feeding practices, tillage type and crop type. Holos estimates the type (CO_2, CH_4, and N_2O) and source of emissions (AAFC, 2024).

Other models have been used to estimate GWPs when employing various waste management strategies of FLW. Using a life cycle assessment (LCA), Eriksson et al. (2015) found that diverting banana, lettuce, and bread wastes to landfill had GWPs of 1.4, 0.21, and 1.9 kg CO_2e kg food waste, respectively compared to GWPs of -0.011, -0.013, and -0.61 kg CO_2e kg food waste when banana, lettuce, and bread wastes were used for animal feed, respectively.

2.3.2 Land and water use

As previously stated, livestock production requires substantial land and water resources. Close to 30% of the world's arable land and 32% of its freshwater are utilized to produce feed for livestock (Herrero et al., 2013). Land sparing effects have been reported when including FLW in the diets of livestock. Using LCA, bread waste that was valorized as animal feed, resulted in land sparing effects that were 40% higher than achieved if it was directed towards anaerobic digestion or incineration (Brancoli et al., 2020). Quintero-Herrara et al. (2021) demonstrated that inclusion of broccoli stems in the diet of dairy cattle at 11% diet DM reduced land use intensity by 0.002 m^2a kg^{-1} live weight (LW). Finally, feeding 7.1 kg fresh citrus waste hd^{-1} d^{-1} to 160 dairy cows in the US, was shown to have a yearly land sparing effect of 0.09 ha hd^{-1} (Baker et al., 2024).

According to the Water Footprint Framework, water use can be classified into 3 categories: i) green water (rainwater); ii) blue water (surface and groundwater); iii) grey water (wastewater; Mekonnen and Hoekstra, 2011). Most of the water associated with Canadian beef cattle production is consumed by feed crops (75%; Canadian Roundtable for Sustainable Beef, 2016). Clean drinking water is also a necessity but accounts for less than 1% of the water footprint of beef cattle (Legesse et al. 2018a).

The water use intensity of Canadian beef was recently estimated at 657 L kg^{-1} LW (Aboagye et al., 2024). In addition, Legesse et al. (2018a) demonstrated that water use intensity decreased from 557 to 459 L kg^{-1} boneless beef weight between 1981 and 2011. The observed decrease was attributed to improved production efficiency associated with increased crop yields and animal productivity, as well as increased use of crop by- or co-products in beef cattle diets (Legesse et al., 2017). Baker et al. (2024) demonstrated that 83,040 m^3 of water used for crops could be spared annually by feeding 7.1 kg hd^{-1} d^{-1} of fresh citrus waste to 160 dairy cattle. Additionally, FLW is often high in water content (Dou et al., 2018) and therefore including it in the diet of livestock can reduce drinking water requirements. Sheep fed a diet containing 10% DM of fruit or vegetable waste were observed to consume 22% and 14% less potable water, respectively (Sahoo et al., 2021).

2.3.3 Ammonia emissions

Ammonia (NH_3) is a colourless water-soluble gas with a potent odour and can have negative impacts on human health and the environment. It is highly reactive with atmospheric sulfuric and nitric acids, contributing to air pollution with the potential for serious impacts on human health (WHO, 2021). Excess NH_3 can be toxic to some vegetation and fish, altering the biodiversity in nutrient-poor ecosystems (AAFC, 2021). In other cases, NH_3 can lead to eutrophication, causing algae blooms and hypoxic waters, decreasing aquatic biodiversity (NOAA, 2024). Furthermore, although NH_3 is not a GHG, it readily oxidizes in soil, forming nitrate which through denitrification, can produce N_2O (Prosser et al., 2019).

Agricultural processes such as use of nitrogen fertilizer and manure management account for 93% of Canada's anthropogenic NH_3 emissions (AAFC, 2021). Emission factors can vary greatly depending on manure composition, collection and storage as well as environmental conditions (Hristov et al., 2011). Reduced excess dietary crude protein (CP) can mitigate these emissions from animal production systems by reducing nitrogen (N) excretion (Hristov et al., 2011). Legesse et al. (2018b) concluded that NH_3 emission intensity associated with Canadian beef (kg of NH_3 kg^{-1} of beef) was decreased by 20% from 1981 to 2011, due to increased reproductive efficiency, average daily gain, and carcass weights as well as dietary modifications.

To the author's knowledge there is no published literature regarding NH_3 emissions when feeding FLW to livestock. However, there is work underway investigating the effects of including cull potatoes at 15% and 30% diet DM on NH_3 emissions for cattle compared to conventional corn or barley diets (G. Mengistu, unpublished).

2.4 Current strategies and opportunities to reduce FLW

Nikkel et al. (2019) estimated that 32% of Canada's food waste is avoidable and could be redistributed for further use. New efforts have been and continue to be explored to combat FLW. Loop Resource, founded in British Columbia, has been diverting unsaleable food from groceries store to livestock operations (loopresource.ca, 2024) by working with grocery stores and livestock producers across Canada to divert grocery waste to livestock feed. To ensure waste meets CFIA regulations, which prohibit the feeding of raw meat to livestock, it is sorted by department and all partnered farms are required to follow CFIA guidelines.

2.4.1 Use of ensiling to increase the utilization of FLW

Currently most methods of utilizing FLW as animal feed are on an "as-is" basis and do not include preservation. Consequently, FLW must be used immediately due to the rapid spoilage of high moisture commodities such as fruits and vegetables (Bakshi et al., 2016). Although limited in scope at present, ensiling is a potential strategy to increase the shelf-life of FLW as most beef feedlots and dairies in Canada have the knowledge and the infrastructure necessary to ensile forages. In addition, 30-45% of Canadian cow-calf operations also feed silage as an over-winter-feeding method (BCRC, 2020).

Ensiling cull potatoes and/or peels is a method of preservation that is currently in use by some cattle producers. However, due to their high moisture content (80%), it is strongly recommended to ensile potatoes with dry forages or dried by-products (wheat middlings or beet pulp) to achieve optimal moisture levels for ensiling (Lardy et al., 2015). Ensiling carrot or pumpkin with crop sorghum has been explored in Australia (Forwood et al., 2019). These authors found that when vegetable matter was included at 20% DM, digestibility and total volatile fatty acid concentration of the silage were increased, producing high-quality alternative feed for ruminants.

Froetschel et al. (2014) observed that GBP could be readily ensiled, reaching and maintaining anaerobic stability in storage as demonstrated by a low pH and a sufficient water-soluble carbohydrate content, and the production of fermentation acids (lactic and acetic). These authors also observed that regardless of the variability in nutrient content, the total digestible nutrient (TDN) values attained were relatively consistent at approximately 85%, DM basis, which would allow producers to feed GBP silage as an energy source.

Dou et al. (2022) examined the potential to ensile discarded fruits and vegetables alone or with crop residues using mini silos. When fruit and vegetables were ensiled without the addition of crop residues or other forages, both ADF (13.6%) and NDF (15.9%) content were found to be lower than conventional forage silage (> 25%, > 40%, respectively). Ensuring adequate fibre in cattle diets is imperative to maintain rumen health through the prevention of metabolic disorders, such as acidosis. Fibre promotes rumen motility and saliva production, both of which help buffer against ruminal acids (Heinrichs and Kmicikewycz, 2023). When fruit and vegetables were ensiled with drier crop residues, the silage produced was comparable in ADF (20.4-40.4%) and

NDF (32.4-57.8%) content to conventional silage (Dou et al., 2022). These authors concluded that large scale ensiling practices for GBP need to be further explored.

2.5 Conclusion

To reduce the significant quantity of FLW generated in Canada, it is paramount that all players along the agri-food value chain contribute to the solution. Livestock should not be overlooked as a key player in this solution. By incorporating food waste into the diets of livestock, nutrients that would otherwise be lost are recycled, producing nutrient dense foodstuffs for a growing human population. Additional benefits include the potential to reduce the environmental impact of the sector including improved water and land use efficiency, as well as lower GHG and NH3 emissions. Further research is necessary to quantify the environmental benefits of including FLW as livestock feed. This information is essential for producers, processors and government to enhance decision-making and policy/program development regarding use of FLW.

3 HYPOTHESIS AND OBJECTIVES

3.1 Hypothesis

The use of bread by-products as an ingredient in the diet of cattle in a feedlot production system will result in a reduced environmental footprint compared to a standard corn-based diet. Environmental footprint criteria include i) land use, ii) greenhouse gas emissions [carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O)], iii) ammonia emissions (NH_3), and iv) water use.

3.2 Objectives

Evaluate and compare the environmental impact of feedlot cattle fed a standard corn-based diet or a bread by-product-based diet using existing data including diet composition and animal performance metrics (body weights, average daily gain, dry matter intake, feed efficiency) along with region-specific climate and crop yield data to assess:

i. Greenhouse gas emissions associated with crop and animal production.
ii. Land use associated with feed production for feedlot cattle diets.
iii. Ammonia emissions associated with feedlot housing, manure storage and land application.
iv. Water use associated with crop production, animal requirements, and beef processing.

4 MANUSCRIPT

ENVIRONMENTAL IMPACTS ASSOCIATED WITH FEEDING BREAD BY-PRODUCTS TO GROWING AND FINISHING FEEDLOT CATTLE

4.1 ABSTRACT

Environmental impacts of feedlot cattle when bread by-products (BBy) are substituted for corn in feedlot diets were evaluated and included land and water use, greenhouse gas (GHG) and ammonia (NH_3) emissions. Land, water, off-farm feed production and NH_3 emissions were estimated with the use of spreadsheet models, while Holos was used to model whole-farm GHG emissions. Data from two experiments using growing (Dvorak et al., 2001) and finishing (Guiroy et al., 2000) steers fed BBy at 40% and 55% of diet DM, respectively were used. Steers fed BBy had lower DMI than those offered the corn-based diet (9.26 vs. 10.08 kg d^{-1}, Dvorak et al., 2001; 10.17 vs. 11.39 kg d^{-1}, Guiroy et al., 2000), reducing the demand for corn grain, corn silage and hay. The lower DMI did not affect animal performance and resulted in improved feed:gain ratios of 6.17 vs. 7.04 and 6.64 vs. 7.22 for growing and finishing steers, respectively. Additionally, as BBy contained higher crude protein (15.9% DM, Dvorak et al., 2001; 16.0% DM, Guiroy et al., 2000) than corn (~10% DM), less soybean meal was required to meet protein requirements. In the growing experiment, inclusion of BBy resulted in the use of 45% less land and 37% less water, with GHG and NH_3 emissions reduced by 14 and 4%, respectively. In the finishing experiment, feeding BBy reduced land and water use by 63% and 61%, respectively while GHG emissions declined by an estimated 19% and NH_3 emissions were unaffected. Furthermore, GHG emissions associated with BBy from production to waste management were 24% and 53% lower when diverting BBy from landfill to growing and finishing feedlot diets, respectively. Consequently, inclusion of bread waste in feedlot diets is an effective strategy to reduce the environmental footprint of feedlot cattle. This research adds to the growing body of literature

that suggests that feeding food loss and waste to livestock improves the sustainability of food production systems.

4.2 INTRODUCTION

An increased global demand for animal products in middle- and low-income countries (World Bank, 2021) is expected to result in an increased demand for sustainably produced feed. As an exporting nation of many agricultural products including beef, Canada is well positioned to respond to the increased demand for animal products. However, food production growth may challenge land and water resources which may be further exacerbated by climate change (Hein, 2020). Simultaneously, one fifth of all food produced globally is wasted (UNEP, 2024b), leading to inefficient use of land and water resources while emitting large amounts of greenhouse gases (GHG) from landfills. Many food items, although no longer acceptable for human consumption, can be diverted from landfills and fed to livestock as an alternative to conventional feedstuffs.

Bread and bakery products account for 9% of the 35.5 million tonnes of food loss and waste (FLW) in Canada (NZWC, 2022; Nikkel et al., 2019). As bread waste is energy dense, it may be included in beef feedlot diets as a substitute for traditional energy feeds such as corn. A few studies have measured the growth performance of feedlot cattle fed bread or bakery products (Milton and Brandt, 1994; Guiroy et al., 2000; Dvorak et al., 2001), but research to quantify the environmental impact of this practice has not been conducted.

Feeding FLW to livestock is not a novel idea and many producers are including crop by-products such as oat hulls, canola meal and other forms of FLW in contemporary diets. However, defined inclusion rates that meet nutrient requirements so as to achieve growth performance targets are essential if FLW is to be widely included in livestock diets. Further, to the author’s knowledge, there are no other North American studies that have examined the environmental impact of including bread waste in growing or finishing diets of feedlot cattle. The purpose of this study was to explore the impact of including bread waste in the diets of growing and finishing feedlot steers on land and water use as well as GHG and NH3 emissions.

4.3 MATERIALS AND METHODS

4.3.1 Experimental design

Experimental site and climate characterization

The municipality of Hanover, Manitoba was selected as the site in which the model feedlot was located as bread waste has historically been included in the diets of feedlot cattle in this area. Site information such as hardiness zone, ecodistrict, and soil characteristics were categorized as 3B, Steinbach, and coarse dark gray chernozem, respectively (*Holos* 4.0.0.576). Long term (1993-2022) monthly average temperature and precipitation values (Table 4.1) were obtained from the Steinbach weather station (5MB0223) as provided by Agriculture and Agri-Food Canada (AAFC).

Table 4.1 Long-term average (1993- 2022) temperature and precipitation for Steinbach, MB (AAFC, 2023).

Month	Mean temperature (ºC)	Mean precipitation (mm)
January	-15.71	17.26
February	-13.89	13.90
March	-5.17	19.85
April	3.83	29.31
May	11.26	73.60
June	17.37	91.07
July	19.83	82.48
August	18.69	78.23
September	13.64	59.61
October	5.80	42.55
November	-3.62	24.21
December	-12.11	21.73

Animal management

Diet composition and animal performance metrics were obtained from two existing data sets that utilized bread by-products (BBy) as a substitute for corn in: i) growing steers (Dvorak et al., 2001) and ii) finishing steers (Guiroy et al., 2000).

Growing steers

Dvorak et al. (2001) examined the growth performance of 44 Angus-cross beef steers, stratified by weight, and randomly assigned (11 steers/pen) to a roughage-based grower diet (2 pens/diet) containing either corn or BBy (Table 4.2).

Table 4.2 Ingredient composition of growing diets with corn or bread by-product (BBy).

Ingredient composition, % DM	Corn diet	BBy diet
Dry rolled corn	39.5	3.0
Bread by-product	-	40.0
Alfalfa hay	30.0	30.0
Corn silage (40% grain)	20.0	20.0
Partially desugared beet molasses	4.0	4.0
Soybean meal (49% CP)	4.8	1.2
Feather meal	0.4	0.4
Blood meal	0.1	0.1
Salt	0.3	0.3
Limestone	0.24	-
Dicalcium phosphate	0.105	0.535
Vitamins and minerals	0.045	0.045
Monensin premix (Rumensin 80)	0.0125	0.0125
Finely ground corn	0.4995	0.4075

(Adapted from Dvorak et al., 2001)

Both diets contained alfalfa hay, corn silage, beet molasses, a mixed protein supplement (soybean meal; SBM, feather meal, blood meal), salt, vitamins and minerals (Table 4.2) as well as Monensin (27.5 mg/kg DM). The control diet contained 39.5% dry rolled corn whereas the experimental diet contained BBy at 40% DM. The BBy consisted mainly of crusts (70.2% DM) that arose from a crustless sandwich product (Table 4.3). The nutrient composition of lots of BBy remained fairly consistent with < 1 SD variance for DM, CP, TDN, ADF, calcium and phosphorus. However, the SD associated with crude fat and NDF was between 2 and 3. Growing steers were fed for 84-d (June-September) in confinement.

Table 4.3 Average nutrient composition of bread by-products from growing and finishing experiments.

	Bread by-products	
Nutrient, % DM	Growing[a]	Finishing[b]
Dry matter	70.2	67.6
Crude protein	15.9	16.0
Crude fat	4.9	3.3
TDN	87.0	86.5
ADF	1.2	1.9
NDF	7.2	3.0
Ca	0.09	0.10
P	0.15	0.20

[a] Data from Dvorak et al., 2001
[b] Data from Guiroy et al., 2000

Diets were formulated to meet the ruminal degradable and metabolizable protein requirements based on NRC Nutrient Requirements for Beef Cattle (1996) for growing cattle with an average initial weight of 302.07 kg. This resulted in CP of 13.8% and 14.1% DM for the corn and BBy diets, respectively.

Feed intake was recorded daily, and dry matter intake (DMI) was calculated and expressed in kg hd^{-1} d-1 (Table 4.4). Weight of each steer was measured on 3 consecutive days (d 82-84) and averaged to obtain a final weight. The average initial and final weights were used to calculate average daily gain (ADG) and feed efficiency (g feed kg^{-1} gain). It was assumed that after the 84-d feeding period, growing steers would require further growth to reach slaughter weight and therefore environmental metrics were estimated on a per head or live weight basis.

Table 4.4 Weight and feeding characteristics of growing feedlot steers fed a diet including corn or bread by-product (BBy).

Item	Diet			
	Corn		BBy	
n =	11	11	11	11
Avg initial weight	301.99	302.5	302.37	301.43
Avg final weight	420.19	424.85	431.02	424.96
DMI (kg hd^{-1} d^{-1})	10.09	10.08	9.36	9.15
ADG (kg hd^{-1} d^{-1})	1.41	1.46	1.53	1.47
Feed:gain ratio	7.17	6.92	6.11	6.22

(Adapted from Dvorak et al., 2001)

Finishing steers

Guiroy et al. (2000) used 120 15-month-old Angus crossbred feeder steers blocked into three weight classes: i) light weight (L, *n* =20; 340 kg); ii) medium weight (M, *n* =60; 364 kg), and iii) heavy weight (H, *n* =40; 385 kg). Half of the steers in each group were placed on a conventional corn-based finishing diet, while the other half were fed a BBy-based diet. Steers were implanted (Synovex-S, Fort Dodge Animal Health, Overland Park, KS) at the start of the experiment.

The conventional corn-based and BBy-based diets are described in Table 4.5. The BBy fed to finishing steers consisted of surplus and left-over bakery waste and other food processing plants (Table 4.3). Both diets included Monensin (33 mg kg^{-1} DM) and were formulated to meet NRC (1996) nutrient requirements for growing beef cattle with a target weight of 553 kg. Diets were isonitrogenous (12.7% CP) and met ruminal microbial N requirements for optimal microbial growth.

Table 4.5 Ingredient composition of corn-based and bread by-product (BBy)-based diets fed to finishing steers.

Ingredient composition, % DM	Corn-based diet	BBy-based diet
Dry whole shelled corn	63.3	15.3
Bread by-product	-	55.2
Corn silage (40% grain)	18.1	18.4
Timothy hay	9.0	6.2
Soybean meal (49% CP)	7.2	2.5
Limestone	1.2	1.2
Mineral mix	1.2	1.2
Vitamin premix	0.1	0.1
Rumensin premix, g kg^{-1} feed	0.033	0.033

(Adapted from Guiroy et al. 2000)

Guiroy et al. (2000) reported one DMI (kg hd^{-1} d^{-1}), ADG (kg hd^{-1} d^{-1}), and feed:gain ratio per diet regardless of initial weight class. Days on feed were estimated using the following equation:

$$DOF = (553\ kg - Initial\ weight) \div ADG \tag{4.1}$$

Where:

DOF = days on feed

553 kg = target finished weight

ADG = average daily gain associated with the diet fed (kg hd^{-1} d^{-1})

Table 4.6 Weight, feeding and carcass characteristics of light (L), medium (M), and heavy (H) weight finishing steers fed corn- or bread by-product (BBy)-based diets.

