IMPACTS OF CARBON SEQUESTRATION ON LIFE CYCLE EMISSIONS IN MIDWESTERN USA BEEF FINISHING SYSTEMS

By

Paige L. Stanley

A THESIS

Submitted to
Michigan State University
in partial fulfillment of the requirements
for the degree of

Animal Science – Master of Science

2017

ABSTRACT

IMPACTS OF CARBON SEQUESTRATION ON LIFE CYCLE EMISSIONS IN MIDWESTERN USA BEEF FINISHING SYSTEMS

By

Paige L. Stanley

Beef cattle have been identified as the largest contributor to greenhouse gas (GHG) emissions from the livestock sector. Through life cycle analysis (LCA), studies have concluded that grass-fed beef production systems have a higher GHG intensity than feedlot-finished (FL) beef. However, these studies have only used one grazing management system, continuous grazing, to model the environmental impacts of grass-fed production. Adaptive multi-paddock (AMP) grazing is a management system with improved animal and forage productivity, as well as potential soil carbon (SOC) sequestration, compared with continuous grazing through high animal stocking on short grazing intervals with pasture recovery periods. To examine the impacts of AMP grazing and SOC sequestration on net GHG emissions, a comparative LCA was executed for two finishing systems in the Upper Midwest: AMP grazing and FL using, on-farm data from the Lake City AgBioResearch Center. Impact scope included GHG emissions from enteric CH_4, feed and mineral supplement production, manure, and on-farm energy use and transportation, as well the potential C sink arising from SOC sequestration. Across-farm SOC data showed a four-year C sequestration rate of 3.59 Mg C $ha^{=1}$ yr^{-1}. After including SOC into the GHG footprint, emissions from the AMP system were reduced from 9.62 to -7.92 kg CO_2-e kg CW^{-1}, while FL emissions remained at 7.02 kg CO_2-e kg CW^{-1}. This indicates that AMP grazing might offset emissions through soil C sequestration and therefore that the finishing stage may be a net C sink. More long-term research is needed to confirm soil C sequestration in other ecoregions and its potential impact on GHG mitigation in beef production system.

ACKNOWLEDGEMENTS

I am fortunate enough to have many people to thank for helping me throughout this process. First and foremost, I would like to thank my advisor, Dr. Jason Rowntree for responding to my cold-call email more than two years ago, albeit a few months delayed. If not for your appreciation of my passion for sustainable agriculture (which, I hope translated through that email) and willingness to accept a non-traditional graduate student into the Animal Science Department, I would not be where I am now. I would also like to thank Dr. Beede, who played a vital role in making it possible for me to join this program. I will forever be grateful for your kindness, understanding, and generosity. Both of you have made me into a better scientist and have inspired me beyond words. I can only hope that this work has made both of you proud and not regret the risk you took on me two years ago.

I would also like to thank my other committee members, Drs. DeLonge, Siegford, and Hamm. Dr. Marcia DeLonge, thank you so much for taking the time to provide your insight, wisdom and experience during my research. You were absolutely vital to my project, and I cannot thank you enough for the opportunities you provided. Dr. Siegford, thank you for always being willing to listen and being open minded about my project. I learned so much about animal welfare through you. The opportunities you provided during my time at MSU are so appreciated. Dr. Hamm, I sincerely appreciate your logistical and philosophical support and helping in getting this project off the ground. You truly are a trailblazer. It is only through standing on the shoulders of giants like all of you, that people like me are able to achieve success.