	Diet					
	Corn-based			BBy-based		
Performance metric	L	M	H	L	M	H
n =	10	30	20	10	30	20
Initial weight (kg)	340	364	385	340	364	385
DOF	134	119	106	138	123	109
DMI (kg hd^{-1} d^{-1})*		11.39*			10.17*	
ADG (kg hd^{-1} d^{-1})*		1.59*			1.54*	
Feed:gain ratio*		7.22*			6.64*	
Dressing percentage*		62.6*			62.4*	

(Adapted from Guiroy et al. 2000)

* One average value per diet based on a SAS (1998) analysis comparing weighted means

Hot carcass and boneless weights were determined using Equations 4.2 and 4.3.

$$HCW = DP \times slaughter\ weight \quad (4.2)$$

$$Boneless\ beef = HCW \times 0.71 \quad (4.3)$$

Where:

HCW = hot carcass weight

DP = dressing percentage, Table 4.6

Boneless beef = weight after trim and cutting into useable portions without bones, kg

0.71 = percentage of HCW converted to boneless beef (Mekonnen and Hoekstra, 2010)

Cropping system inputs and yields for growing and finishing experiments

Corn grain and silage as well as hay, were assumed to be grown on-farm with reduced-till management. Urea was incorporated or partially injected into the soil in the spring at the recommended rates (*Holos* 4.0.0.576) for corn grain and corn silage to generate targeted yields

(Table 4.7). No synthetic fertilizer was applied to hay. Soybeans as a source of SBM and wheat to produce BBy were assumed to be grown in the Hanover region, while the remaining feed ingredients were assumed to be purchased.

Table 4.7 Crop yields, fertilizer rates and loss percentages.

Feed crop	Yield (kg DM ha^{-1})	Fertilizer application rate (kg ha^{-1})	Storage loss (%)	Feeding loss (%)
Grain corn	6805.28	471.0	5	5
Silage corn	9817.13	394.3	10	5
Alfalfa hay	3709.05	-	15	5
Timothy hay	5315.00	-	15	5
Soybeans	2048.77	-	5	5
Spring Wheat	4593.69	-	-	-

It was assumed that the soybeans were transported to a local crushing plant with 1 tonne of soybeans producing 792 kg of SBM (USSEC, 2023). Soybean meal was then transported to the feedlot for inclusion in the diets.

Canada Western Red Spring was assumed to be the wheat class used to produce the BBy used in both the growing and finishing diets, as it is the predominant wheat class used for bread production in Canada (Cereals Canada, 2024). The processing of wheat and manufacturing of the bread was assumed to occur within 85 km of the location in which it was grown.

4.3.2 Calculating land use requirements

Total feed crop land requirements (ha) were calculated per diet using DMI (kg hd^{-1} d^{-1}), ingredient inclusion rate (% DM), number of animals, DOF, and yield (kg DM ha^{-1}) for each crop type (Equation 4.4). Field losses were included in yield estimates, with storage and feeding losses for each crop obtained from the literature (Ball et al.,1998; Kertz, 1998; Manitoba Agriculture, 2024; Schwab, 2021; Table 4.7). Five-year average (2017-2022) crop yields for alfalfa hay, corn grain, corn silage, soybeans, and red spring wheat were obtained from Manitoba Agricultural Services Corporation (https://www.masc.mb.ca) for the Hanover region. As wheat was not grown for feed, land needed to grow it was not attributed to steers fed BBy, nor were storage or feeding losses considered. Timothy hay yield was obtained from Seed Manitoba (2013).

$$Crop\ land\ required = \frac{DMI * P * n * DOF * (1 + l)}{yield} \tag{4.4}$$

Where:

DMI = dry matter intake (kg hd^{-1} d^{-1})

P = proportion of ingredient in diet (%)

n = number of steers in treatment group

DOF = number of days steers in treatment were on feed

l = storage and feeding loss (%)

Yield = kg DM ha^{-1}

4.3.3 Estimating greenhouse gas emissions

Holos

The *Holos* model (4.0.0.576; 2024) was used to estimate GHG emissions and soil carbon using IPCC Tier II (2019) emissions factors specific to Canadian conditions. This model uses Canadian weather and soil data to estimate emissions and soil carbon. *Holos* was used to estimate GHG emissions from feed crop production as well as those associated with the feeding of growing (Dvorak et al., 2001) and finishing steers (Guiroy et al., 2000).

Greenhouse gases included in net emissions were i) methane (CH_4) from enteric fermentation and manure management; ii) nitrous oxide (N_2O) from nitrogen (N) inputs, crop residues and animal manure; and iii) carbon dioxide (CO_2) from fossil fuel consumption, electricity use, and production of fertilizer.

Enteric CH_4 is a product of rumen fermentation and was estimated using gross energy intake of the feed consumed (MJ hd^{-1} d^{-1}), a diet-specific CH_4 conversion factor, the energy content of CH4 (55.65 MJ kg^{-1}) and a reduction factor associated with the inclusion of an ionophore in the diet (Alemu and Pogue, 2024). Methane emissions derived from solid manure (deep bedding) was calculated based on the amount of volatile solids (VS) excreted (kg hd^{-1} d^{-1}), CH4 producing capacity of beef cattle (m^3 CH_4 kg^{-1} VS) and a CH_4 conversion factor based on beef cattle in a deep bedded system (Alemu et al. 2024a).

Direct N_2O emissions were estimated from nitrification - denitrification processes, crop N inputs, crop residues, and manure. The calculated emission factor for N_2O considered topography, precipitation, and soil texture of the region in a reduced tillage system. Manure produced on farm was assumed to be applied to the land the following year, with emissions proportionately divided among crop fields within same calendar year. The emission rate of N_2O from manure considered total excreted N (kg hd^{-1} d^{-1}) and an emission factor specific to beef in a deep bedding manure system (Alemu et al. 2024b). Indirect N_2O emissions associated with leaching, runoff, and volatilization of fertilizer and manure were also estimated with consideration given to precipitation, potential evapotranspiration, and temperature.

Carbon dioxide emissions were calculated by considering fossil fuel consumption based on regional tillage, soil and crop types. A conversion factor of GJ of diesel to kg CO_2 (70 kg GJ^{-1}) was used with consideration for area of land cultivated. As cattle were housed in feedlots (confined, no barn) on a deep bedding manure system, CO_2 emissions depended on the number of days on feed, number of animals, annual electricity required per head (65.7 kWh hd^{-1}), and the Manitoba-specific electricity conversion to CO2 (1.0×10^{-3} kg CO2 kWh^{-1}). Carbon dioxide emissions from spreading solid manure considered the volume of manure, a conversion factor (0.0248 GJ 1000 L^{-1}) and the diesel conversion factor for the equipment used for spreading.

Mass allocation

An *Excel*-based model was used to determine GHG emissions (kg CO_2e kg^{-1} DM) and mass allocation (Mengistu et al., 2022) from imported SBM and BBy. Manitoba-specific carbon footprints were obtained from Desjardins et al. (2020) for soybeans (0.08 kg CO_2e kg^{-1} DM) and spring wheat (0.37 kg CO_2e kg^{-1} DM) production. Emissions from processing soybeans to SBM (0.016 kg CO_2e kg^{-1} DM; McGeough et al., 2012), wheat to flour (0.16 kg CO_2e kg^{-1} DM) and flour to bread (0.03 kg CO_2e kg^{-1} DM) were obtained from Espinoza-Orias et al. (2011). As the primary product derived from processed soybeans is oil for human consumption (AAFC, 2006), only the proportion allotted to SBM (79.2%; USSEC, 2023) was considered in GHG emission estimates. Similarly, as the bread was not produced for feed, GHG emissions from BBy were not attributed to steers.

Transport was assumed to be by heavy-duty diesel trucks with average emission factors (g L^{-1} fuel) for CO_2, CH_4, and N_2O (ECCC, 2020a) converted to CO_2e to generate a single emission factor. Average truck fuel consumption of 2.48 km L^{-1} was obtained from Holtshausen et al. (2021) and load capacity was assumed to be 43,000 kg (https://agri-trans.ca/equipment). Transport emissions from SBM were calculated based on the distance from field to processing plant (30 km), and from the processing plant to the feedlot (33 km). Spring wheat transport emissions were calculated based on the distance from field to the mill (69 km), the mill to the bakery (84 km), and the bakery to feedlot (5 km).

It was assumed that if BBy was not used as cattle feed, it would be disposed of in the landfill nearest to the bakery (4 km) using a transport truck in a single trip. Landfill emissions for food waste were used as proxy for BBy obtained from Lee et al. (2017) with an emission factor of 2.6 kg CO_2e kg^{-1} dry food waste.

4.3.4 Estimating ammonia emissions

Nitrogen intake and excretion

Ammonia emissions from cattle were based on total ammoniacal N (TAN) derived from urine and feces as well as manure storage and land application as described by Legesse et al. (2018b). Estimated daily excreted N ($N_{\text{excretion rate}}$) was calculated by estimating the difference between daily N intake and daily N retention. Daily N intake (IPCC 2019; Equation 4.5) was estimated per head using gross energy (GE) intake of the steer, proportion of CP in the diet and net energy requirements for maintenance ($NE_{\text{maintenance}}$).

$$N\ intake = \frac{GE}{18.45} \times \frac{CP\%}{100} \div 6.25$$

(IPCC, 2019; 4.5)

Where:

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

GE = gross energy intake, MJ hd^{-1} d^{-1}

18.45 = conversion factor for dietary GE per kg of DM, MJ kg^{-1}

CP% = percent of crude protein in diet

6.25 = conversion from kg of CP to N, $(kg\ N)^{-1}$

$$GE = \left[\left(\frac{NE_m}{REM}\right) + \left(\frac{NE_g}{REG}\right)\right] \div \frac{DE}{100}$$

(IPCC, 2019; 4.6)

Where:

NE_m = net energy required by steer for maintenance, MJ d^{-1}

REM = ratio of net energy available in diet for maintenance to digestible energy consumed

NE_g = net energy required for growth, MJ d^{-1}

REG = ratio of net energy available for growth in diet to digestible energy consumed

DE = digestible energy expressed as a percentage of gross energy, % TDN

$$NE_m = Cf_{ia} \times BW^{0.75}$$

(IPCC, 2006; 4.7)

Where:

Cf_{ia} = animal category coefficient adjusted for temperature, MJ d^{-1} kg^{-1}

BW = average live weight of steer during the feeding period, kg

$$REM = \left[1.123 - (4.092 \times 10^{-3} \times DE) + (1.126 \times 10^{-5} \times (DE)^2) - \left(\frac{25.4}{DE}\right)\right]$$

(IPCC, 2006; 4.8)

$$NE_g = 22.02 \times \left(\frac{BW}{C_d \times FW}\right)^{0.75} \times ADG^{1.097}$$

(IPCC, 2006; 4.9)

Where:

C_d = animal description coefficient with a value of 1.0 for steers

FW = final live body weight of the steer, kg

ADG = average daily weight gain of the steer, kg

$$REG = 1.164 - (5.160 \times 10^{-3} \times DE) + (1.308 \times 10^{-5} \times DE^2) - \left(\frac{37.4}{DE}\right)$$

(IPCC, 2006; 4.10)

Daily N retention was estimated using Equation 4.11.

$$N_{retention} = \frac{ADG \times \left[268 - \left(\frac{7.03 \times NE_g}{ADG}\right)\right]}{1000 \times 6.25}$$

(IPCC, 2019; 4.11)

Where:

$N_{retention}$ is expressed in kg N animal^{-1} d^{-1}

268 = constant derived from Equation 3-8 NRC (1996), g protein kg^{-1} hd^{-1}

7.03 = constant derived from Equation 3-8 NRC (1996), g protein MJ^{-1} hd^{-1}

6.25 = conversion from kg dietary protein to kg dietary N, kg protein (kg N)$^{-1}$

Nitrogen excreted through urine ($TAN_{excreted}$) was estimated using Equation 4.12, while N excreted in fecal matter ($FecalN_{excreted}$) was assumed to account for the remainder (Equation 4.13).

$$TAN_{excreted} = N_{excreted} \times Fraction_{Urinary-N}$$

(Dämmgen and Hutchings, 2008; 4.12)

Where:

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

$Fraction_{Urinary-N}$ = fraction of N excreted in urine, 0.57 as CP in diet was between 9 and 15%, DM (Chai et al., 2014).

$$FecalN_{excreted} = N_{excreted} * \left(1 - Fraction_{Urinary-N}\right)$$

(Dämmgen and Hutchings, 2008; 4.13)

Where:

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

Quantification of NH3 emissions

Animal housing

Ammonia emissions associated with growing and finishing steers confined in feedlots were estimated using Equation 4.14.

$$NH_{3feedlot} = TAN_{excreted} \times EF_{feedlot\ adj} \times \frac{17}{14}$$

(Legesse et al., 2018b; 4.14)

Where:

$NH_{3feedlot}$ = NH_3 emissions from feedlot confinement, kg NH_3 hd^{-1} d^{-1}

$EF_{feedlot\ adj}$ = temperature-adjusted emission factor for feedlot confinement

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

$$EF_{feedlot\ adj} = T_{adj} \times EF_{feedlot}$$

(Legesse et al., 2018b; 4.15)

Where:

T_{adj} = the adjusted temperature for animals confined in feedlots, kg (NH_3-N) kg^{-1}

$EF_{feedlot}$ = 0.9 kg (NH_3-N) kg^{-1}

$$T_{adj} = \frac{1.041^{T}}{1.041^{17.7}}$$

(Legesse et al., 2018b; 4.16)

Where:

T = average temperature of the feeding period, °C

$TAN_{excreted}$ (Mg TAN hd^{-1} $period^{-1}$) and NH_3 emissions (Mg NH_3 hd^{-1} $period^{-1}$) were multiplied by the number of days in each feeding period to estimate periodic $TAN_{excreted}$ ($PTAN_{excreted}$) and periodic NH_3 ($PNH_{3feedlot}$), respectively.

Manure storage

Manure was assumed to be stockpiled as it is common practice in confined feedlots (Legesse et al., 2018b). Manure storage was assumed to begin 2 months after the start of the feeding trials as described by Sheppard and Bittman (2012). Total NH_3-N mass movement from feedlot to storage was estimated by calculating the difference between $PTAN_{excreted}$ and $PNH_{3feedlot}$ as seen in Equation 4.17.

$$PTAN_{storage} = PTAN_{excreted} - PNH_{3feedlot} \times \frac{14}{17}$$

(Legesse et al., 2018b; 4.17)

Where:

$PTAN_{storage}$ = TAN in manure that is moved to storage; Mg TAN hd^{-1} $period^{-1}$

$PTAN_{excreted}$ = TAN in manure excreted by feedlot cattle, Mg TAN hd^{-1} $period^{-1}$

$PNH_{3feedlot}$ = temperature-corrected periodic NH_3 emissions from feedlot cattle, Mg NH_3 hd^{-1} $period^{-1}$

14/17 = conversion of NH_3 to NH_3-N

Ammonia volatilization from stored manure ($NH_{3emissions}$) was estimated by considering TAN in manure during storage and a temperature adjusted stockpiling emission factor. Variation in TAN during manure storage is defined in Equation 4.18.

$$PTAN_{storage2} = PTAN_{storage} \times \left(1 - F_{nitrify}\right) + (PON_{storage} \times F_{mineralize})$$

(Legesse et al., 2018b; 4.18)

Where:

$PTAN_{storage2}$ = adjusted TAN in manure storage, Mg TAN hd^{-1} $period^{-1}$

$F_{nitrify}$= fraction of TAN nitrified during storage, 0.14 (Chai et al., 2014)

$PON_{storage}$ = periodic organic N from fecal matter in storage, Mg organic N hd^{-1} $period^{-1}$

$F_{mineralize}$ = fraction of organic N mineralized as TAN during storage, 0.28 (Chai et al., 2014)

Manure land application

Manure was assumed to be spread in April of the following year in accordance with the Livestock Manure and Mortalities Management Regulation (Government of Manitoba, 1998). As the modeled farms were assumed to use no/low-till management, all manure was assumed to be spread on untilled land. The volatilization of NH_3 that occurs during application was calculated by multiplying TAN with an untilled land specific EF (0.79) which was adjusted for temperature of the application period. The periodic TAN moved from storage to land was estimated using Equation 4.19.

$$PTAN_{land} = \left(PTAN_{storage2} - PNH_{3emissions} \times \frac{14}{17}\right)$$

(Legesse et al., 2018b; 4.19)

Where:

$PTAN_{land}$ = periodic TAN applied to untilled land during the application period per animal, Mg TAN hd^{-1} $period^{-1}$

$PNH_{3emissions}$ = periodic $NH_{3emission,s}$ from stored manure, Mg NH_3 hd^{-1} $period^{-1}$

4.3.5 Estimating water use

Crop production

As irrigation is uncommon in Manitoba (Statistics Canada, 2023), moisture retained in the soil as a result of previous precipitation in addition to the precipitation during the growing season was assumed to meet crop water requirements. Therefore, only green water was included in calculations. Green water use associated with feed crop production were calculated based on potential evapotranspiration (PET) or crop-specific actual evapotranspiration (AET) for the region, proportions of feed in each diet, and crop yield as described by Legesse et al. (2018a).

Evapotranspiration (ET) data was obtained from AAFC's Versatile Soil Moisture Budget model using data from the Steinbach weather station (5MB0223). Potential evapotranspiration was used to estimate water use for Timothy and alfalfa hays since soil moisture was not a limiting factor. Crop-specific AET data was used to estimate water use for corn, soybeans, and wheat. Potential evapotranspiration values were estimated using the Priestley-Taylor method (1972) which calculates ET of a reference crop (grass or alfalfa) when crop is at full cover and soil water is not a limiting factor (Al-Kaisi, 2000), while AET was calculated using the modelled phenology stage and K_c of the crop (Y. Zhang, AAFC, personal communication).

The growth stage length for crops were obtained from the literature and were based on growing region and altered to accommodate harvest (Allen, 2002; Legesse et al., 2018a). The four growth stages included initial, development, mid-season, and late-season. Both Timothy and alfalfa were assumed to start spring growth April 25th (D. Cattani, personal communication). May 1st was assumed as the planting date for optimal crop yield for remaining crops according to a nine-year average (MASC, 2019). Only the proportion of soybeans associated with SBM (79.2%) was included in water use estimates.

Wheat production, milling and bread manufacturing

As the bread in this study was not produced for feed, water requirements for wheat were not attributed to steers fed the BBy. Water utilized for the milling and baking of wheat flour is negligible, accounting for less than 1% of the total water footprint of bread (Matohlang Mohlotsane et al., 2018) and therefore was excluded.

Animal water consumption and processing

Total water consumption per animal was estimated using the daily water requirement of feedlot steers as described by Hicks et al. (1988; Equation 4.20), multiplied by the number of days in the feeding period.

$$DWI = -6.0716 + (0.70866 \times MT) + (2.432 \times DMI) - (3.87 \times PP) - (4.437 \times DS)$$

(Hicks et al. 1988; 4.20)

Where:

DWI = daily water intake, L d^{-1}

MT = average maximum temperature during the feeding period, °C

DMI = dry matter intake, kg d^{-1}

PP = average daily precipitation during the feeding period, cm

DS = dietary salt, % DM

Water used for cleaning and maintaining feedlots was deemed negligible (Beaulieu, 2007; Legesse et al., 2018a) and was not included in total water use. A value of 16.5 L kg^{-1}

boneless beef produced was assumed for processing water requirements, as per Legesse et al. (2018a).