Without a doubt, my family deserves an award for all they have done for me during my master's research. To my mom, and my biggest fan: I cannot begin to find the words in my heart to tell you how much I appreciate you. You have supported my crazy ideas and demanding academic career from the second I started on this journey, even when it continues to put hundreds of miles between us. For listening to me vent, encouraging me when I thought there was no way I could do it, and reminding me that great things do not come without sacrifice, thank you. Thank you so much to my grandparents, who's support has been unwavering throughout this time. I am unworthy of such an amazing and encouraging family. Lastly, I would like to thank Zachary Yongue, who has constantly challenged me and made me a better version of myself. The little things (cooked dinners, late night glasses of wine, and laughs when I was at my brink) have not gone unnoticed. Thank you for your patience and for being my much, much more laid back and better half.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

KEY TO ABBREVIATIONS

ADG…Average daily gain

AMP…Adaptive multi-paddock grazing

C…Carbon

CEC…Cation exchange capacity

CH_4…Methane

CO_2…Carbon dioxide

CO_2-e…Carbon dioxide equivalent

CW…Carcass weight

CL…Clay loam soil

DDGs…Dried distillers grains with solubles

DMI…Dry matter intake

DM…Dry matter

ECS…Elemental combustion system

EF…Emission factor

FL…Feedlot

$Frac_{leach}$…Fraction of nitrogen leached

GHG…Greenhouse gas

GWP…Global warming potential

HMC…High moisture corn

IPCC…Intergovernmental Panel on Climate Change

ISO…International Organization for Standardization

Kg…Kilogram

LCA……………………………………………………Life cycle analysis

LW……………………………………………………..Live weight

MIG…………………………………………………....Management-intensive grazing

Mg…………………………………………………….Megagram or ton

MMt……………………………………………..……..Millions of metric tons

MSU……………………………………………………Michigan State University

N………………………………………………………Nitrogen

N-eq……………………………………………………Nitrogen equivalent

NE_g……………………………………………………Net energy for gain

NE_m……………………………………………………Net energy for maintenance

N_{ext}……………………………………………………..Nitrogen excreted

NH_3……………………………………………………Ammonia

NO……………………………………………………..Nitric oxide

NO_x……………………………………………………Nitrogen oxide species

NRC……………………………………………………National Research Council

N_2O……………………………………………………Nitrous oxide

O_2………………………………………………………Oxygen gas

P……………………………………………………….Phosphorus

P-eq……………………………………………..……...Phosphorus equivalent

PE………………………………………………………Potential evapotranspiration

S………………………………………………………..Sandy soil

SF_6……………………………………………………...Sulfur hexafluoride

SL………………………………………………………Sandy Loam soil

SOC………….………………………………….Soil organic carbon

SOM………….…………………………………....Soil organic matter

TMR………….……………………………….....Total mixed ration

VFA………….……………………………………Volatile fatty acid

WHC……….……………………………………Water holding capacity

Y_m……….……………………………………Enteric methane conversion factor

CHAPTER 1

LITERATURE REVIEW

1.1. Introduction

The purpose of this review is to survey and critically evaluate relevant literature about beef production life cycle analysis (LCA) and determine relative strengths and weaknesses of each model. Subsequently, emerging areas of research that may warrant inclusion into beef LCAs will be identified. By doing this, we aim to gain a clearer understanding of the environmental impacts of beef production in a variety of systems, and also to address the gaps in knowledge and science that need to be filled in the future. Overall, the goal of this review and assessment is to advance knowledge and aid in creating better understanding of opportunities for beef production systems to work in concert with nature and the environment to provide food for mankind.

In recent years, much attention has been called to the greenhouse gas (GHG) intensity of beef production. In 2006, Livestock's Long Shadow, a keystone report by the Food and Agriculture Organization (FAO et al., 2006), estimated that livestock are responsible for 18% of all GHG emissions, and cites livestock production as one of the top three most significant contributors to environmental degradation. Alternatively, and considerably less in comparison, the EPA estimated that in entirety, US agriculture contributed 8.3% of all domestic GHG emissions (EPA, 2016). While the methodology and validity of Livestock's Long Shadow's estimates have come under scrutiny (Pitesky et al., 2009), the report spawned much research in recent years to attempt overall system cost-benefit analysis of livestock, especially beef.

Despite its large contribution to GHG emissions and global warming, the livestock sector is an important source of income for more than 1.3 billion people, and provides food protein products for people across the globe, and food security in impoverished areas (Herrero et al., 2013). When considering cropland used to feed livestock, cattle production is a driver behind land-use on nearly 80% of all agricultural land (FAO, 2016).

To determine where research and potential GHG mitigation strategies should focus with respect to beef production, life cycle analyses (LCAs) can be used to assess the "cradle-to-gate" environmental impact of production systems (ISO, 2006). To date, these efforts for beef systems have yielded variable, inconsistent, and confusing results, and there is no consensus about the best environmental management practices.

1.2. Life cycle analysis (LCA) as a tool

Life cycle analyses are used to identify the cradle-to-gate environmental impacts associated with a product. For agriculture, and more specifically, livestock, impacts include airborne gas emissions from all stages of the animal's life and emissions associated with feed, fuel, and processes indirectly related to the animal during its life. These emissions are expressed in the common functional unit, kg CO_2-e kg carcass weight^{-1} , where: CO_2-e is the CO_2 equivalents representing the impacts of different GHGs relative to that of CO_2 incorporating their respective different global warming potentials (GWP). Global warming potential defines how much heat each specific gas can trap in the atmosphere over a period of time relative to CO_2, which has a GWP of 1 (EPA, 2016a). The gases commonly considered include CO_2, CH_4, and N_2O, with GWPs of 1, 34, and 298, respectively (EPA, 2016a).

The LCA model outcomes assess system strengths and weaknesses from an environmental perspective. Principles and frameworks have been developed by the International Organization for Standardization and aid in the definition and scope for designing a LCA for any given system (ISO, 2006).

While LCAs are a useful tool for both identification and comparison of environmental impacts, their application to a complex industry such as beef production presents challenges. Variability in management (IPCC, 2006; Pitesky et al., 2009), animal genetics (IPCC, 2006; Pitesky et al., 2009), regionality (Pitesky et al., 2009; Stackhouse-Lawson et al., 2012), and predictive equations (IPCC, 2006) can create significant and impactful differences in outcomes (Pitesky et al., 2009). Because of this, Stackhouse et al. (2012) and Halberg (2005) indicate that LCAs should be conducted and applied in a regional context, specific to management and climate of the respective area. Additionally, other environmental assessment tools have been utilized, including ecological footprint analysis (EFA), land-based versus product-based accounting, simulation programs (i.e., DAYCENT), and on-farm tools such as green accounts, ecopoints, and the DIALECTE method (Halberg et al., 2005).

While IPCC-compliant livestock LCAs contain the same common “key categories,” such as emissions from enteric fermentation, feed production, and manure management, methods of accounting and addition of other non-key emissions categories (i.e., on-farm emissions) vary significantly among studies. Limited data availability has also been influential in producing differing results. For example, in lieu of region-specific or on-farm data that are not often available, IPCC Tier 1 accounting methods are commonly used, which employ default values that can be applied based on previous supporting research. In some cases, however, such as the IPCC default enteric methane conversion factor Y_m, these default values contribute significant uncertainty to the model outcome (IPCC, 2006; Rowntree et al., 2016).

Although LCAs are imperfect in both accounting methods and identification of all environmental impacts, they are currently the most common tool for delineating overall environmental impacts from beef production.

1.3. Comparison and selection criteria

While it is clear that agriculture plays a significant role in GHG emissions and climate change, results from studies attempting to quantify these emissions from beef finishing systems are variable. The variation surrounding these results stem from not only different quantification methods and variations in location and management practices, but also because many studies fail to recognize relevant interactions involving carbon (C) flow within the ecosystem as a whole.

Several life cycle analyses (LCA) have been conducted on beef production models in order to evaluate environmental and production tradeoffs associated with different management practices **(ISO, 2006)**. Areas of LCA examination have included different stages of production, differing levels of intensification, combination dairy-beef production systems, and origin of feed products, amongst others (Vries et al., 2015). However, even LCAs that evaluate similar production systems may have very different outcomes based on internal assumptions and methodological anomalies (Vries et al., 2015).