4.4 RESULTS

4.4.1 Impact of bread waste inclusion in feedlot diets on land use requirements

Land requirements for growing steers fed the BBy diet (GBBy) were 0.10 ha hd^{-1} compared to 0.19 ha hd^{-1} for growing steers fed the corn diet (GC), resulting in a 45% decrease in land use (Figure 4.1). The most notable reductions in land requirements with GBBy were observed for corn grain (93%) and soybeans (76%), while corn silage and alfalfa hay were only 9% and 8% lower, respectively, as compared to GC. Land requirements for finishing steers were 0.09, 0.10, and 0.11 ha hd^{-1} for H steers fed the BBy-based diet (HBBy), M steers fed the BBy-based diet (MBBy), and L steers fed the BBy-based diet (LBBy), respectively. This is compared to 0.24, 0.27, and 0.30 ha hd^{-1} for L steers fed the corn-based diet (LC), M steers fed the corn-based diet (MC), and H steers fed the corn-based diet (HC), respectively. Land requirements across all three treatment groups fed the BBy-based diet (FBBy) were 63% lower as compared to those fed the corn-based diet (FC; Figure 4.2). When comparing FBBy to FC, reductions in land requirements for corn grain and soybeans were 78% and 68%, respectively. Land requirements for Timothy hay and corn silage were 36% and 6% lower with FBBy as compared to FC. In both corn and BBy-based diets, the ranking of land requirements was H < M < L steers.

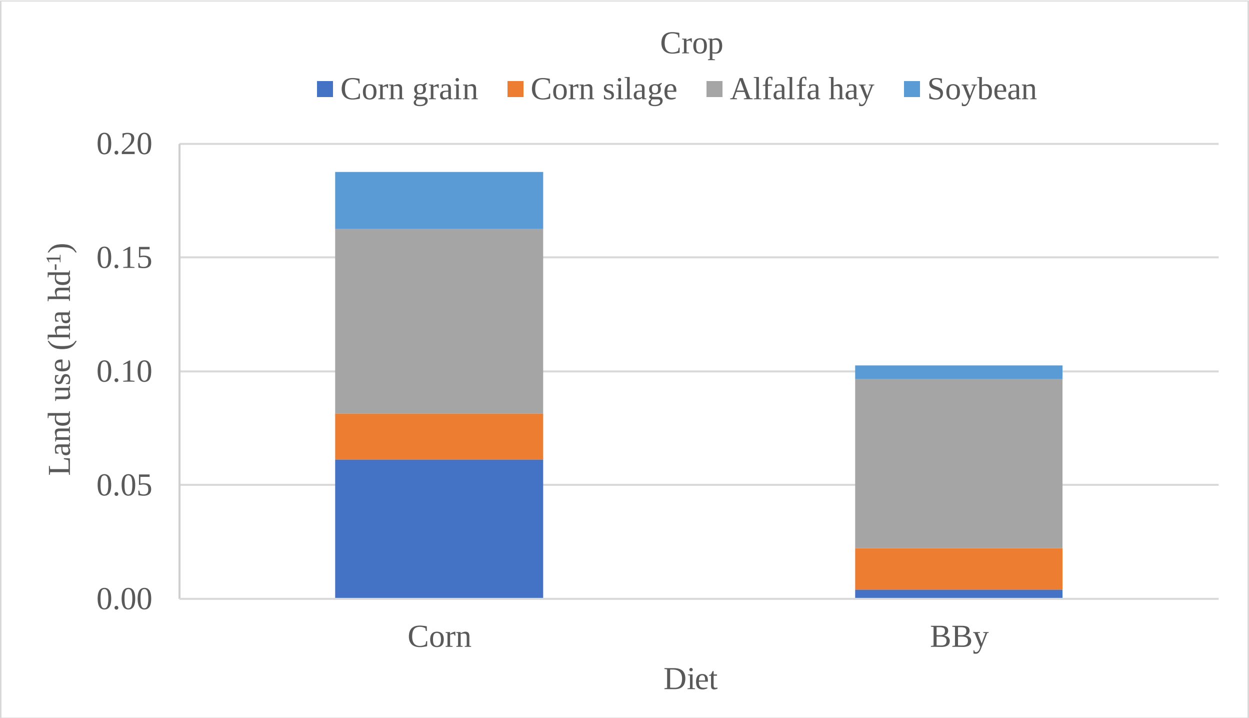

Figure 4.1 Land required for production of feed in corn or bread by-product (BBy) diets fed to growing steers.

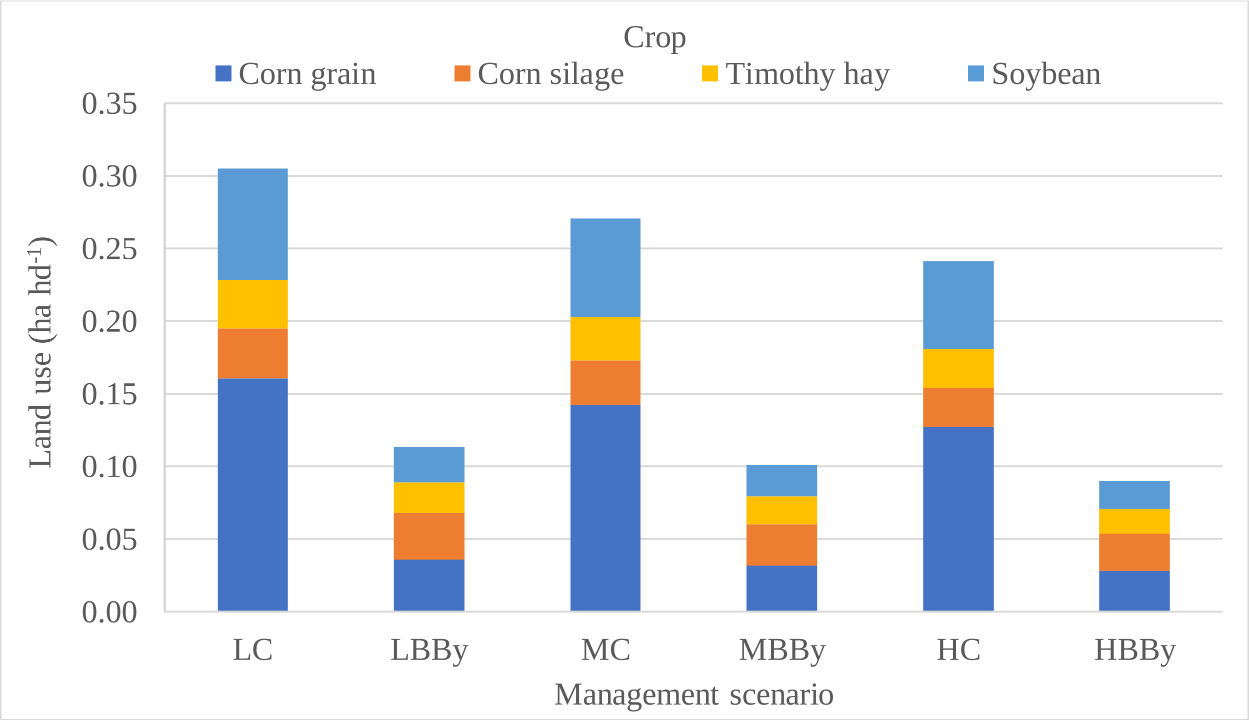

Figure 4.2 Land required for production of feed for the three weight groups (L = Light, M =Medium, H = Heavy) of steers fed corn (C) or bread by-product (BBy) finishing diets.

4.4.2 Impact of bread waste inclusion in feedlot diets on GHG emissions

On-farm emissions

Total on-farm GHG emissions associated with GBBy were 14% lower (696 kg CO_2e hd^{-1}) than GC (805 kg CO_2e hd^{-1}; Figure 4.3). The reduction in emissions was primarily associated with crop production as a result of a 75% reduction in upstream CO_2 and 61% reduction in farm energy CO_2. Direct and indirect N_2O were also reduced by 28% and 27%, respectively. Minimal differences of less than 2% were observed in enteric and manure CH_4 emissions between GBBy vs GC.

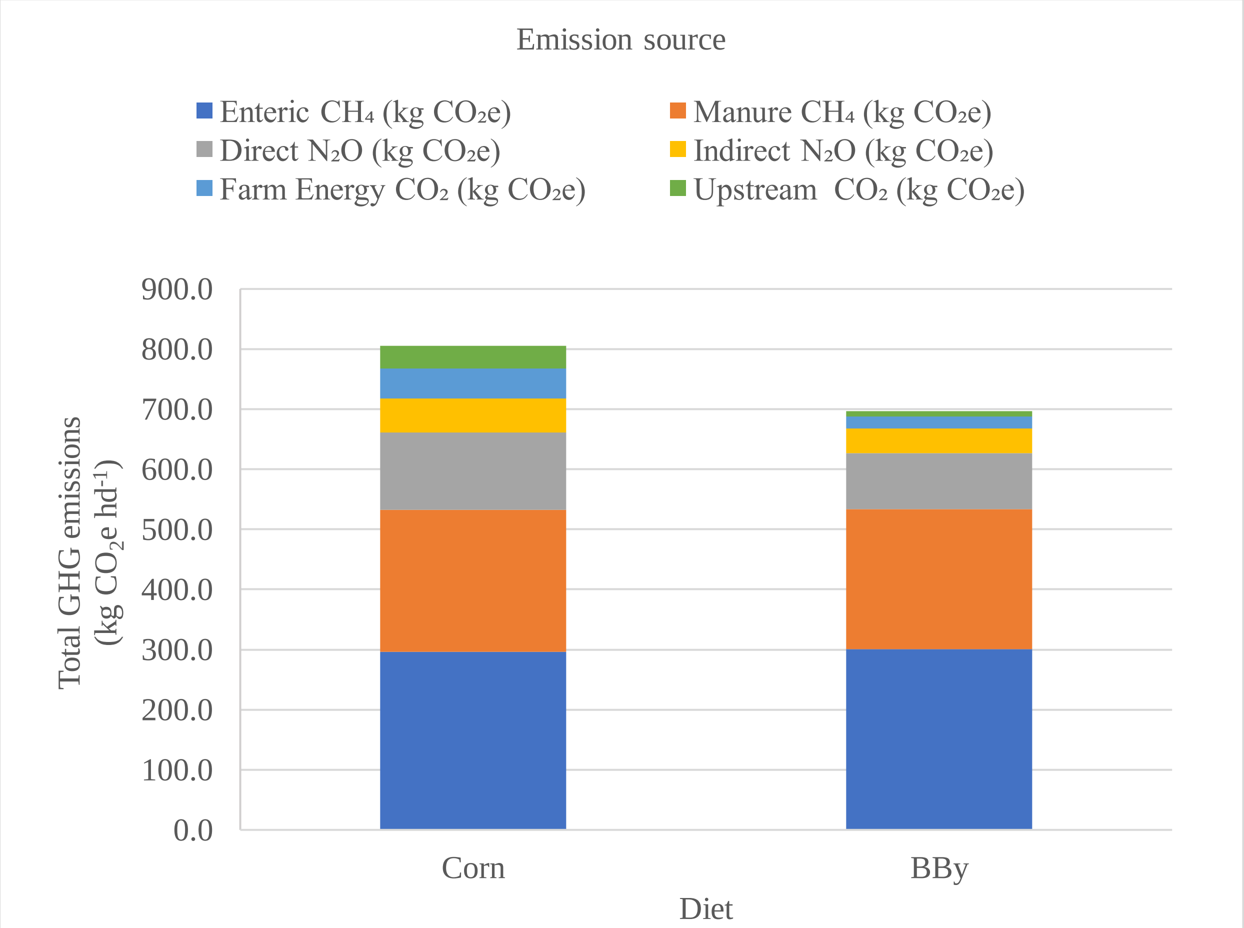

Figure 4.3 Total on-farm greenhouse gas (GHG) emissions by source for growing steers fed corn or bread by-product (BBy) diets.

The GHG emissions associated with LBBy, MBBy, and HBBy were 1051, 947, and 846 kg CO_2e hd^{-1} respectively, compared to 1319, 1173, and 1053 kg CO_2e hd^{-1} for LC, MC, and HC, respectively (Figure 4.4). Consequently, GHG emissions were 20% lower for FBBy than FC. Reduced on-farm emissions were primarily associated with crop production with a 67% and 64% reduction in upstream and farm energy CO_2, respectively, for FBBy compared to FC. Furthermore, direct and indirect N_2O emissions were reduced by 34% for FBBy as compared to FC. Only small reductions were observed in enteric (< 1%) and manure CH_4 (5%) emissions from FBBy as compared to FC. In both corn and BBy-based diets, GHG emissions ranked H < M < L.

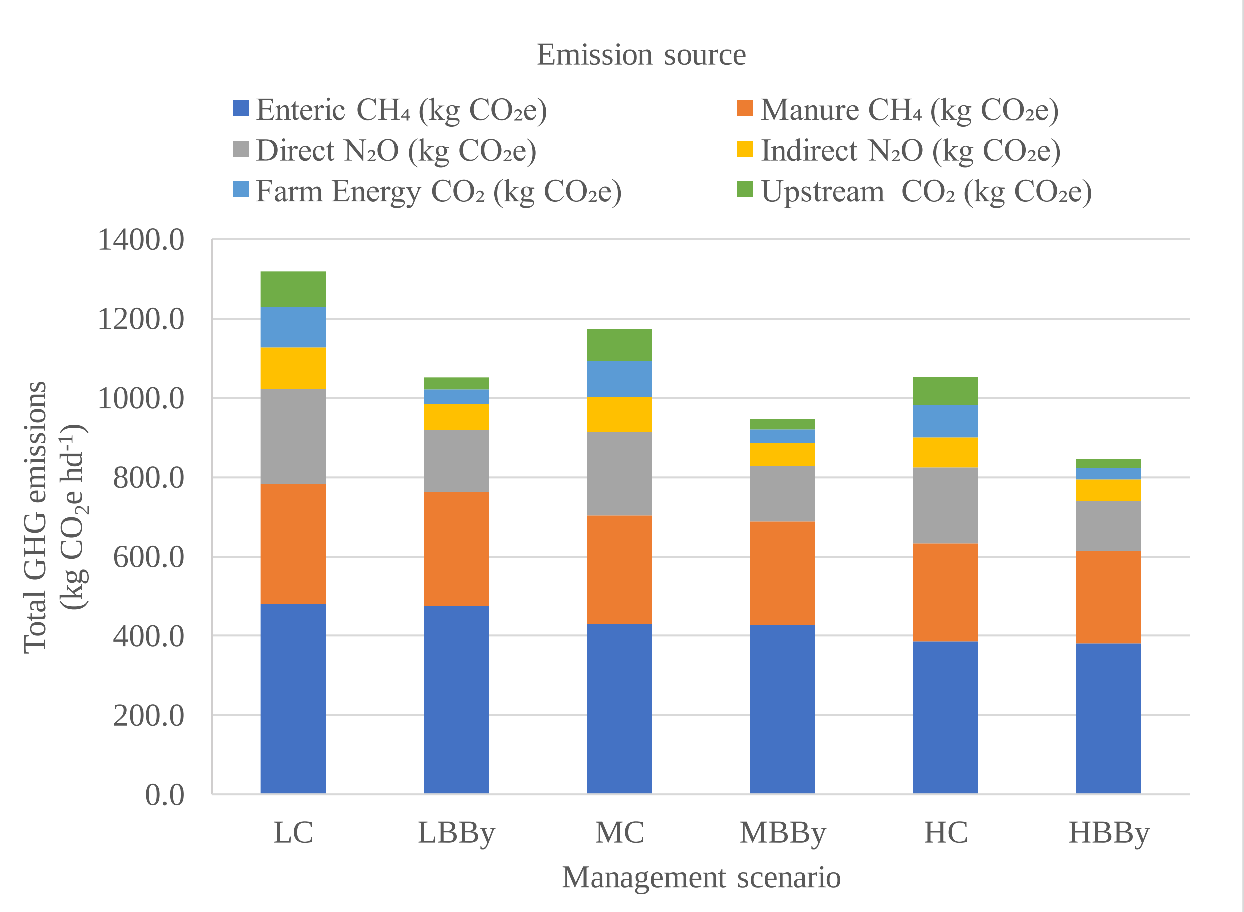

Figure 4.4 Total on-farm greenhouse gas (GHG) emissions by source for three weight classes (L = Light, M =Medium, H = Heavy) of steers fed corn (C) or bread by-product (BBy) finishing diets.

The majority of on-farm GHG emissions were derived from processes associated with steers, accounting for 78%, 91%, 70%, and 86% of total GHG emissions for GC, GBBy, FC, and FBBy, respectively (Figure 4.5). Further, both GC and FC had a higher proportion of GHG emissions associated with the production of feed crops (22% and 30%, respectively) as compared to GBBy and FBBy (9% and 14%, respectively).

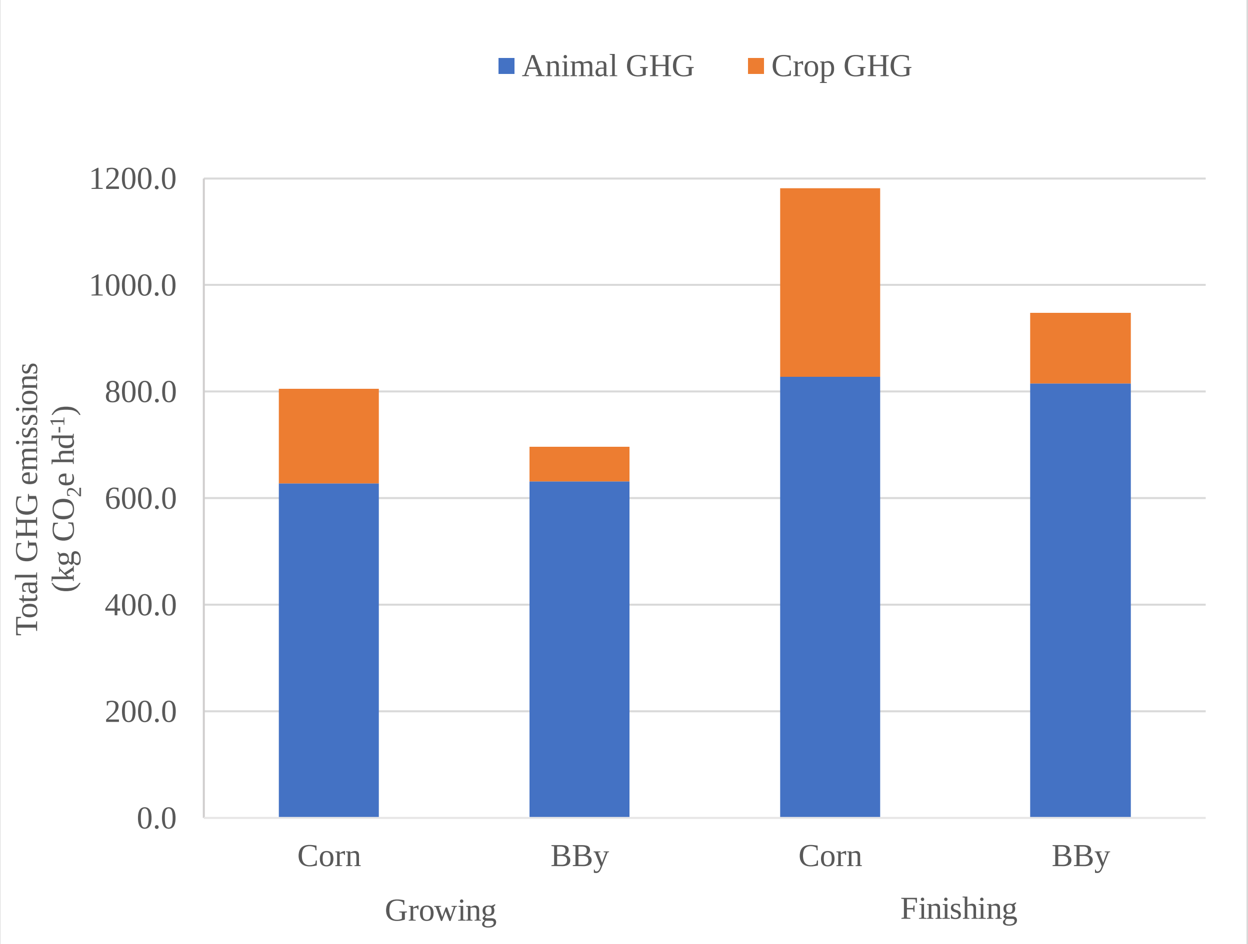

Figure 4.5 Total greenhouse gas (GHG) emissions associated with animal or crop production processes for growing and finishing beef steers offered corn or bread by-product (BBy) diets.