Therefore, the focus of this effort was to review the current beef LCA literature, highlight how key assumption and methodological differences contribute to differing results among different beef production LCAs, and identify major gaps that may present significant uncertainty in our overall conclusions regarding beef production systems.

One major area of comparison investigated by beef production LCAs is finishing strategy. As of 2017, 96% of cattle in the U.S. were finished in feedlots and 4% were broadly "grass finished" (Cheung, 2017; USDA, 2014). Because differences in finishing system also heavily impact calculated emissions from feed production, second only to enteric methane (CH_4) as the largest contributor to net GHG emissions from beef cattle, this selection category was an important identifier (Herrero et al., 2016; Lupo et al., 2013; Pelletier et al., 2010). Additionally, feed production is the primary energy consumer in cradle-to-gate beef cattle life cycles (Pelletier et al., 2010; Pimentel, 2008).

Based on the criteria of comparing feedlot to grass-fed beef cattle finishing systems and reporting cradle-to-gate emissions, eight LCAs were selected for comparative evaluation (Capper, 2012; Cardoso et al., 2016; Casey and Holden, 2006; Dudley et al., 2014; Lupo et al., 2013; Pelletier et al., 2010; Stackhouse-Lawson et al., 2012; Stewart et al., 2009). These LCAs represented the Upper Midwest (Pelletier et al., 2010), Northern Great Plains (Lupo et al., 2013), Canada (Stewart et al., 2009), Ireland (Casey and Holden, 2006), California (Stackhouse-Lawson et al., 2012), Brazil (Cardoso et al., 2016), and two non-region specific scenarios (Capper, 2012; Dudley et al., 2014).

1.4. Overall consensus of literature

1.4.1. LCA data consensus

It is indisputable that livestock contribute significantly to GHG emissions (FAO, 2012). Overall, it is estimated that livestock are responsible for 14.5% of global GHG emissions, with cattle alone contributing about 65% of that (Gerber, 2013). Per unit of beef produced, LCAs estimate cradle-to-gate emissions ranging from 14.50 to 29.40 kg CO_2-e per kg carcass weight and 24.30 to 34.9 kg CO_2-e per kg carcass weight for grain and grass-fed systems, respectively (Capper, 2012; Pelletier et al., 2010). On average, LCAs reported that enteric methane is the largest system emission, contributing an estimated 40 to 70% of overall emissions from beef production (Gerber, 2013; Herrero et al., 2013; Herrero et al., 2016; Lupo et al., 2013; Pelletier et al., 2010; Powers, 2014; Ripple et al., 2014; Stackhouse-Lawson et al., 2012; Stewart et al., 2009; Wang et al., 2015). Feed production is the second largest emissions source category; however, the contribution varies largely depending upon management and production system.

Feed production is estimated to be responsible for between 0 to 45% of overall emissions, depending upon feed input and fertilization practices (Gerber, 2013). Methane and nitrous oxide (N_2O) emissions arising from manure management represent between 10 to 20% of emissions, and emissions associated with on-farm energy use contribute less than 5% (FAO et al., 2006; Herrero et al., 2016; Lupo et al., 2013).

1.4.2. GHG footprint literature ranges

Unanimously, all LCAs examined in this review concluded that grass-fed cattle have a greater cradle-to-gate GHG footprint than grain-fed cattle, largely due to higher enteric CH_4 production in the grass-fed production system. The LCAs investigated showed average emissions of 23.28 and 30.80 kg CO_2-e per kg of carcass weight for conventional feedlot finished and grass-fed systems, respectively. Grass-fed cattle emit more enteric CH_4, in general, than grain-fed cattle for two reasons: they consume lower energy, higher fiber diets and require more time to reach market weight as a consequence. Diet effect on enteric CH_4 production is discussed in a later section. More influential than diet is overall time to finish. Conventional feedlots commonly finish steers in 18 months whereas grass-finished steers may take 6-10 additional months to reach market weight. However, this is highly dependent on animal genetics and management, with finishing times ranging from 18-20 months on high quality pastures (Rowntree et al., 2016) to 28 months on low quality pastures (Capper, 2012; Pelletier et al., 2010). Overall, models indicate that these factors contribute to greater CH_4 production in grass-fed beef production systems, and therefore greater overall GHG footprints than grain-finishing systems (Powers, 2014).