Off-farm emissions

Greenhouse gas emissions associated with SBM in growing diets were 77% lower for GBBy (0.96 kg CO_2e hd^{-1}) as compared to GC (4.20 kg CO_2e hd^{-1}). In finishing diets, SBM emissions were 68% lower for FBBy (L=3.8; M=3.4; H=3.0 kg CO_2e hd^{-1}) compared to FC

(L=11.9; M=10.6; H=9.4 kg CO_2e hd^{-1}). Emissions from the production of soybeans accounted for 84% of the total emissions from SBM production while processing accounted for 15% and transport for < 2% (Figure 4.6).

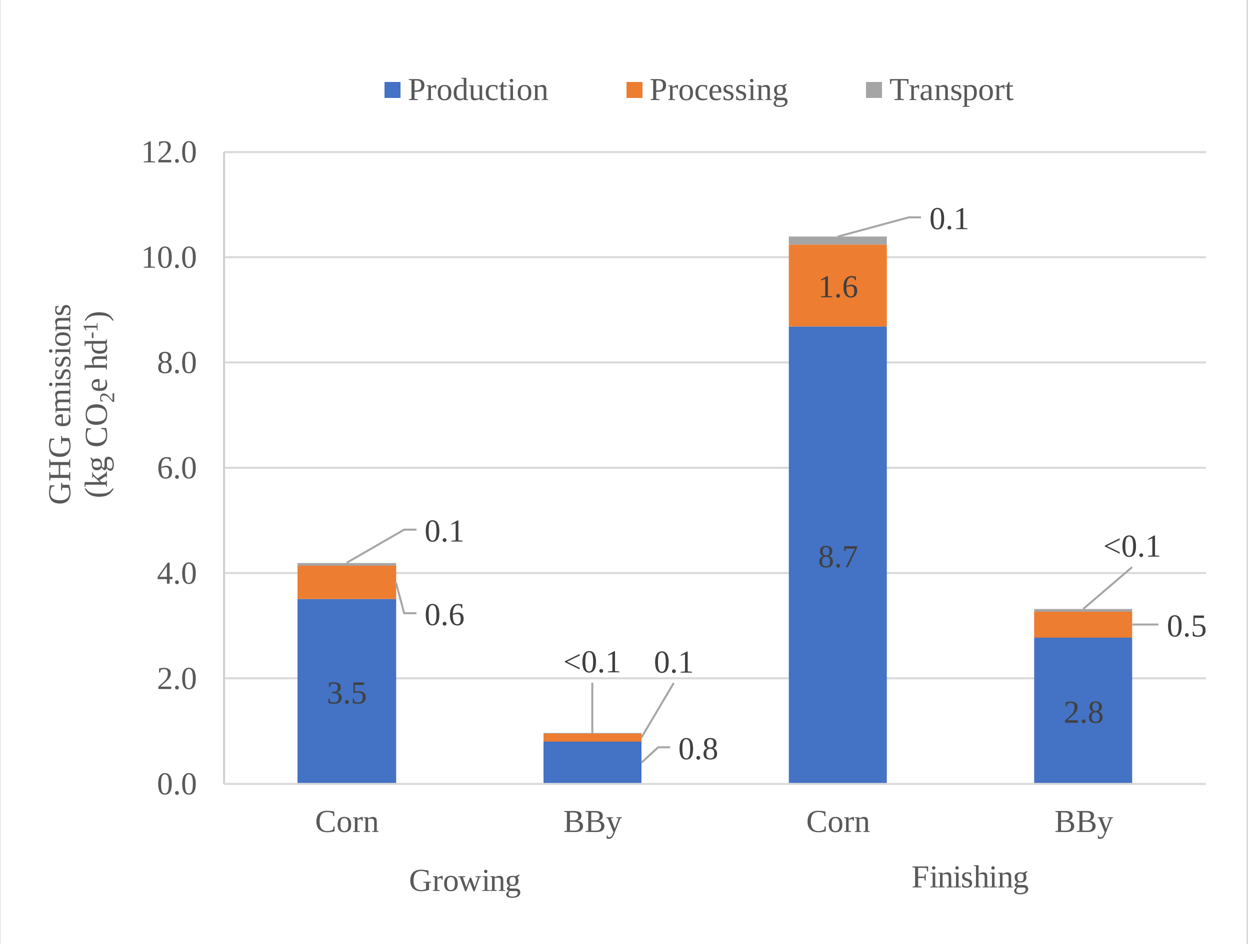

Figure 4.6 Average greenhouse gas (GHG) emissions by source associated with soybean meal production for inclusion in diets of growing and finishing steers.

Greenhouse gas emissions associated with production and waste management of BBy were estimated to be 24% less when BBy was redirected from landfill to feed growing steers (kg CO_2e; Figure 4.7).

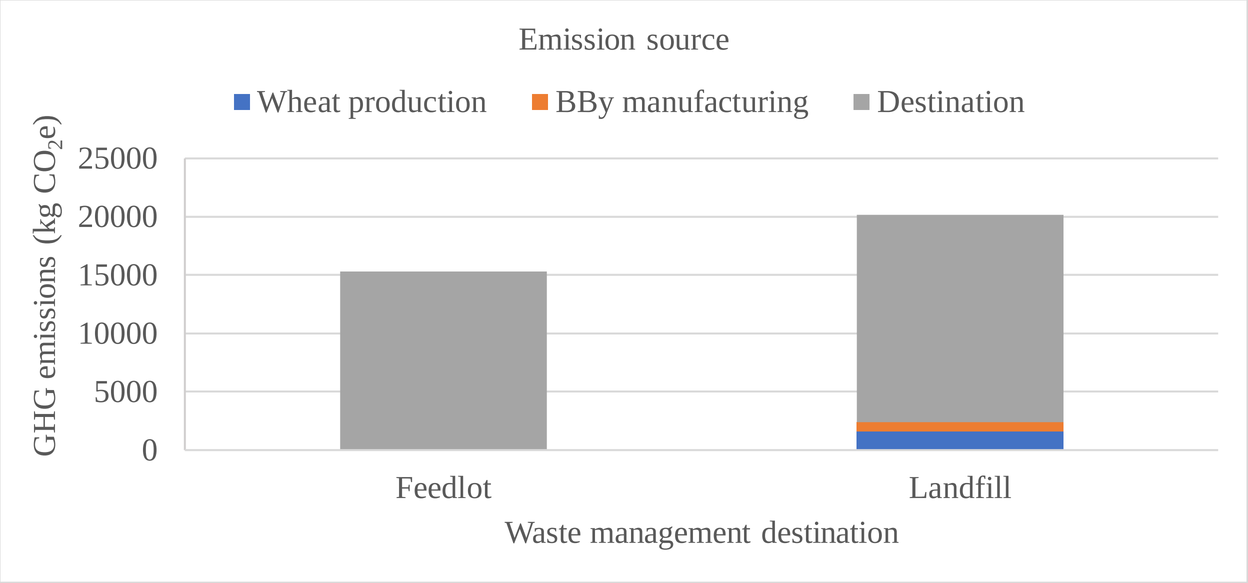

Figure 4.7 Greenhouse gas emissions (GHG) associated with bread by-product (BBy) that is fed to growing steers in a feedlot or landfilled as waste management strategies (not including transportation).

Greenhouse gas emissions associated with production and waste management of BBy were estimated to be 53% less when BBy was redirected from landfill to feed finishing steers (kg CO_2e; Figure 4.8).

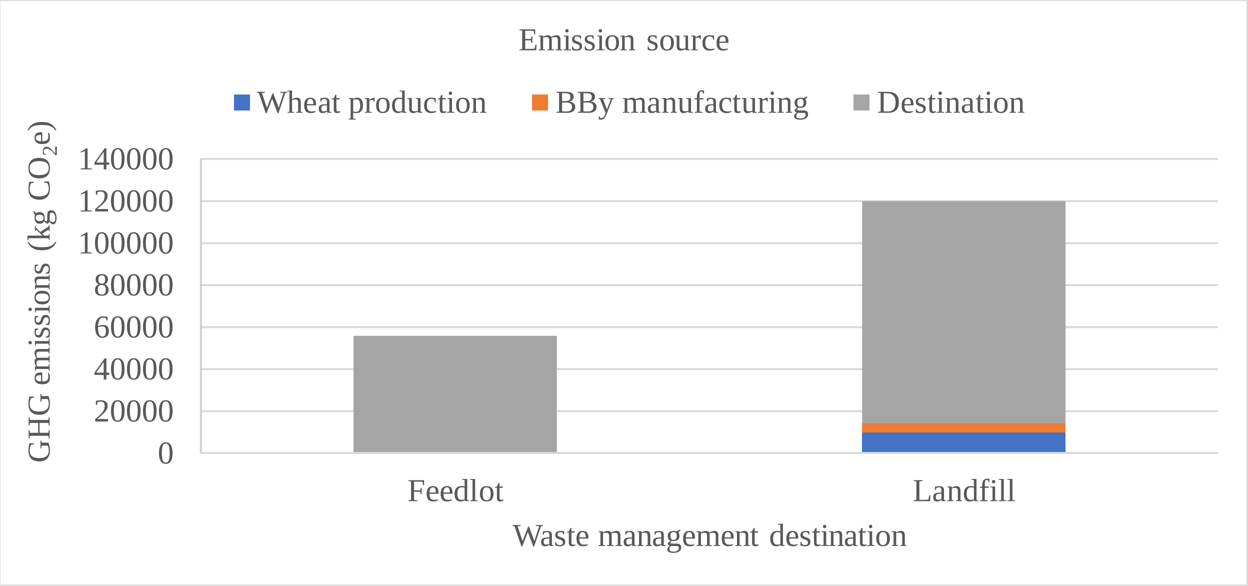

Figure 4.8 Greenhouse gas emissions (GHG) associated with bread by-product (BBy) that is fed to finishing steers in a feedlot or landfilled as waste management strategies (not including transportation).

4.4.3 Impact of bread waste inclusion in feedlot diets on ammonia emissions

Nitrogen excretion

Nitrogen excretion from GBBy (0.176 kg hd^{-1} d^{-1}) was 3% lower than GC (0.181 kg hd^{-1} d^{-1}). Nitrogen excretion from LBBy (0.167 kg hd^{-1} d^{-1}) was 5% lower compared to LC (0.175 kg hd^{-1} d^{-1}), while both MBBy (0.170 kg hd^{-1} d^{-1}) and HBBy (0.174 kg hd^{-1} d^{-1}) had reductions of 4% compared to MC (0.177 kg hd^{-1} d^{-1}) and HC (0.182 kg hd^{-1} d^{-1}). The proportion of excreted N was 84% - 86% for growing and finishing steers (all treatments) fed both diets (Figure 4.9).

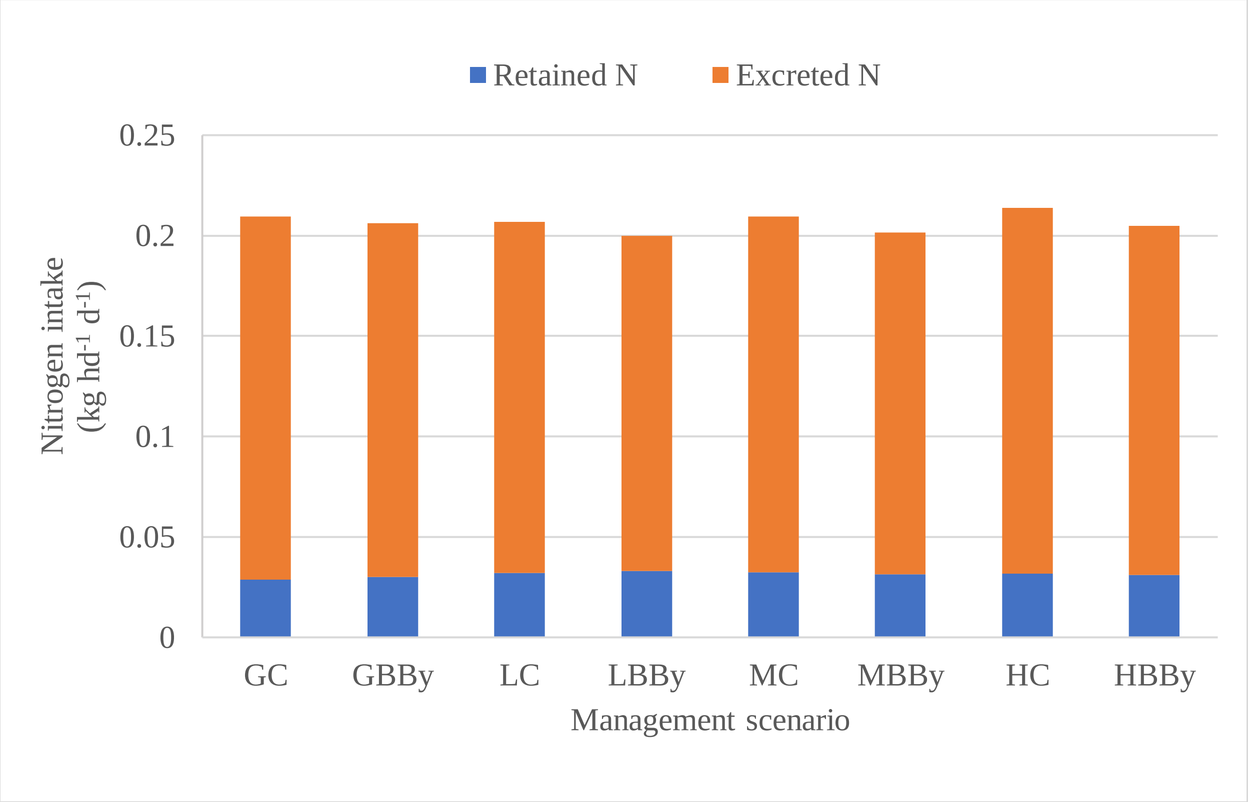

Figure 4.9 Nitrogen retention and excretion for growing (G) and finishing steers (L=Light weight; M=Medium weight; H=Heavy weight) fed corn (C) or bread by-product (BBy) diets.

Ammonia emissions

Ammonia emission intensity associated with GBBy (10.5 kg NH_3 hd^{-1}) was 4% lower than GC (10.8 kg NH_3 hd^{-1}; Figure 4.10). Emissions associated with housing accounted for 88% of NH_3 emissions from growing steers during the feeding period with manure storage and land application both accounting for 6% of emissions. Estimated NH_3 emission intensities were 1.3, 1.0, and 1.5% lower for FBBy (L=15.5; M=14.5; and H=13.2 kg NH_3 hd^{-1}) compared to FC (L=15.7; M=14.7; and H=13.4 kg NH_3 hd^{-1}). Housing accounted for 83% of NH_3 emissions in L steers and 86% in both M and H steers. Manure storage and land application accounted for 8% and 9% of NH_3 emissions in L steers, respectively and 7% each for both M and H steers (Figure 4.10).

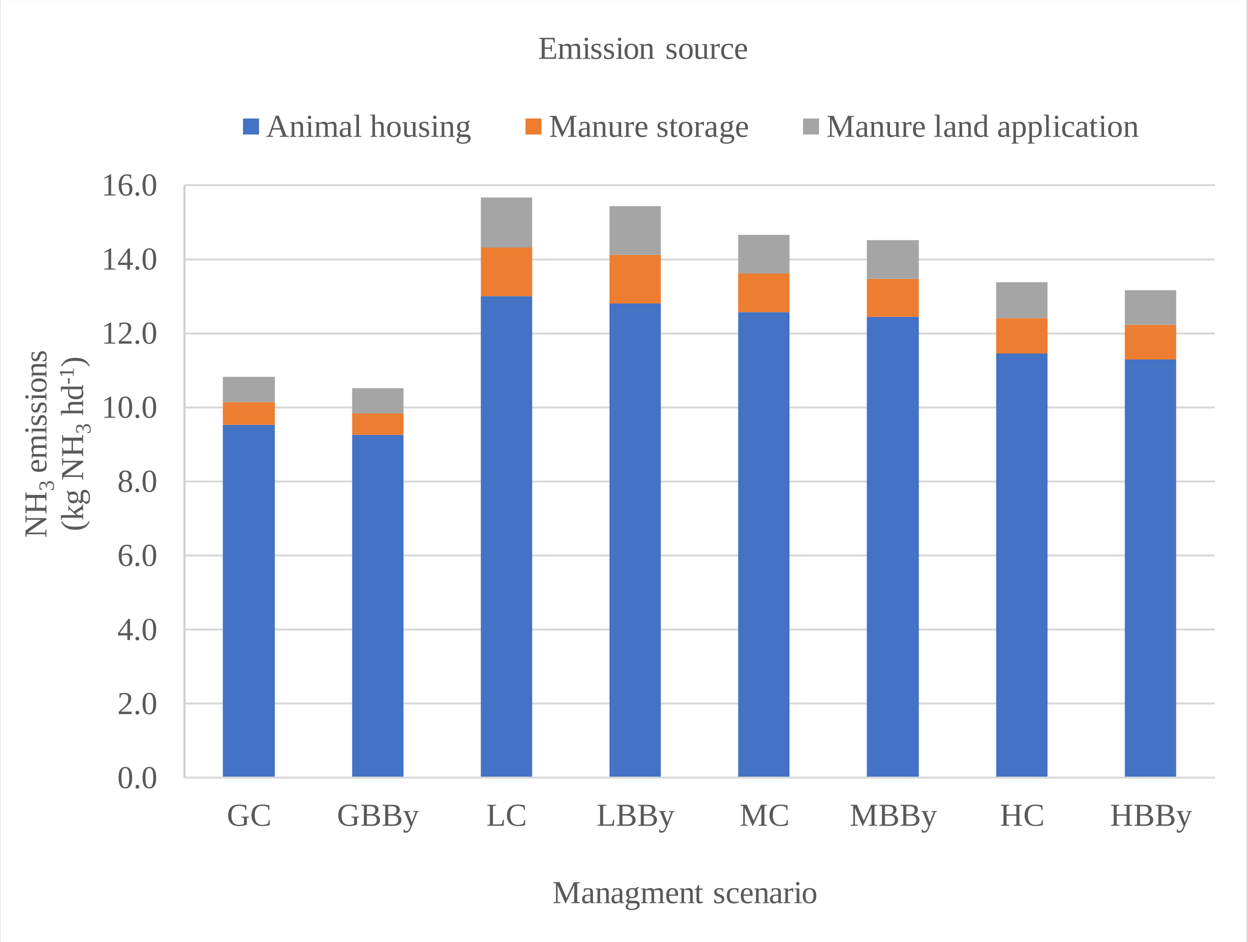

Figure 4.10 Ammonia (NH_3) emissions by source for growing (G) and finishing steers (L=Light weight; M=Medium weight; H=Heavy weight) fed corn (C) or bread by-product (BBy) diets.