1.4.3. Methane (CH_4) emission ranges and sources

Enteric CH_4 production is quite easily the most substantial GHG contributor from beef production, representing an estimated average of 40 to 80% of the total GHG footprint. While manure CH_4 contributes less than 3% of total emissions (Broucek, 2014; Powers, 2014), cow-calf and pasture finishing contribute more enteric CH_4 due to greater utilization of more fibrous feedstuffs, longer lifetime, and lower finishing weight compared with feedlot production (Capper, 2012; Cardoso et al., 2016; Lupo et al., 2013; McAllister et al., 1996; Pelletier et al., 2010; Stackhouse-Lawson et al., 2012).

Enteric CH_4 emissions are often modeled using IPCC Tier 2 methodologies. These equations represent enteric CH_4 as a function of gross energy (determined by bomb calorimetry) of feeds, animal characteristics, and a default CH_4 conversion factor (Y_m). Some factors that are not included in this calculation that potentially can contribute to significant differences in estimates of enteric methane production include breed, effects of environmental temperature and associated depression in digestibility, forage quality, diet composition, and variability in rumen microbiota (IPCC, 2006; McAllister et al., 1996; Pitesky et al., 2009). Additionally, the validity of default values for IPCC Y_m has come under scrutiny. The default Y_m recommended by the IPCC is 3.0% and 6.5% for feedlot and grazing cattle, respectively. Studies suggest that Y_m can overestimate enteric CH_4 emissions for forage based and grass-fed beef production systems as much as 20%, while concurrently underestimating CH_4 production from feedlot cattle by up to 35% (Cardoso et al., 2016; Kebreab et al., 2008; Rowntree et al., 2016; Stackhouse-Lawson et al., 2012).

Dietary carbohydrate sources (concentrate versus forages) differ mainly in their extent of digestibility (concentrates are potentially considerably more digestible than forages depending upon plant maturity at harvest) and in the different profile end products of ruminal fermentation that are made available to the animal post-absorption for metabolizable energy utilization. Ruminal metabolism of diets high in concentrates differs dramatically from that of mainly forage-based diets, and thus the amount of enteric CH_4 produced per unit of dry feed mass. Thus, differences in estimates of Y_m are based on rumen metabolism of the different respective feed types. When ruminants are fed high concentrate diets, ruminal production of the volatile fatty acid (VFA) proprionate (three-carbon molecule) is favored. Because microbial fermentation of carbohydrate feed to produce proprionate conserves more available carbons, less CH_4 is produced, and more digestible and metabolizable energy is conserved compared with high-forage diets (more fibrous) in which ruminal production of the VFA acetate is favored. When fibrous carbohydrates are fermented, the predominate VFA is acetate (two-carbon molecule), and some CO_2 is produced and converted to CH_4 in the rumen. The carbons converted to CO_2 and CH_4 are "lost" in terms of supplying metabolizable energy for animal productivity (Church, 1993; Johnson and Johnson, 1995). This typically lengthens the time to finished weight for forage-fed compared with concentrate-fed beef cattle.

Across studies, Stewart (2009) indicated that enteric CH_4 emissions accounted for 60 to 70% of the total beef production GHG footprint. Lupo et al. (2013), Casey and Holden (2006), Stackhouse et al. (2012), and Capper (2012) reported similar contributions, whereas lower proportions (~40%) were reported by Pelletier (2010) and Dudley (2014), and higher contributions (~80%) were reported by Cardoso (2016).

Studies suggest enteric CH_4 mitigation opportunities through the use of higher quality (more digestible) forages and pastures, including incorporation of legumes and use of feed additives that may directly reduce ruminal CH_4 production such as ionophores and fats, thus increasing efficiency of feed utilization to produce beef (Capper, 2012; Johnson and Johnson, 1995; Stewart et al., 2009).