When observing NH_3 emissions on a kg NH_3 hd^{-1} d^{-1} basis, there was a 3% reduction in NH3 emissions with GBBy as compared to GC, while there was a 4% reduction on a kg NH_3 hd^{-1} d^{-1} basis with FBBy compared to FC (Figure 4.11). Light steers fed the BBy-based diet had the lowest emissions (0.117 kg NH_3 hd^{-1} d^{-1}), while HC had the highest (0.126 kg NH_3 hd^{-1} d^{-1}) emissions of the finishing diet. Ammonia emissions from housing (kg NH_3 hd^{-1} d^{-1}) L steers were 8% and 10% lower than M and H steers, respectively. Regardless of diet, kg NH_3 emissions hd^{-1} associated with H steers were less than M steers, which were less than L steers. However, when calculated on a kg NH_3 hd^{-1} d^{-1} basis, emissions from L steers were less than M steers, which were less than H steers.

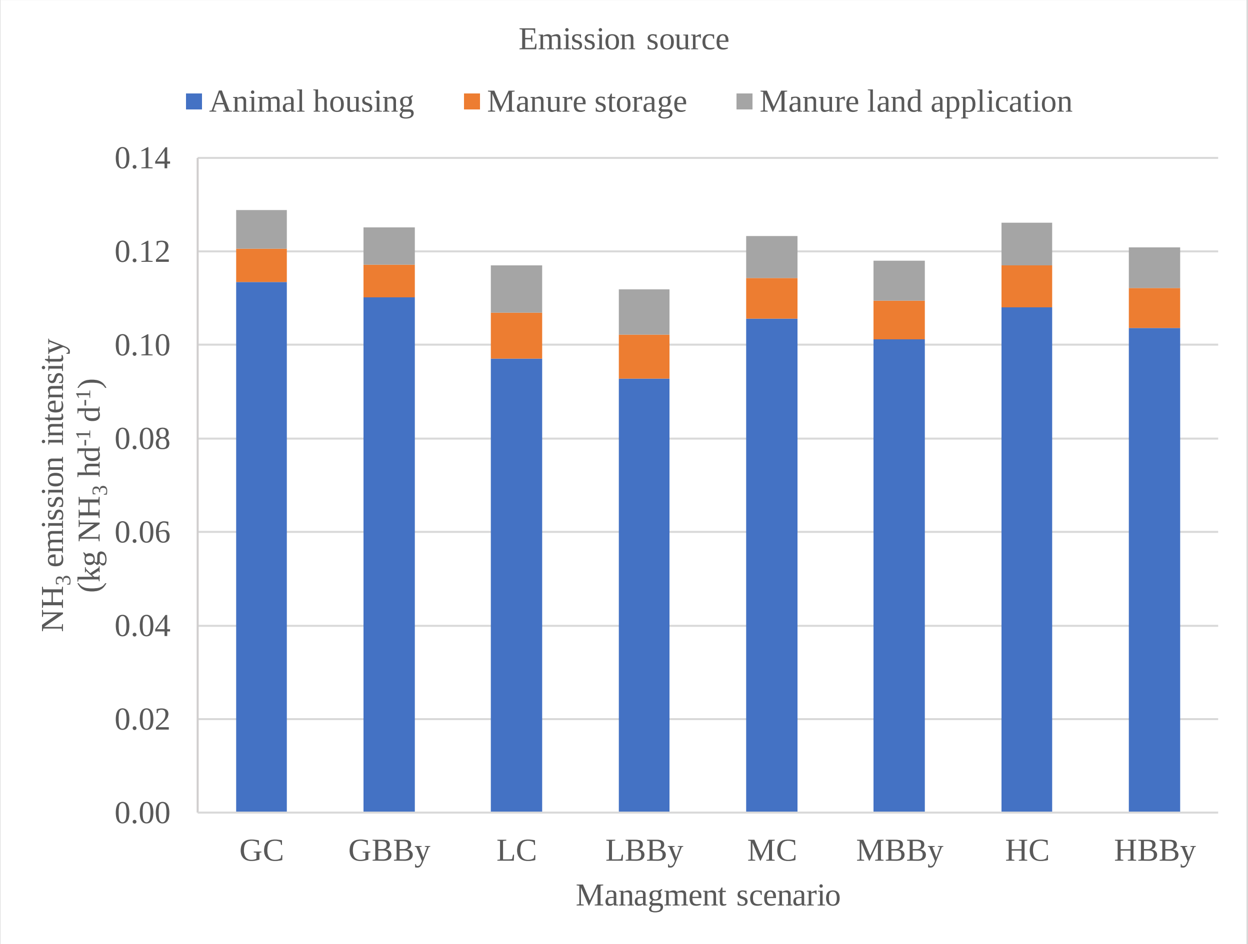

Figure 4.11 Ammonia (NH_3) emission intensity by source for growing (G) and finishing steers (L=Light weight; M=Medium weight; H=Heavy weight) fed corn (C) or bread by-product (BBy) diets.

4.4.4 Impact of bread waste inclusion in feedlot diets on water use

More than 99% of water use for all steers was attributed to crop production (Figure 4.12). However, crop water requirements were 36% lower for GBBy compared to GC (Figure 4.13). Similarly, crop water use associated with FBBy was 62% lower compared to FC. Water requirements for the corn grain in GBBy and FBBy diets were 92% and 78% lower, respectively as compared to the conventional corn diets. Water requirements for soybean production declined in both growing (77%) and finishing diets (68%) in the BBy diet as compared to the corn diet. The water requirements for corn silage for GBBy and FBBy were 8% and 6% lower, respectively, compared to the corn diets. Further, the water requirements for hay were 8% and 37% lower for GBBy and FBBy, respectively as compared to the corn diets.

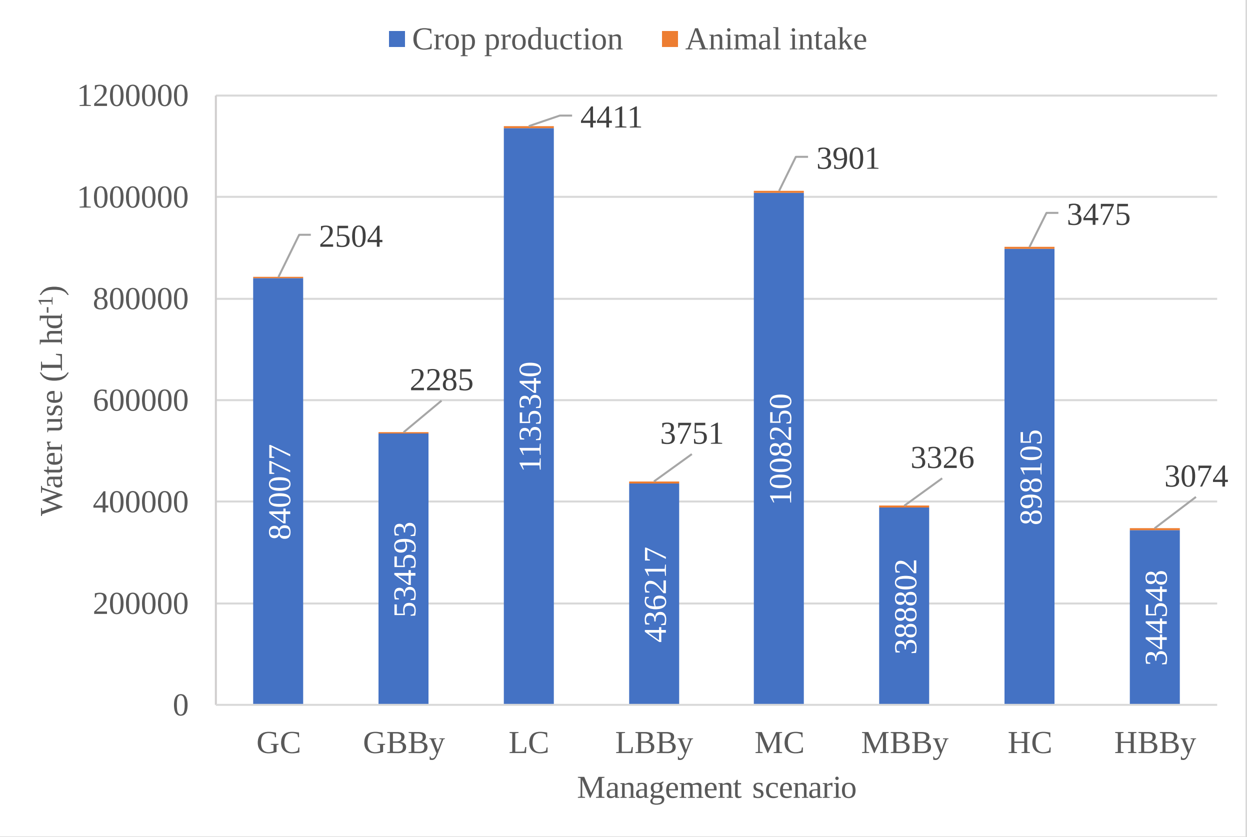

Figure 4.12 Water use associated with crop production and drinking water for growing (G) and finishing steers (L=Light weight; M=Medium weight; H=Heavy weight) fed corn (C) or bread by-product (BBy) diets.

The majority of water required for crop production in growing steer diets was associated with the production of alfalfa hay for GC (55%) and GBBy (80%) diets. However, in finishing diets, corn accounted for the greatest proportion of water use for FC (56%) and FBBy (32%). Within diets, water requirements for H steers were lower than M steers, which were lower than L steers.

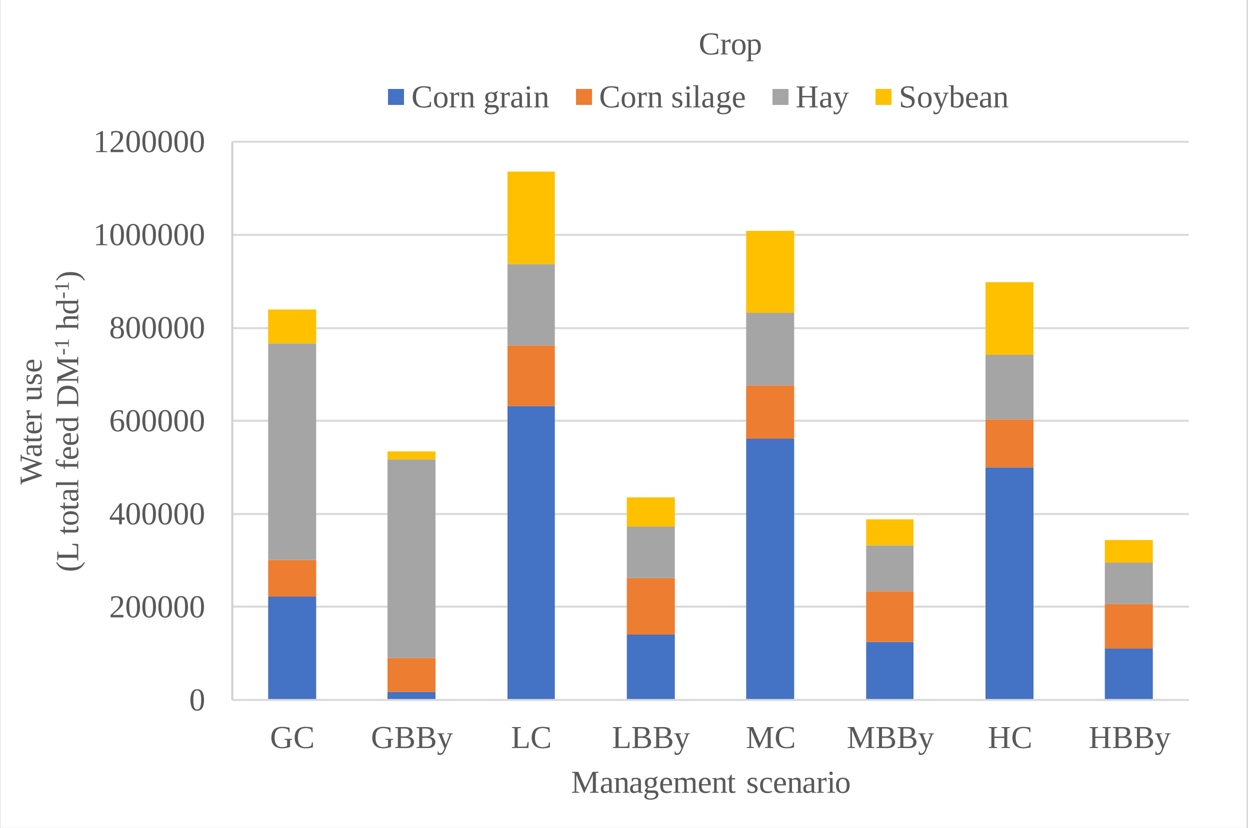

Figure 4.13 Total water use by feed crop for growing (G) and finishing steers (L=Light weight; M=Medium weight; H=Heavy weight) fed corn (C) or bread by-product (BBy) diets.

Water consumption was 9% less for GBBy than GC (Figure 4.14). Including BBy in growing steer diets also resulted in a 37% reduction in water use intensity (WUI) on a LW basis. Furthermore, animal water intake was 15% lower for HBBy and MBBy as compared to their corn-fed counterparts. However, there was only a 12% reduction when LBBy were compared to those fed corn. Compared to steers fed corn diets, water use intensity for FBBy was 61% lower on a LW basis and 21%, 19%, and 18% lower on boneless beef basis for LBBy, MBBy, and HBBy, respectively.

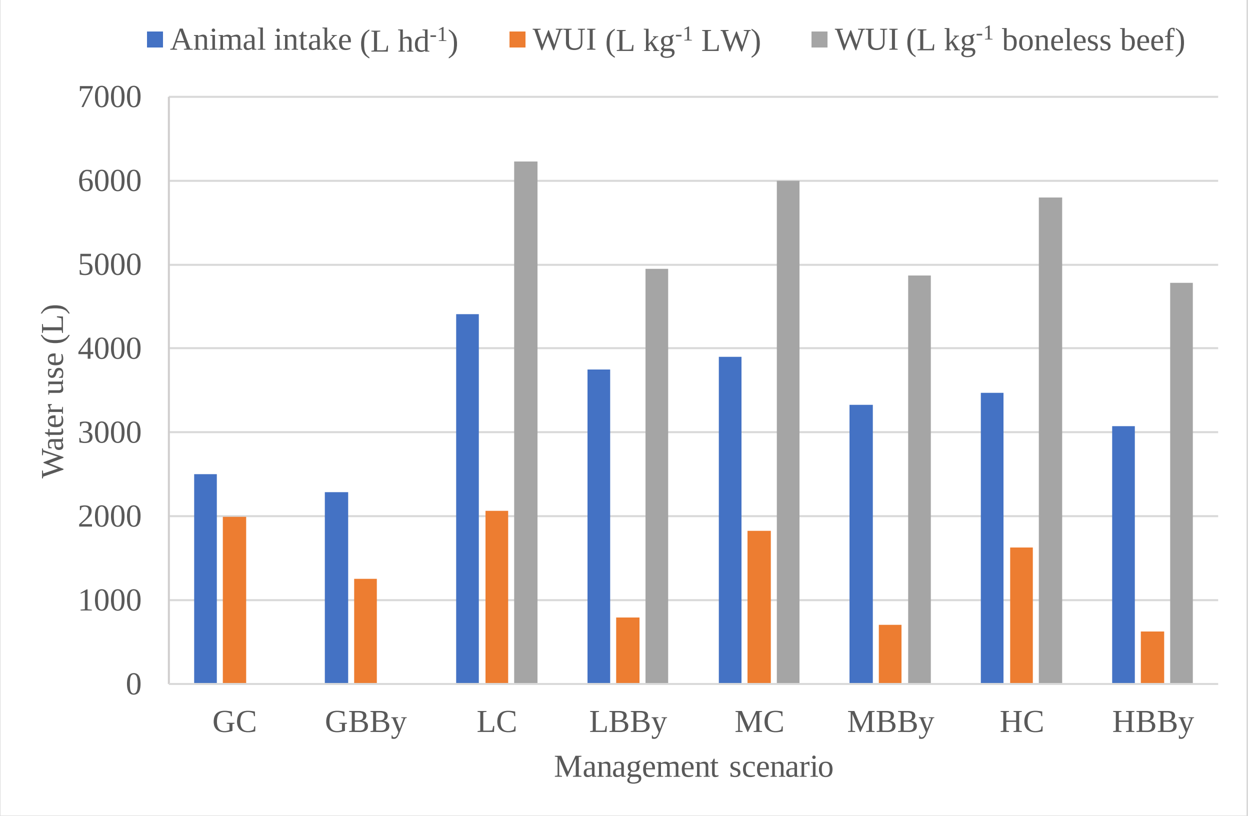

Figure 4.14 Animal water intake and water use intensities (WUI) for growing (G) and finishing steers (L=Light weight; M=Medium weight; H=Heavy weight) fed corn (C) or bread by-product (BBy) diets.

4.5 DISCUSSION

Land use requirements

The observed 45% (Figure 4.1) and 63% (Figure 4.2) decrease in land requirements associated with GBBy and FBBy, compared to GC and FC, respectively, can primarily be attributed to substitution of BBy for corn grain as an energy source in these diets. In addition, BBy has a higher CP content than corn (16% vs. 8-10% CP DM; Lardy, 2018; Guiroy et al., 2000), reducing the amount of SBM that was needed in the diet to meet CP requirements.

Although the proportion of corn silage and alfalfa hay were the same in the corn and BBy growing diets (Table 4.2), land use was reduced by 9% and 8%, respectively, for GBBy as compared to GC. Lower land use was a reflection of the lower DMI (9.3 kg d^{-1}) of steers fed GBBy as compared to GC (10.1 kg d^{-1}), reducing demand for both feeds. The lower DMI, with no difference in ADG (1.5 vs. 1.4 kg d^{-1}), resulted in an improvement in the feed:gain ratio. Similarly, FBBy had a lower DMI compared to FC (10.17 vs 11.39 kg d^{-1}) across all weight classes, with a lower proportion of Timothy hay in the BBy-based diet (Table 4.5), leading to a 36% decrease in the land required to produce Timothy hay. The relatively small reduction (6%) in the land required to produce corn silage for the FBBy can be attributed to a lower DMI. Although not statistically significant, FBBy had a lower ADG (1.54 vs. 1.59 kg d^{-1}) compared to FC resulting in 3-4 more days on feed for the steers to reach the target weight of 553 kg, but the feed:gain ratio of FBBy steers remained lower than FC steers (6.64 vs 7.22; $P = < 0.01$).

Observed differences in feed:gain ratios may be attributed to different degrees of processing and carbohydrate type. The corn in both growing and finishing diets were minimally processed (dry rolled and whole shelled, respectively), compared to the BBy which contained processed starch due to the milling of the wheat and baking of the bread. Guiroy et al. (2000) attributed the improved feed:gain ratio to 90% B_1 carbohydrate fraction (rumen escaped starch) of BBy compared to 63% observed in the corn. Starch digested in the small intestine rather than in the rumen produces less CH_4 and direct absorption of glucose can more energetically favorable for the host (Owens and Soderlund, 2006). While starch in dry rolled and whole corn grain is more likely to pass through the rumen undigested compared to a steam flaked or high-moisture corn, it is poorly digested within the small intestine, reducing the energy available for growth (Owens and Soderlund, 2006). Alternatively, BBy may have been more digestible than the corn fed in both growing and finishing diets due to the extensive nature of its processing. Another explanation for improved feed efficiency could be due to a more optimal metabolizable glucose (MG) to metabolizable protein (MP) ratio. Sun et al. (2018) found that a diet containing a MG/MP ratio of 1.13 resulted in improved growth and dietary protein and energy utilization in dairy heifers.