1.4.4. Nitrous oxide (N_2O) emission ranges and sources

In beef production systems, N_2O emissions arise from two primary sources: feed crop fertilization and manure management. From either source, nitrous oxide is produced through both direct and indirect pathways. Direct N_2O is the result of nitrification and denitrification of N in livestock manure and in land-applied synthetic fertilizer. Nitrous oxide emerges indirectly from two pathways: volatilization of nitrogen (N) from manure or fertilizer to ammonia (NH_3) and nitrogen oxide species (NO_x), which have downstream effects on soil and waterways, and leached manure-N from manure or fertilizer into runoff water (EPA, 2016b).

Nitrous oxide (N_2O) from feed crop production is the second most predominant emission category for beef production quantitatively. Mainly, this is attributable to large amounts of N_2O from fertilizer application. Because N_2O has a GWP 265 to 298 times greater than that of CO_2 and can last in the environment for over 100 years, it is considered a potent and impactful GHG. Camargo (2013), using the Farm Energy Analysis Tool (FEAT), indicated that across several different crop species, N fertilizer had the highest impact (44%) on overall footprint, which was also supported by Stewart (2009) and Stackhouse (2012). Additionally, corn grain and corn silage had the largest GHG impact among a number of crop species examined (Gustavo et al., 2013). In this way, N_2O emissions arising from beef production can greatly alter the overall CO_2-e footprint of a beef production system, and therefore must be accounted for properly.

Nitrous oxide also represents a significant portion of manure emissions, which is the third leading emissions category in beef production. In the feedlot during manure handling, organic N and NH_3 are transformed to N_2O first by aerobic, and then anaerobic processes. Chemically,

NO_x is oxidized to NO, with N_2O produced as a byproduct. Therefore, direct N_2O emissions from cattle manure is most likely to occur when it is handled as a solid. While much of beef cattle manure in the U.S. is handled as a solid, either on pasture or in a dry lot, some larger feedlot operations practice liquid manure management (EPA, 2016b; Pitesky et al., 2009). In total, these direct emissions account for only an estimated 5% of total N_2O emissions in 2013 (EPA, 2013). Once manure is spread onto cropland, a common fertilization method, indirect N_2O emissions predominate via volatilization and leaching (EPA, 2016b). From both direct and indirect sources, N_2O from domestic beef cattle manure management accounted for an estimated 7.8 MMT CO_2-e, more than any other livestock category (EPA, 2016b).

Unfortunately, while N_2O emissions included in beef LCAs vary widely based on cattle diet, climate, soil health, N application rate, and type of crop, many researchers use default emissions factors because of a lack of regional data (IPCC, 2006). This contributes to significant uncertainty, as the amount of N excreted by the animal depends on N content of the diet, and the extent and rate of N runoff and leaching affected by regional moisture levels, and the level of aeration that can largely affect the overall emission.

Management decisions such as high levels of fertilizer application and irrigation can increase the proportion of N_2O in the overall beef CO_2-e footprint, representing in some cases 37.5 to 65% of the total (Stewart et al., 2009). Additionally, Stewart (2009) reported that pasture fertilizer reduction resulted in the largest decrease (30%) in modelled overall emissions from beef production. This conclusion was supported by Casey (2006), who indicated significant correlations between fertilizer application rate on feedstuffs and total GHG emissions, primarily because of an increased N_2O contribution. These results estimated up to a 26% decrease in N_2O emissions when using "organic" cropping methods versus conventional due to restrictions on use of fertilizer in the organic system. However, this was accompanied by a slight increase in manure-derived N_2O emissions due to aerobic manure conditions in the organic system (Casey and Holden, 2006). Comparing grass-fed production to conventional beef production with feedlot finishing, Lupo et al., (2013) demonstrated that manure N_2O emissions were nearly six-fold greater in the conventional system, with more than half of the total arising from the feedlot finishing stage. When examining contribution to indirect N_2O emissions via NH_3, Stackhouse et al., (2012) indicated that the feedlot phase contributed approximately 30% more than the cow-calf (grazing) phase due to diet and manure management differences.