The lower feed:gain ratios in steers fed BBy led to a reduction in demand for all feed ingredients and therefore land requirements, an outcome also reported for finishing steers (Milton and Brandt, 1994) and fattening pigs (Sirtori et al., 2007). Substitution of FLW for main dietary ingredients has also resulted in a reduction in land use in other studies. For example, Quintero-Herrara et al. (2021) observed that substituting broccoli stems for corn silage in the diet of dairy cattle at a rate of 11% DM decreased land use intensity by 0.002 $m^2a\ kg^{-1}$ fat-and-protein corrected milk (FPCM). Further, dairy cows fed 7.1 kg fresh citrus waste $hd^{-1}\ d^{-1}$ in place of proportions of corn and rye silages, grass hay, and ground corn of the control diet spared 0.09 ha hd^{-1} (Baker et al., 2024).

On-farm greenhouse gas emissions and emission intensity

The overall reductions in on-farm GHG emissions (kg CO_2e kg hd^{-1}) for both GBBy (14%; Figure 4.3) and FBBy (20%; Figure 4.4) can mainly be attributed to the need to grow less corn. As a result, differences in farm energy and upstream CO_2 emissions were greatest between the two diets for both growing (61% and 75%, respectively) and finishing (64% and 67%, respectively) steers. Farm energy CO_2 emissions are primarily associated (>99%; Woods et al., 2010) with the burning of fuel in farm machinery during planting and harvesting, while upstream CO_2 emissions are associated with the production of synthetic fertilizer. The observed reduction in direct N_2O emissions from GBBy (28%) and FBBy (34%) reflect lower fertilizer use, as well as fewer crop residues available for decomposition. Consequently, indirect N_2O losses through volatilization, leaching, and run-off were reduced by 27% and 34% in growing and all finishing steers, respectively fed BBy compared to corn diets. The larger magnitude of the overall reduction in emissions between GC and GBBy compared to FC and FBBy reflects the greater inclusion rate of corn in the finishing (63% DM) as compared to the growing diet (40% DM). Minimal differences in enteric and manure CH_4 emissions were expected as the diets had similar nutrient profiles, including dietary NDF within both the growing and finishing experiments. Diets that contain more forage and higher NDF content lead to higher CH_4 emissions on an intensity basis (Santander et al., 2023).

Reduced GHG emissions associated with substituting FLW in diets of livestock have been previously reported. When broccoli stems were substituted for maize silage in the diet of dairy cattle at a rate of 11% DM, GHG emission intensity was reduced by 118 g CO_2e kg^{-1} FPCM (Quintero-Herrera et al., 2021). Reductions of 4% in CH_4 and 18% in N_2O emissions (kg CO_2e), occurred when fruit and vegetable waste were substituted for concentrate at 10% DM in the diets of sheep fed a mixed hay and concentrate diet (70:30; Sahoo et al., 2021). These authors speculated that the fruit and vegetable wastes used in their study were potential sources of phytochemicals with anti-methanogenic properties, leading to a decrease in CH_4 emissions. A reduction in GHG emissions (387,360 kg CO_2e) associated with feeding fresh citrus waste to cattle as compared to a conventional control diet (corn and rye silages, grass hay, and corn grain) has also been reported (Baker et al., 2024).

Off-farm greenhouse gas emissions

As a result of the higher CP of BBy (15.9% DM in growing steers; 16% in finishing steers) compared to corn (10% DM), there was a 75% and 65% decrease, in the inclusion of SBM in GBBy and FBBy diets, respectively (Figure 4.6). Greenhouse gas emissions associated with SBM production and processing were 77% and 68% lower for GBBy and FBBy, respectively, as compared to the corn diets. Although the majority (84%) of the emissions associated with SBM arose from the cultivation of soybeans, 15% of GHG emissions were from the process of oil extraction, which were reduced when the demand for SBM was lower. As transportation accounted for less than 2% of GHG emissions from SBM, they did not differ between the two diets. As noted by Desjardins et al. (2020), the carbon footprint associated with the cultivation of soybeans in Manitoba (0.08 kg CO_2e kg^{-1} DM) is significantly lower than the Canadian average (0.52 kg CO_2e kg^{-1} DM). This difference can be attributed to higher adoption rates of practices that support carbon sequestration in the Prairie Provinces, a climate that leads to lower N_2O emissions (drier), and large fields that allow for more efficient use of farm machinery and lower fossil fuel consumption (Desjardins et al, 2020). Therefore, higher emissions may be observed in other regions of the country.

By diverting BBy from landfill to the diets of feedlot steers, GHG emissions from growing and finishing steer production systems were reduced by 24% and 53%, respectively (Figures 4.7 and 4.8). The magnitude of difference between growing and finishing steers fed BBy can be attributed to the differences in proportion of BBy included in the diet, 40% DM and 55% DM, respectively. In this study, feedlot and bakery were in close proximity to each other (<5 km) resulting in minimal transportation emissions. The Impact Calculator developed by ReFED (https://insights-engine.refed.org/impact-calculator) suggests that a 45% reduction may be realized by diverting bread and bakery waste from landfill to animal feed when considering GHG emissions from livestock production systems.

The carbon footprint estimated for bread in this study (0.56 kg CO_2e kg^{-1} DM) is similar to that reported by for bread production in Sweden (0.48 kg CO_2e kg^{-1} DM; Eriksson et al., 2015). These same authors observed that the global warming potential (GWP) of bread waste from supermarkets decreased from 1.9 kg CO_2e kg^{-1} f to -0.13 kg CO_2e kg^{-1} when it was used as feed as opposed to landfilled. They also found that when including farm to retail emissions in their calculations, waste management had a substantial influence over the life cycle emissions of bread due to the relatively low carbon footprint associated with its production and high amount of waste (10% of supermarket waste included in the study). Diverting BBy to landfill as opposed to feeding it to cattle results in a substantial increase in GHG emissions.

Nitrogen excretion

Although the CP content of the growing diet containing BBy was slightly higher (14.1% CP) than the diet containing corn (13.8% CP), there was a 3% reduction in N excretion from GBBy compared to GC (Figure 4.9). The observed differences in N excretion can be attributed to an 8% lower DMI coupled with a 5% higher ADG for GBBy compared to GC. This resulted in an improved feed:gain ratio (6.17 vs. 7.05), with greater N retention (14.6% vs 13.6%). A similar trend was observed in finishing steers, where despite receiving isonitrogenous diets (12.7% CP), compared to steers fed the corn diet, N excretion rates were 4.6%, 4.0%, and 4.4% lower for LBBy, MBBy, HBBy, respectively. The DMI of FBBy was 11% lower than FC, however the ADG of FBBy was 3% higher, improving the feed:gain ratio (6.64 vs 7.22) and retaining more N. The highest N retention was observed in LBBy (17%) with the lowest in HC (15%). The percentage of retained N in both growing and finishing steers in these studies was within the 10-20% range reported for typical beef cattle feedlot diets (Hristov et al., 2011). Average N excreted among all growing and finishing steers was 85% of the total N intake, which is similar (84%) to that reported for beef cattle in Alberta (Chai et al., 2014). Total N intake that was lost as NH_3-N was 50% for GB and 51% for GC. Finishing steers were estimated to lose 46% (L), 48% (M), and 49% (H) of dietary N as NH_3-N. These numbers were similar to those reported by Legesse et al. (45%; 2018b) with regard to the amount of N intake that is lost as NH_3-N in feedlot diets.

NH_3 emissions and emission intensity

The 8 to 10 % reduction in NH_3 emission intensity (kg NH_3 hd^{-1} d^{-1}; Figure 4.11) in L steers fed both diets as compared to M and H steers, respectively, can be attributed to the feeding period which extended into the fall to enable these steers to reach the target weight of 553 kg. This lowered the average temperature of the feeding period for L steers to 14.4°C as compared to 16.2 C for both M and H groups, lowering NH_3 volatilization as reported by Koenig and McGinn (2016). Based on emission factors developed by Sheppard and Bittman (2012), NH_3 emission rates increase by 1.5-fold for every 10°C above 15°C (Legesse et al. 2018b). To the author's knowledge, there is no published literature that has included NH_3 emissions associated with the feeding of FLW. However, small reductions in NH_3 emissions were observed when cull potatoes were included in cattle diets at inclusion rates of 15% and 30% DM compared to traditional corn or barley grain diets (Mengistu, unpublished).

Water requirements

Crop production accounted for more than 99% of total water use in all management scenarios (Figure 4.12), agreeing with findings of Legesse et al. (2018a). The 36% reduction in crop water use in GBBy compared to GC resulted from a 92% and 77% reduction in the water needed for corn grain and soybean production, respectively (Figure 4.13). Furthermore, as GBBy resulted in lower DMI, the demand for corn silage and alfalfa hay were also reduced, further lowering the water demand for crop production. Similar trends were observed for FBBy, with a 62% reduction in total water use and 78% and 68% reductions in water use for corn grain, and soybeans, respectively. Water use for Timothy hay was also reduced (37%) for FBBy as its inclusion in the diet decreased from 9% to 6% with the addition of BBy to the diet. Further, lower DMI for FBBy resulted in lower demand for corn silage, although there was a slightly higher proportion of corn silage in the diet compared to FC. The magnitude of difference in total water use between growing and finishing steers can be attributed to larger proportions of corn silage and hay included in the growing diets which were not substituted with BBy.

The 9% reduction in water intake of GBBy compared to GC can be attributed to the lower DMI of GBBy as all other factors that influence water consumption (maximum temperature, precipitation and dietary salt) remained constant between the two diets (Figure 4.14). Water consumption is positively correlated with DMI (Brew et al., 2011), therefore steers with a lower DMI are expected to consume less water. The reduction in DMI and feed demand resulted in a 37% reduction in water use intensity (L kg^{-1} LW) associated with GBBy as compared to GC. Similarly, the 14% average reduction in water intake of FBBy compared to FC can also be attributed to reduced DMI. The 61% decrease in WUI (L kg^{-1} LW) for FBBy compared to FC was mainly attributed to lower water demand for corn production. This difference became less pronounced (21%, 19%, and 18% for L, M, and H, respectively) after water associated with processing was considered as the amount of water associated with crop production became a smaller proportion of total water requirements. Furthermore, the wheat required to produce BBy included in growing and finishing diets required 4 million and 25 million litres of water, respectively, that otherwise would have been wasted if BBy was landfilled (data not shown).

Baker et al. (2024) demonstrated that feeding fresh citrus waste to cattle, as compared to a diet containing corn and rye silages, grass hay, and corn grain, spared 83,040 m^3 of water that otherwise would have been used to produce feed crops. Often FLW is higher in water content than conventional feedstuffs (Dou et al., 2018) and its inclusion in the diet of livestock has been observed to reduce water intake. When sheep were fed a diet including 10% DM of fruit waste or vegetable waste, water intake was reduced by 22% and 14%, respectively (Sahoo et al., 2021). This reduction was due to the higher moisture content from the fruit (130% increase) and vegetable waste (119% increase) as compared to a control diet of hay and concentrate (70:30).

4.6 CONCLUSIONS

The results from this study suggest that utilizing BBy in feedlot diets can reduce the environmental impact of feedlot cattle by decreasing the demand for land and water as well as reducing GHG and NH_3 emissions as compared to a traditional corn diet. Growing steers fed BBy compared to corn had a 45%, 14%, 4%, and 37% reduction in land use (ha hd^{-1}), GHG emissions (kg CO_2e kg hd^{-1}), NH_3 emissions (g NH_3 kg^{-1} LW), and water use (L kg^{-1} LW), respectively. Finishing steers fed the BBy also had a 63%, 19%, 1%, and 61% in land use (ha hd^{-1}), GHG emissions (kg CO_2e kg hd^{-1}), NH_3 emissions (g NH_3 kg^{-1} LW), and water use (L kg^{-1} LW), respectively, as compared to those fed corn.

The observed improvements in these sustainability metrics can primarily attributed to the land and water that were not required for feed production, as well as the GHG emissions prevented from the production of conventional feed. Additionally, GHG emissions that would have been emitted from decaying BBy in landfill were avoided. Furthermore, the BBy feeding trials examined in this study demonstrate that BBy diets improved feed:gain, resulting in a lower demand for all ingredients.

Therefore, it is evident that utilizing BBy in beef cattle diets not only contributes to a reduction in the diversion of FLW to landfill and landfill-associated GHG emissions, but also supports a circular food economy by retaining BBy within the food production chain. Finally, by exploring the use of food waste as animal feed, this study suggests the potential for broader application of FLW within the livestock industry as a mitigation strategy for the food waste crisis.

5 GENERAL DISCUSSION

The Government of Canada announced its Strengthened Climate Plan which by 2030, includes a target of achieving GHG emission 30% lower than those reported in 2005 (ECCC, 2020b). The FAO (2017) has further suggested that integration of livestock into the circular food economy, by increasing the amount of human inedible by-products or waste products in animal diets, is a solution to climate change. Thus, incorporating food waste into livestock diets presents an opportunity to aid in addressing national and global sustainability challenges. Although feeding food waste to livestock has occurred for centuries, there has been a relatively recent resurgence in interest to improve the efficiency of food systems and mitigate their environmental burden. While published literature regarding the environmental impact of these practices is sparse, it is necessary to encourage widespread adoption of sustainable practices. Therefore, the aim of this study was to contribute to the limited body of literature investigating the environmental impacts of feeding FLW to livestock, with a focus on feedlot cattle.

Although there were notable reductions in GHG emissions from both growing and finishing steers fed diets containing BBy, the largest proportion of GHG emissions associated with feedlot cattle arose from enteric and manure CH_4 emissions, which differed little between BBy and corn diets. This suggests that feeding BBy will not be an appropriate strategy to mitigate animal derived CH_4. However, other FLW commodities have been reported to contain anti-methanogenic properties. Fruit waste comprised of papaya, pineapple and oranges included in the diet of sheep at 10% of diet DM, reduced enteric CH_4 emissions by 22% compared to a hay:concentrate (70:30) control diet (Sahoo et al., 2021). Additionally, these same authors observed a 13% reduction in CH_4 when vegetable waste comprised of okra, spinach, cauliflower, and cabbage was included in the diet at 10% diet DM. Furthermore, compared to a diet consisting of alfalfa and concentrate, substituting 35% of a cereal-based concentrate with tomato or cucumber waste in the diet of dairy goats reduced enteric CH_4 emissions by 39% and 29%, respectively (Romero-Huelva et al., 2012).

Crop by-products have been used in livestock diets for millennia. However, there is debate whether GHG emissions associated with the production of by-products and waste products should be allocated to the livestock consuming these feed by-products. For some crops like soybeans, the proportion of by-product (SBM) is greater on a mass basis than the primary product (oil) and therefore the system relies on there being a market for SBM to be economically viable. Wang et al. (2024) argued that the crop by-products used in their study (cotton seed and soybean hulls) should not have the GHG emissions associated with their production allocated to livestock as their primary purpose is for human consumption. Likewise, as food is produced for humans, GHG emissions from food waste should not be re-allocated to livestock as it is an effective waste management strategy that promotes circularity. Regardless, as estimated in this study and supported by ReFED (2023), GHG emissions from discarded food are larger when it is sent to landfill than when it is fed to livestock. Similarly, as the primary source of BBy is not produced for feed, the resources (land, water, and fertilizer) expended for wheat production and baking of bread should not be allocated to beef production.

Although feeding FLW to livestock has both performance and environmental advantages, there are multiple challenges. The Canadian Food Inspection Agency (CFIA) regulates all manufactured animal feed, sold and imported in Canada. Currently the Canadian Feed Ingredients Table (CFIA, 2024a) includes 8 recycled food products, 4 of which (bakery waste, potato waste, snack food waste, dairy food by-product) specify that they must be dried and/or ground. Furthermore, the rapid spoilage of FLW (owing to its high moisture content) requires that it be fed or preserved within a few days after being received by producers. Although there are many methods to dry food waste (Nath et al., 2023), the process produces more GHG emissions from energy use or as a result of the manufacturing of desiccants and increases cost, potentially reducing the economic viability of using FLW. The low cost of food by-products and waste is one of the reasons producers are amenable to including it in livestock diets. Distance from FLW sources to livestock operations is also expected to influence feasibility with longer distances increasing transportation costs and the opportunity for spoilage. In order for large feedlots to feed FLW, sufficient amounts must be available on a consistent basis with a consistent nutrient profile. The BBy fed in this study had a consistent nutrient profile when it was received from the same source (Dvorak et al., 2001; Guiroy et al., 2000). However, when the source of by-products change, complete chemical analysis is required to formulate these ingredients into balanced diets. This also highlights the challenges of utilizing mixed (multi-ingredient) food waste from grocery stores or the service sector on a large-scale. Another challenge when utilizing FLW from retail is the presence of plastic packaging. In order to avoid feeding plastic to livestock, plastic packaging must be removed which could be time and labour intensive if not mechanized. Furthermore, if plastic is not effectively removed before feeding, consumption of plastic by livestock could result in gastro-intestinal illness, injury or sudden death (Nongcula et al., 2017).

Ensiling may offer a strategy to preserve FLW, thereby extending the shelf life and preventing the growth of potentially harmful molds and mycotoxins. Several studies have demonstrated that food waste can be ensiled to produce silage with desirable nutrient and fermentation profiles (Froetschel et al., 2014; Garcia Rodriguez et al., 2024). Froetschel et al.

(2014) observed that energy (approximately 85% TDN) of ensiled FLW was the least variable nutritive component when comparing FLW across lots, suggesting that FLW may be used as a substitute for other high energy feeds. Ensiling allows for producers to receive large batches of FLW from the same source, reducing seasonal variability without the need to feed it immediately to avoid spoilage. Combining high moisture commodities (fruits and vegetables) with drier commodities (BBy) can produce a silage with a higher DM (Garcia Rodriguez et al., 2024), thereby avoiding nutrient losses and soil contamination as a result of effluent (Froetschel et al., 2014; Garcia Rodriguez et al., 2024). Further investigation is required to expand our knowledge of nutritional quality and sensory characteristics of ensiled FLW. Sensory issues such as texture, odour, and taste of some FLW may negatively affect palatability and DMI and have often been overlooked (Favreau- Peigné et al., 2013) due to the sparsity of information (Boumans et al., 2022).

The magnitude of the environmental impact associated with diverting FLW from landfill may differ based on the environmental metrics considered (Vandermeersch et al., 2014). While Eriksson et al. (2015) found that bread waste had a lower GWP when diverted from landfill to anaerobic digestion compared to when it was used as animal feed, Vandermeersch et al. (2014) observed that diverting food waste to animal feed resulted in a land sparing effect that was not observed with anaerobic digestion. However, both studies agreed that regional differences exist when examining the environmental impacts of food waste valorization. Additionally, Eriksson et al. (2015) stated that reducing the use of landfill as a waste management strategy would have the greatest potential for mitigating GHG emissions from retail food waste. A ban on sending organic wastes to landfills, as is the case in Sweden, would quickly encourage processors and retailers to develop alternative strategies, such as composting or partnering with livestock producers, to mitigate landfill emissions from food waste.

6 GENERAL CONCLUSION

In this study, including BBy into the diets of feedlot steers was observed to reduce land and water use as well as GHG and NH_3 emissions as compared to a conventional corn diet. However, despite substantial environmental benefits, utilizing FLW in livestock diets will only occur if it proves to be economically viable. Improving economic viability is possible if processors and retailers are incentivized to invest in green management strategies, such as diverting FLW from landfills to livestock operations. Furthermore, offering additional incentives to livestock producers beyond low opportunity cost feed through subsidized transport and/or storage infrastructure may expedite the adoption FLW as feed.

Several other challenges must be addressed to effectively use FLW as livestock feed. One significant issue is the heterogeneity of the nutrient profile, especially when multi-commodity waste from the retail sector is used. This variability complicates the formulation of balanced diets for livestock. Additionally, FLW, particularly fruits and vegetables, has a high moisture content, leading to rapid spoilage and increased transport costs. Moreover, regulatory challenges add another layer of complexity. While ensuring that FLW is safe for animal consumption is of upmost importance, regulatory approval that requires robust testing as well as sorting and processing can be costly and time consuming, further delaying the adoption of feeding FLW. These factors present logistical and economic hurdles that must be overcome to make FLW a viable feed option.

In conclusion, Canadian and US livestock producers are globally recognized for their high standards of animal care, and the quality and efficiency of their milk, meat, and egg production. Livestock producers have shown interest, ingenuity, and investment in replacing traditional feeds with by-products. By incorporating FLW into livestock diets, producers are addressing global challenges, while simultaneously improving the sustainability of animal-based production systems and enhancing food security. However, in order to optimize the opportunities associated with FLW in the diet of livestock, further collaboration between researchers, producers, regulators and policymakers is essential.

7 REFERENCES

Aboagye, I.A., Valappil, G., Dutta, B., Imbeault-Tétreault, H., Ominski, K.H., Cordeiro, M.R., Kröbel, R., Pogue, S.J. and McAllister, T.A. (2024). An Assessment of the Environmental Sustainability of Beef Production in Canada. Canadian Journal of Animal Science. doi:https://doi.org/10.1139/cjas-2023-0077.

Agriculture and Agri-Food Canada (2006). Crop Profile for Soybean in Canada. [online] Available at: https://publications.gc.ca/collections/collection_2009/agr/A118-10-25-2006E.pdf [Accessed 26 Jul. 2024].

Agriculture and Agri-food Canada (2021). Agricultural Ammonia Indicator. [online] agriculture.canada.ca. Available at: https://agriculture.canada.ca/en/environment/resource-management/indicators/agricultural-ammonia-indicator. [Accessed 25 Jul. 2024].

Agriculture and Agri-food Canada (2024). Holos (Version 4.0.0.576) [Computer program]. Available at: https://agriculture.canada.ca/en/agricultural-production/holos [Accessed 25 Jul. 2024].

Alemu, A.W. and Pogue, S.J. 2024. Livestock enteric methane emissions. Holos Version 4.0 – Algorithm Documentation, 131-147.

Alemu, A.W., Pogue, S.J., Beaulieu, D. and Mantle, P. (2024a) Manure carbon and CH4 emissions from livestock manure. Holos Version 4.0 – Algorithm Documentation, 156-159.
Alemu, A.W., Pogue, S.J., Beaulieu, D. and Mantle, P. (2024b) Manure nitrogen and direct N2O emissions from livestock manure in housing. Holos Version 4.0 – Algorithm Documentation, 164-189.

Al-Kaisi, M. (2000). Crop water use or evapotranspiration. [online] Available at: https://crops.extension.iastate.edu/encyclopedia/crop-water-use-or-evapotranspiration. [Accessed 23 Aug. 2024].

Allen, R.G. (2002). Crop evapotranspiration : guidelines for computing crop water requirements. [online] Rome: Food And Agriculture Organization Of The United States. Available at: http://agris.fao.org/agris-search/search.do?recordID=XF1999085851. [Accessed 26 Jul. 2024].

Babbitt, C.W., Neff, R.A., Roe, B.E., Siddiqui, S., Chavis, C. and Trabold, T.A. (2022). Transforming wasted food will require systemic and sustainable infrastructure innovations. Current Opinion in Environmental Sustainability, 54, p.101151. doi: https://doi.org/10.1016/j.cosust.2022.101151.

Baker, L., Bender, J., Ferguson, J., Rassler, S., Pitta, D., Chann, S. and Dou, Z. (2024). Leveraging dairy cattle to upcycle culled citrus fruit for emission mitigation and resource co-benefits: A case study. Resources, conservation and recycling, 203, pp.107452–107452. doi:https://doi.org/10.1016/j.resconrec.2024.107452.

Bakshi, M.P.S., Wadhwa, M. and Makkar, H.P.S. (2016). Waste to worth: vegetable wastes as animal feed. CAB Reviews: Perspectives in Agriculture, Veterinary Science, Nutrition and Natural Resources, 11(012). doi:https://doi.org/10.1079/pavsnnr201611012.

Ball, D., Bade, D., Lacefield, G., Martin, N. and Pinkerton, B. (1998). Minimizing losses in hay storage and feeding. National Forage Information Circular, 98, p.16.

Beaulieu, M.S. (2007). Estimation of water use in Canadian agriculture in 2001. [online] Statistics Canada. Available at: https://www150.statcan.gc.ca/n1/en/catalogue/21-601-M2007087 [Accessed 26 Jul. 2024].

Beef Cattle Research Council (2020). Silage Cost of Production - BeefResearch.ca. [online] Beef Research. Available at: https://www.beefresearch.ca/blog/silage-cost-of-production/. [Accessed 25 Jul. 2024].

Boumans, I.J.M.M., Schop, M., Bracke, M.B.M., de Boer, I.J.M., Gerrits, W.J.J. and Bokkers, E.A.M. (2022). Feeding food losses and waste to pigs and poultry: Implications for feed quality and production. Journal of Cleaner Production, [online] 378, p.134623. doi:https://doi.org/10.1016/j.jclepro.2022.134623.

Brancoli, P., Bolton, K. and Eriksson, M. (2020). Environmental impacts of waste management and valorisation pathways for surplus bread in Sweden. Waste Management, 117, pp.136–145. doi:https://doi.org/10.1016/j.wasman.2020.07.043.

Brew, M.N., Myer, R.O., Hersom, M.J., Carter, J.N., Elzo, M.A., Hansen, G.R. and Riley, D.G. (2011). Water intake and factors affecting water intake of growing beef cattle. Livestock Science, 140(1-3), pp.297–300. doi:https://doi.org/10.1016/j.livsci.2011.03.030.

Bryant, J.R. and Snow, V.O. (2008). Modelling pastoral farm agro-ecosystems: A review. New Zealand Journal of Agricultural Research, 51(3), pp.349–363. doi:https://doi.org/10.1080/00288230809510466.

Canadian Food Inspection Agency (2024a). Canadian Feed Ingredients Table (CFIT) - inspection.canada.ca. [online] Canada.ca. Available at: https://inspection.canada.ca/en/canadian-feed-ingredients-table-cfit [Accessed 25 Jul. 2024].

Canadian Food Inspection Agency (2024b). RG-1 Regulatory Guidance:Chapter 3 – Guidance on data requirements for feed approval and registration - inspection.canada.ca. [online] Canada.ca. Available at: https://inspection.canada.ca/en/animal-health/livestock-feeds/regulatory-guidance/rg-1/chapter-3 [Accessed 26 Jul. 2024].

Canadian Roundtable for Sustainable Beef (2016). Nation Beef Sustainability Assessment and Strategy summary report. [online] Calgary, AB. Available at: https://crsb.ca/wp-content/uploads/2021/12/Assessment-and-Strategy_summary_report_2016.pdf [Accessed 25 Jul. 2024].

Cereals Canada (2024). Wheat – Cereals Canada. [online] Available at: https://cerealscanada.ca/wheat/. [Accessed 26 Jul. 2024].

Chai, L., Kröbel, R., Janzen, H.H., Beauchemin, K.A., McGinn, S.M., Bittman, S., Atia, A., I. Edeogu, Macdonald, D.J. and Dong, R. (2014). A regional mass balance model based on total ammoniacal nitrogen for estimating ammonia emissions from beef cattle in Alberta Canada. Atmospheric Environment, 92, pp.292–302. doi:https://doi.org/10.1016/j.atmosenv.2014.04.037.

Dämmgen, U. and Hutchings, N.J. (2008). Emissions of gaseous nitrogen species from manure management: A new approach. Environmental Pollution, 154(3), pp.488–497. doi:https://doi.org/10.1016/j.envpol.2007.03.017.

Desjardins, R., Worth, D., Vergé, X., Maxime, D., Dyer, J. and Cerkowniak, D. (2012). Carbon Footprint of Beef Cattle. Sustainability, 4(12), pp.3279–3301. doi:https://doi.org/10.3390/su4123279.

Desjardins, R.L., Worth, D.E., Dyer, J.A., X. P. C. Vergé and McConkey, B. (2020). The Carbon Footprints of Agricultural Products in Canada. pp.1–34. doi:https://doi.org/10.1007/978-981-13-7916-1_1.

Dou, Z., Toth, J.D. and Westendorf, M.L. (2018). Food waste for livestock feeding: Feasibility, safety, and sustainability implications. Global Food Security, 17, pp.154–161. doi:https://doi.org/10.1016/j.gfs.2017.12.003.

Dou, Z., Toth, J.D., Pitta, D.W., Joseph, J.S., Hennessy, M.L., Vecchiarelli, B., Nagaraju Indugu, Chen, T., Li, Y., Sherman, R., Deutsch, J., Hu, B., Shurson, G.C., Parsons, B. and Baker, L.D. (2022). Developing Novel Feeds for Cattle from Food Waste and Crop Residue Biomass to Strengthen the Regenerative Capacity of Agri-Food Systems. Research Square (Research Square). doi:https://doi.org/10.21203/rs.3.rs-1542984/v1.

Dvorak, W.W., Bauer, M.L., Lardy, G.P., Loe, E.R. (2001). Bread by-product for growing beef steers: Effects on steer performance and by-product variability. Journal of Animal Science. 79, 11

Environment and Climate Change Canada (2020a). National Inventory Report 1990–2018: Greenhouse Gas Sources and Sinks in Canada. [online] Available at: https://publications.gc.ca/collections/collection_2020/eccc/En81-4-2018-2-eng.pdf [Accessed 26 Jul. 2024].

Environment and Climate Change Canada (2020b). A healthy environment and a healthy economy. [online] Gatineau, QC: Environment and Climate Change Canada. Available at: https://www.canada.ca/en/services/environment/weather/climatechange/climate-plan/climate-plan-overview/healthy-environment-healthy-economy.html [Accessed 26 Jul. 2024].

Environment and Climate Change Canada (2024). Global warming potentials. [Online]. Available at: https://www.canada.ca/en/environment-climate-change/services/climate-change/greenhouse-gas-emissions/quantification-guidance/global-warming-potentials.html. [Accessed 26 Jul. 2024].

Eriksson, M., Strid, I. and Hansson, P.A. (2015). Carbon footprint of food waste management options in the waste hierarchy – a Swedish case study. Journal of Cleaner Production, 93, pp.115–125. doi:https://doi.org/10.1016/j.jclepro.2015.01.026.

Espinoza-Orias, N., Stichnothe, H. and Azapagic, A. (2011). The carbon footprint of bread. The International Journal of Life Cycle Assessment, 16(4), pp.351–365. doi:https://doi.org/10.1007/s11367-011-0271-0.

Falkowski, P., Scholes, R.J., Boyle, E.E.A., Canadell, J., Canfield, D., Elser, J., Gruber, N., Hibbard, K., Högberg, P., Linder, S. and Mackenzie, F.T. (2000). The Global Carbon Cycle: A Test of Our Knowledge of Earth as a System. Science, 290(5490), pp.291–296. doi:https://doi.org/10.1126/science.290.5490.291.

Favreau-Peigné, A., Baumont, R. and Ginane, C. (2012). Food sensory characteristics: their unconsidered roles in the feeding behaviour of domestic ruminants. animal, 7(5), pp.806–813. doi:https://doi.org/10.1017/s1751731112002145.

FAO (2011). Global Food Losses and Food Waste- Extent, Causes and Prevention. [online] Available at: https://www.fao.org/4/mb060e/mb060e00.pdf [Accessed 25 Jul. 2024].

FAO (2013). Impacts on natural resources Food wastage footprint. [online] Available at: https://www.fao.org/4/i3347e/i3347e.pdf. [Accessed 25 Jul. 2024].

FAO (2017). Livestock solutions for climate change. [online] openknowledge.fao.org. FAO ; Available at: https://openknowledge.fao.org/items/2985e4e2-3c37-4e7c-aa7c-3655de93d53c. [Accessed 26 Jul. 2024].

FAO (2020). The State of Food Security and Nutrition in the World 2020. FAO, IFAD, UNICEF, WFP and WHO. doi:https://doi.org/10.4060/ca9692en.

Forwood, D.L., Hooker, K., Caro, E., Huo, Y., Holman, D.B., Meale, S.J. and Chaves, A.V. (2019). Crop Sorghum Ensiled With Unsalable Vegetables Increases Silage Microbial Diversity. Frontiers in Microbiology, 10. doi:https://doi.org/10.3389/fmicb.2019.02599.

Fouli, Y., Hurlbert, M. and Kröbel, R. (2021). Greenhouse gas emissions from Canadian agriulture estimates and measurements. In: The School of Public Policy. [online] Available at: https://www.policyschool.ca/wp-content/uploads/2021/11/JSC5_GHG-Emissions_Fouli-et-al.pdf.

Froetschel, M.A., Ross, C., Stewart, R.L., Azain, M.J., Michot, P. and Romdhane Rekaya (2014). Nutritional value of ensiled grocery food waste for cattle. Journal of Animal Science, 92(11), pp.5124–5133. doi:https://doi.org/10.2527/jas.2014-8126.

Garcia Rodriguez, V., Vandestroet, L., Abeysekara, V.C., Ominski, K., Bumunang, E.W., McAllister, T., Terry, S., Miranda-Romero, L.A. and Stanford, K. (2024). Optimizing Silage Strategies for Sustainable Livestock Feed: Preserving Retail Food Waste. Agriculture, 14(1), pp.122–122. doi:https://doi.org/10.3390/agriculture14010122.

Gooch, M. and Felfel, A. (2014). Canada's Annnual Food Waste - $27 Billion Revisited. [online] Value Chain Management International Inc. Available at: www.vcm-international.com [Accessed 25 Jul. 2024].

Gooch, M., Felfel, A., and Marenick, N. (2010). Food waste in Canada. Opportunities to increase the competitiveness of Canada's agri-food sector, while simultaneously improving the environment. Value Chain Management Centre. Canada. Available at: https://www.researchgate.net/publication/285832431_Food_waste_in_Canada_opportunities_to_increase_the_competitiveness_of_Canada%27s_agri-food_sector_while_simultaneously_improving_the_environment [Accessed 25 Jul. 2024].

Gooch, M., Bucknell, D., LaPlain, D., Dent, B., Whitehead, P., Felfel, A., Nikkel, L., Maguire, M. (2019). The Avoidable Crisis of Food Waste: Technical Report; Value Chain Management International and Second Harvest; Ontario, Canada.

Government of Manitoba (1998). Livestock Manure and Mortalities Management Regulation. [online] Available at: https://web2.gov.mb.ca/laws/regs/current/_pdf-regs.php?reg=42/98 [Accessed 26 Jul. 2024].

Guiroy, P.J., Fox, D.G., Beermann, D.H. and Ketchen, D.J. (2000). Performance and meat quality of beef steers fed corn-based or bread by-product-based diets. Journal of Animal Science, 78(3), p.784. doi:https://doi.org/10.2527/2000.783784x.

Halmemies-Beauchet-Filleau, A., Rinne, M., Lamminen, M., Mapato, C., Ampapon, T., Wanapat, M. and Vanhatalo, A. (2018). Review: Alternative and novel feeds for ruminants: nutritive value, product quality and environmental aspects. Animal, 12, pp.s295–s309. doi:https://doi.org/10.1017/s1751731118002252.

Hein, T. (2020). Agriculture in Canada | The Canadian Encyclopedia. [online] www.thecanadianencyclopedia.ca. Available at: https://www.thecanadianencyclopedia.ca/en/article/agriculture-in-canada. [Accessed 26 Jul. 2024].

Heinrichs, J. and Kmicikewycz, A. (2023). Sub-Acute Rumen Acidosis and Physically Effective Fiber. [online] Available at: https://extension.psu.edu/sub-acute-rumen-acidosis-and-physically-effective-fiber.

Helmcke, S., Hundertmark, T., Musso, C., Ong, W.J., Oxgaard, J. and Wallach, J. (2022). Climate impact of plastics. [online] McKinsey & Company. Available at: https://www.mckinsey.com/industries/chemicals/our-insights/climate-impact-of-plastics#/ [Accessed 25 Jul. 2024].

Herrero, M., Havlik, P., Valin, H., Notenbaert, A., Rufino, M.C., Thornton, P.K., Blummel, M., Weiss, F., Grace, D. and Obersteiner, M. (2013). Biomass use, production, feed efficiencies, and greenhouse gas emissions from global livestock systems. Proceedings of the National Academy of Sciences, [online] 110(52), pp.20888–20893. doi:https://doi.org/10.1073/pnas.1308149110.

Hicks, R.B., Owens, F.N., Gill, D.R., Martin, J.J. and Strasia, C.A. (1988). Water intake by feedlot steers. [online] Oklahoma Agricultural Experiment Station. Available at: https://extension.okstate.edu/programs/beef-extension/research-reports/site-files/documents/1988/88-45.pdf [Accessed 26 Jul. 2024].

Holtshausen, L., Benchaar, C., Kröbel, R. and Beauchemin, K.A. (2021). Canola Meal versus Soybean Meal as Protein Supplements in the Diets of Lactating Dairy Cows Affects the Greenhouse Gas Intensity of Milk. Animals, 11(6), p.1636. doi:https://doi.org/10.3390/ani11061636.

Hristov, A.N., Hanigan, M., Cole, A., Todd, R., McAllister, T.A., Ndegwa, P.M. and Rotz, A. (2011). Review: Ammonia emissions from dairy farms and beef feedlots. Canadian Journal of Animal Science, 91(1), pp.1–35. doi:https://doi.org/10.4141/cjas10034.

IPCC (2006). Emissions from livestock and manure management. In: 2006 IPCC Guidelines for National Greenhouse Gas Inventories. [online] Available at: https://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/4_Volume4/V4_10_Ch10_Livestock.pdf [Accessed 26 Jul. 2024].

IPCC (2019). 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories — IPCC. [online] Ipcc.ch. Available at: https://www.ipcc.ch/report/2019-refinement-to-the-2006-ipcc-guidelines-for-national-greenhouse-gas-inventories/. [Accessed 26 Jul. 2024].

Jose, V.S., Sejian, V., Bagath, M., Ratnakaran, A.P., Lees, A.M., Al-Hosni, Y.A.S., Sullivan, M., Bhatta, R. and Gaughan, J.B. (2016). Modeling of Greenhouse Gas Emission from Livestock. Frontiers in Environmental Science, 4. doi:https://doi.org/10.3389/fenvs.2016.00027.

Jurgilevich, A., Birge, T., Kentala-Lehtonen, J., Korhonen-Kurki, K., Pietikäinen, J., Saikku, L. and Schösler, H. (2016). Transition towards Circular Economy in the Food System. Sustainability, [online] 8(1), p.69. doi:https://doi.org/10.3390/su8010069.

Kertz, A.F. (1998). Variability in Delivery of Nutrients to Lactating Dairy Cows. Journal of Dairy Science, 81(11), pp.3075–3084. doi:https://doi.org/10.3168/jds.s0022-0302(98)75872-2.

Koenig, K.M. and McGinn, S.M. (2016). Effect of temperature on ammonia emissions from feedlot cattle manure. Journal of Animal Science, 94(suppl_5), pp.569–570. doi:https://doi.org/10.2527/jam2016-1187.

Lardy, G. (2018). Feeding Corn to Beef Cattle. [online] Fargo, North Dakota: NDSU Extension. Available at: https://www.researchgate.net/publication/325654007_Feeding_Corn_to_Beef_Cattle_AS_1238#fullTextFileContent [Accessed 26 Jul. 2024].

Lardy, G., Anderson, V., Dahlen, C. and Carlson, Z. (2015). Alternative Feeds for Ruminants General Concepts and Recommendations for Using Alternative Feeds. [online] Available at: https://www.ndsu.edu/agriculture/sites/default/files/2022-03/as1182_0.pdf. [Accessed 25 Jul. 2024].

Lee, U., Han, J. and Wang, M. (2017). Evaluation of landfill gas emissions from municipal solid waste landfills for the life-cycle analysis of waste-to-energy pathways. Journal of Cleaner Production, [online] 166, pp.335–342. doi:https://doi.org/10.1016/j.jclepro.2017.08.016.

Legesse, G., Ominski, K.H., Beauchemin, K.A., Pfister, S., Martel, M., McGeough, E.J., Hoekstra, A.Y., Kroebel, R., Cordeiro, M.R.C. and McAllister, T.A. (2017). BOARD-INVITED REVIEW: Quantifying water use in ruminant production1. Journal of Animal Science, 95(5), pp.2001–2018. doi:https://doi.org/10.2527/jas.2017.1439.

Legesse, G., Cordeiro, M.R.C., Ominski, K.H., Beauchemin, K.A., Kroebel, R., McGeough, E.J., Pogue, S. and McAllister, T.A. (2018a). Water use intensity of Canadian beef production in 1981 as compared to 2011. Science of The Total Environment, 619-620, pp.1030–1039. doi:https://doi.org/10.1016/j.scitotenv.2017.11.194.

Legesse, G., Kroebel, R., Alemu, A.W., Ominski, K.H., McGeough, E.J., Beauchemin, K.A., Chai, L., Bittman, S. and McAllister, T.A. (2018b). Effect of changes in management practices and animal performance on ammonia emissions from Canadian beef production in 1981 as compared with 2011. Canadian Journal of Animal Science, 98(4), pp.833–844. doi:https://doi.org/10.1139/cjas-2017-0184.

Lipinski, B., Hanson, C., Waite, R., Searchinger, T. and Lomax, J. (2013). Reducing Food Loss and Waste. www.wri.org. [online] Available at: https://www.wri.org/research/reducing-food-loss-and-waste. [Accessed 25 Jul. 2024].

Loopresource.ca (2024). Loop Resource. [online] Available at: https://loopresource.ca/#about-us [Accessed 26 Jul. 2024].

MacRae, R., Siu, A., Kohn, M., Matsubuchi-Shaw, M., McCallum, D., Cervantes, T.H. and Perreault, D. (2016). Making better use of what we have: Strategies to minimize food waste and resource inefficiency in Canada. [online] 3(2), pp.145–215. doi:https://doi.org/10.15353/cfs-rcea.v3i2.143.

Manitoba Agricultural Services Corporation (2019). MMPP - Seeding Dates. [online] Available at: https://www.masc.mb.ca/masc.nsf/mmpp_seeding_dates.html. [Accessed 26 Jul. 2024].

Manitoba Agricultural Services Corporation (2022). MMPP - Variety Yield Data Browser. [online] Available at: https://www.masc.mb.ca/masc.nsf/mmpp_browser_variety.html. [Accessed 26 Jul. 2024].

Manitoba Agriculture (2024). 2024 Cost of Production - Silage. [online] Available at: https://www.gov.mb.ca/agriculture/farm-management/cost-production/pubs/cop-forage-cereal-silage.pdf [Accessed 26 Jul. 2024].

Matohlang Mohlotsane, P., Owusu-Sekyere, E., Jordaan, H., Barnard, J. and van Rensburg, L. (2018). Water Footprint Accounting Along the Wheat-Bread Value Chain: Implications for Sustainable and Productive Water Use Benchmarks. Water, 10(9), p.1167. doi:https://doi.org/10.3390/w10091167.

McGeough, E.J., Little, S.M., Janzen, H.H., McAllister, T.A., McGinn, S.M. and Beauchemin, K.A. (2012). Life-cycle assessment of greenhouse gas emissions from dairy production in Eastern Canada: A case study. Journal of Dairy Science, [online] 95(9), pp.5164–5175. doi:https://doi.org/10.3168/jds.2011-5229.

Mekonnen, M. and Hoekstra, A.Y., 2010. The green, blue and grey water footprint of farm animals and animal products. Volume 2: Appendices.

Mekonnen, M.M. and Hoekstra, A.Y. (2011). The green, blue and grey water footprint of crops and derived crop products. Hydrology and Earth System Sciences, [online] 15(5), pp.1577–1600. doi:https://doi.org/10.5194/hess-15-1577-2011.

Mengistu, G.F., Alemu, A.W., McAllister, T.A., Standford, K., Wittenberg, K., Aboagye, I.A., Cordeiro, M.R.C., Legesse, G., Gunte, K., Omonijo, F. and Ominski, K.H. (2022). Replacing Grains with Potato Waste in Feedlot Cattle: Greenhouse Gas Emissions and Land Use Requirements. In: Program & Abstracts. [online] 8th International Greenhouse Gas & Animal Agriculture Conference. Available at: https://conference.ifas.ufl.edu/ggaa/documents/GGAA-2022-Program-Abstracts-Online.pdf [Accessed 27 Jul. 2024].

Milton, C.T. and Brandt Jr, R.T. (1994). Dried bakery product in high concentrate rations: Finishing steer performance and estimated net energy values. Journal of Animal Science, 72(Suppl 2), p.80.

Mitloehner, F. (2020). The Carbon Impact of Food Waste: The Problem With What We're NOT Eating. CLEAR Center. Available at: https://clear.ucdavis.edu/blog/carbon-impact-food-waste-problem-what-were-not-eating [Accessed 25 Jul. 2024].

Mottet, A., de Haan, C., Falcucci, A., Tempio, G., Opio, C. and Gerber, P. (2017). Livestock: On our plates or eating at our table? A new analysis of the feed/food debate. Global Food Security, [online] 14, pp.1–8. doi:https://doi.org/10.1016/j.gfs.2017.01.001.

Nath, P.C., Ojha, A., Debnath, S., Sharma, M., Nayak, P.K., Sridhar, K. and Inbaraj, B.S. (2023). Valorization of Food Waste as Animal Feed: A Step towards Sustainable Food Waste Management and Circular Bioeconomy. Animals, [online] 13(8), p.1366. doi:https://doi.org/10.3390/ani13081366.

National Research Council (1996). Nutrient Requirements for Beef Cattle (7th Ed.). National Academy Press, Washington, DC.

National Zero Waste Council. (2022). Benchmark Study on Household Food Waste. [online] Available at: https://lovefoodhatewaste.ca/about/food-waste/. [Accessed Sept. 2023]

Nikkel, L., Maguire, M., Gooch, M., Bucknell, D., LaPlain, D., Dent, B., Whitehead, P., Felfel, A. (2019). The Avoidable Crisis of Food Waste: Roadmap; Second Harvest and Value Chain Management International; Ontario, Canada.

NOAA (2024). What is eutrophication? [online] National Ocean Service. Available at: https://oceanservice.noaa.gov/facts/eutrophication.html. [Accessed 26 Jul. 2024].

Nongcula, V., Zhou, L., Nhundu, K. and Jaja, I. (2017). Association between the Prevalence of Indigestible Foreign Objects in the Gastrointestinal Tract of Slaughtered Cattle and Body Condition Score. Animals, 7(12), p.80. doi:https://doi.org/10.3390/ani7110080.

Ominski, K., McAllister, T., Stanford, K., Mengistu, G., Kebebe, E.G., Omonijo, F., Cordeiro, M., Legesse, G. and Wittenberg, K. (2021). Utilization of by-products and food waste in livestock production systems: a Canadian perspective. Animal Frontiers, [online] 11(2), pp.55–63. doi:https://doi.org/10.1093/af/vfab004.

Organization for Economic Cooperation and Development and FAO (2020). OECD-FAO Agricultural Outlook. OECD. doi:https://doi.org/10.1787/1112c23b-en.

Owens, F. and Soderlund, S. (2006). Ruminal and postruminal starch digestion by cattle. [online] Iowa: Pioneer Hi-Bred. Available at: https://www.researchgate.net/publication/228504815_Ruminal_and_postruminal_starch_digestions_by_cattle [Accessed 26 Jul. 2024].

Palangi, V., Taghizadeh, A., Abachi, S. and Lackner, M. (2022). Strategies to Mitigate Enteric Methane Emissions in Ruminants: A Review. Sustainability, 14(20), p.13229. doi:https://doi.org/10.3390/su142013229.

Priestley, C.H.B. and Taylor, R.J. (1972). On the assessment of surface heat flux and evaporation using large-scale parameters. *Monthly weather review*, *100*(2), pp.81-92.

Prosser, J.I., Hink, L., Gubry-Rangin, C. and Nicol, G.W. (2019). Nitrous oxide production by ammonia oxidizers: Physiological diversity, niche differentiation and potential mitigation strategies. Global Change Biology, 26(1), pp.103–118. doi:https://doi.org/10.1111/gcb.14877.

Puente-Rodríguez, D., van Laar, H. and Veraart, M. (2022). A Circularity Evaluation of New Feed Categories in The Netherlands—Squaring the Circle: A Review. Sustainability, 14(4), p.2352. doi:https://doi.org/10.3390/su14042352.

Quintero-Herrera, S., García-León, A.M., Botello-Álvarez, J.E., Estrada-Baltazar, A., Abel-Seabra, J.E., Padilla-Rivera, A. and Rivas-García, P. (2021). The use of broccoli agro-industrial waste in dairy cattle diet for environmental mitigation. Cleaner Environmental Systems, [online] 2, p.100035. doi:https://doi.org/10.1016/j.cesys.2021.100035.

Rajeh, C., Saoud, I.P., Kharroubi, S., Naalbandian, S. and Abiad, M.G. (2020). Food loss and food waste recovery as animal feed: a systematic review. Journal of Material Cycles and Waste Management. doi:https://doi.org/10.1007/s10163-020-01102-6.

Randolph, T.F., Schelling, E., Grace, D., Nicholson, C.F., Leroy, J.L., Cole, D.C., Demment, M.W., Omore, A., Zinsstag, J. and Ruel, M. (2007). Invited Review: Role of livestock in human nutrition and health for poverty reduction in developing countries1,2,3. Journal of Animal Science, 85(11), pp.2788–2800. doi:https://doi.org/10.2527/jas.2007-0467.

ReFED (2023). ReFED - Impact Calculator. [online] Available at: https://insights-engine.refed.org/impact-calculator [Accessed 26 Jul. 2024].

Romero-Huelva, M., Ramos-Morales, E. and Molina-Alcaide, E. (2012). Nutrient utilization, ruminal fermentation, microbial abundances, and milk yield and composition in dairy goats fed diets including tomato and cucumber waste fruits. Journal of Dairy Science, 95(10), pp.6015–6026. doi:https://doi.org/10.3168/jds.2012-5573.

Roquette (2022). Animal nutrition – Pea cream. Available at: https://www.roquette.com/animal-nutrition-pea-cream [Accessed 25 Jul. 2024]

Sahoo, A., Sarkar, S., Lal, B., Kumawat, P., Sharma, S. and De, K. (2021). Utilization of fruit and vegetable waste as an alternative feed resource for sustainable and eco-friendly sheep farming. Waste Management, 128, pp.232–242. doi:https://doi.org/10.1016/j.wasman.2021.04.050.

Santander, D., Clariget, J., Banchero, G., Alecrim, F., Simon Zinno, C., Mariotta, J., Gere, J. and Ciganda, V.S. (2023). Beef Steers and Enteric Methane: Reducing Emissions by Managing Forage Diet Fiber Content. Animals, 13(7), p.1177. doi:https://doi.org/10.3390/ani13071177.

Schwab, D. (2021). Controlling Feed Loss and Spoilage Important for Beef Cattle Production. [online] Available at: https://www.extension.iastate.edu/news/controlling-feed-loss-and-spoilage-important-beef-cattle-production [Accessed 26 Jul. 2024].

Seed Manitoba (2013). Forage Crops. [online] Available at: https://www.seedmb.ca/wp-content/uploads/2016/12/seedmb_2013_forage_crops.pdf [Accessed 26 Jul. 2024].

Sheppard, S.C. and Bittman, S. (2012). Farm practices as they affect NH3 emissions from beef cattle. Canadian Journal of Animal Science, 92(4), pp.525–543. doi:https://doi.org/10.4141/cjas2012-055.

Shurson, G.C. (2020). 'What a Waste'—Can We Improve Sustainability of Food Animal Production Systems by Recycling Food Waste Streams into Animal Feed in an Era of Health, Climate, and Economic Crises?. Sustainability, 12(17), p.7071. doi:https://doi.org/10.3390/su12177071.

Sirtori, F., Acciaioli, A. and Pianaccioli, L. (2007). Effect of use of bread in fattening of Cinta Senese pig. Italian Journal of Animal Science/Italian journal of animal science, 6(sup1), pp.719–721. doi:https://doi.org/10.4081/ijas.2007.1s.719.

Statistics Canada (2023). Table: 38-10-0241-01: Total area that received irrigation by crop type. [online] Available at: https://www150.statcan.gc.ca/t1/tbl1/en/tv.action?pid=3810024101 [Accessed 28 Jul. 2024].

Sun, J., Xu, J., Ge, R., Wang, M., Yu, L. and Wang, H. (2018). Effects of different dietary ratio of metabolizable glucose and metabolizable protein on growth performance, rumen fermentation, blood biochemical indices and ruminal microbiota of 8 to 10-month-old dairy heifers. Asian-Australasian Journal of Animal Sciences, 31(8), pp.1205–1212. doi:https://doi.org/10.5713/ajas.17.0893.

UNEP (2021). UNEP Food Waste Index Report 2021. [online] UNEP - UN Environment Programme. Available at: https://www.unep.org/resources/report/unep-food-waste-index-report-2021. [Accessed 25 Jul. 2024].

UNEP (2024a). Plastic Pollution. [online] UNEP - UN Environment Programme. Available at: https://www.unep.org/plastic-pollution. [Accessed 25 Jul. 2024].

UNEP (2024b). Food Waste Index Report 2024. [online] UNEP - UN Environment Programme. Available at: https://www.unep.org/resources/publication/food-waste-index-report-2024. . [Accessed 26 Jul. 2024].

United Nations (2022). World Population Prospects 2022: Summary of Results. [online] Available at: https://www.un.org/development/desa/pd/sites/www.un.org.development.desa.pd/files/wpp2022_summary_of_results.pdf. [Accessed 25 Jul. 2024].

United States Environmental Protection Agency (2021). International Efforts on Wasted Food Recovery. [online] US EPA. Available at: https://www.epa.gov/international-cooperation/international-efforts-wasted-food-recovery. [Accessed 25 Jul. 2024].

United States Environmental Protection Agency (2024). Understanding Global Warming Potentials | US EPA. [online] US EPA. Available at: https://www.epa.gov/ghgemissions/understanding-global-warming-potentials [Accessed 25 Jul. 2024].

U.S. Soybean Export Council (2023). Conversion Table. [online] Available at: https://ussec.org/resources/conversion-table/. [Accessed 26 Jul. 2024].

Vandermeersch, T., Alvarenga, R.A.F., Ragaert, P. and Dewulf, J. (2014). Environmental sustainability assessment of food waste valorization options. Resources, Conservation and Recycling, 87, pp.57–64. doi:https://doi.org/10.1016/j.resconrec.2014.03.008.

Van Zanten, H.H.E., Herrero, M., Van Hal, O., Röös, E., Muller, A., Garnett, T., Gerber, P.J., Schader, C. and De Boer, I.J.M. (2018). Defining a land boundary for sustainable livestock consumption. Global Change Biology, 24(9), pp.4185–4194. doi:https://doi.org/10.1111/gcb.14321.

Wadhwa M., Bakshi M.P. and Makkar H.P. (2015). Waste to worth: fruit wastes and by-products as animal feed. CAB Reviews: Perspectives in Agriculture, Veterinary Science, Nutrition and Natural Resources, 10(031). doi:https://doi.org/10.1079/pavsnnr201510031.

Wang, K., Du, C., Guo, X., Xiong, B., Yang, L. and Zhao, X. (2024). Crop byproducts supplemented in livestock feeds reduced greenhouse gas emissions. Journal of environmental management, 355, pp.120469–120469. doi:https://doi.org/10.1016/j.jenvman.2024.120469.

Woods, J., Williams, A., Hughes, J.K., Black, M. and Murphy, R. (2010). Energy and the food system. Philosophical Transactions of the Royal Society B: Biological Sciences, [online] 365(1554), pp.2991–3006. doi:https://doi.org/10.1098/rstb.2010.0172.

World Animal Protection (2021). Results are in: Canadians are concerned about the harmful effects of industrial animal agriculture. [online] Available at: https://www.worldanimalprotection.ca/news/results-are-canadians-are-concerned-about-harmful-effects-industrial-animal-agriculture. [Accessed 25 Jul. 2024].

World Bank (2021). Moving towards sustainability: The Livestock Sector and the World Bank. [online] World Bank. Available at: https://www.worldbank.org/en/topic/agriculture/brief/moving-towards-sustainability-the-livestock-sector-and-the-world-bank. [Accessed 26 Jul. 2024].

World Food Program USA (2021). 8 Facts to Know About Food Waste and Hunger. [online] World Food Program USA. Available at: https://www.wfpusa.org/articles/8-facts-to-know-about-food-waste-and-hunger/ [Accessed: 25 July 2024].

World Health Organization (2021). WHO global air quality guidelines: particulate matter (PM2.5 and PM10), ozone, nitrogen dioxide, sulfur dioxide and carbon monoxide. [online] www.who.int. Available at: https://www.who.int/publications/i/item/9789240034228. [Accessed 25 Jul. 2024].

WRAP (2022). The impact of packaging and refrigeration on shelf life. [online] Banbury. Available at: https://www.wrap.ngo/sites/default/files/2022-02/The-impact-of-packaging-and-refrigeration-on-shelf-life.pdf [Accessed 25 Jul. 2024].

Xu, R., Kiarie, E.G., Yiannikouris, A., Sun, L. and Karrow, N.A. (2022). Nutritional impact of mycotoxins in food animal production and strategies for mitigation. Journal of Animal Science and Biotechnology, [online] 13(1). doi:https://doi.org/10.1186/s40104-022-00714-2.

Xu, X., Sharma, P., Shu, S., Lin, T.-S., Ciais, P., Tubiello, F.N., Smith, P., Campbell, N. and Jain, A.K. (2021). Global Greenhouse Gas Emissions from Animal-based Foods are Twice Those of plant-based Foods. Nature Food, [online] 2(9), pp.724–732. Available at: https://www.nature.com/articles/s43016-021-00358-x [Accessed 25 Jul. 2024].

Zhu, J., Luo, Z., Sun, T., Li, W., Zhou, W., Wang, X., Fei, X., Tong, H. and Yin, K. (2023). Cradle-to-grave emissions from food loss and waste represent half of total greenhouse gas emissions from food systems. Nature Food, 4(3), pp.247–256. doi:https://doi.org/10.1038/s43016-023-00710-3.

zu Ermgassen, E.K.H.J., Phalan, B., Green, R.E. and Balmford, A. (2016). Reducing the land use of EU pork production: where there’s swill, there’s a way. Food Policy, 58, pp.35–48. doi:https://doi.org/10.1016/j.foodpol.2015.11.001.