In some studies (Casey and Holden, 2006; Lupo et al., 2013; Pelletier et al., 2010), data for N_2O were not explicitly represented, making it impossible to determine the total N_2O impact.

1.4.5. Other emissions sources

Carbon dioxide is produced from fossil fuel use during transportation of livestock and feed products, production of fertilizer and mineral supplements, and on-farm energy use. Although these CO_2 emissions represent a relatively small proportion of overall emissions, they must be accounted for to accurately estimate the full environmental impact of beef production in life cycle analysis (Pitesky et al., 2009). Animal respiration also results in CO_2 production, however it is not included in LCA calculations because it is assumed to be in balance and equalized with plant C assimilation (IPCC, 2006).

Generally, the total CO_2 contribution to GHG footprint in beef systems is less than 5% (Lupo et al., 2013; Pelletier et al., 2010; Stackhouse-Lawson et al., 2012). Because CO_2 emissions vary widely by farm practices (i.e., irrigation, type of vehicles used) data are often limited, and occasionally many key CO_2 contributing processes are excluded from LCA accounting (Dudley et al., 2014).

1.4.6. Other impacts: land use change, eutrophication, acidification and soil erosion

In addition to GHG emissions, there are other widely accepted environmental impacts associated with beef production, including land use change, eutrophication and diminished water quality, terrestrial acidification and soil erosion. The current lack of data and complexity in correctly modeling contributing mechanisms are the primary reasons that many of these impacts are not included in LCA accounting. However, many have indicated that exclusion of these factors contributes to significant uncertainty in conclusions drawn from LCAs, and may generate much more robust and conclusive results if they are included (Dudley et al., 2014; Janzen, 2011; Lupo et al., 2013; Pelletier et al., 2010).

Beef production remains a global driver of land use change (FAO, 2012). Accordingly, deforestation related emissions can drastically alter the GHG footprint of beef production and represent the highest degree of uncertainty in beef production emissions (Dudley et al., 2014; Rivera-Ferre et al., 2016). As indicated by Dudley (2014), including land-use change (LUC) in a corn-ethanol system based on the government mandated increase in ethanol demand can result in an 500% increase in the GHG intensity of grain-based beef production, due to both conversion of alternative land-uses into cropland, and through the high GHG intensity of ethanol products. Similarly, in arid climates, desertification due to overgrazing is an important consequence of livestock production and management. Because arid ecosystems account for more than 45% of global land surface, soil degradation through overgrazing and subsequent C loss to the atmosphere may contribute to significant C emissions; however, it is difficult to accurately estimate how much (Pitesky et al., 2009).

In addition to agriculturally derived GHG emissions, soil erosion is a further negative externality. It is estimated that 15 million acres of "prime farmland" were lost between 1982 and

2012 primarily due to soil erosion, even though erosion declined by 44% over the same timeframe (USDA, 2015). Because 33% of all arable cropland is used to produce feed for cattle, it is inherent that beef production is responsible for a portion of soil erosion from cultivated soils (FAO, 2012; Pitesky et al., 2009). Therefore, these losses should be accounted for in LCAs representing beef production as a true cost.

Due to excess nutrients (N) and phosphorus (P) introduced into agroecosystems via manure management and feed production as well as associated soil loss, beef production also contributes to eutrophication in waterways and water bodies and can have deleterious effects on human and ecosystem health. Excess N and P entering waterbodies can result directly from manure, either on pasture or applied to cropland, or indirectly from fertilization of feed crops (Lupo et al., 2013; Pelletier et al., 2010). Because corn is the most aggressively fertilized crop in the U.S. and is also the dominant feed component in grain-based feedlot beef production systems, beef production can sometimes give rise to significant algal blooms resulting from excess P coupled with an O_2 deficit (Lupo et al., 2013; Pelletier et al., 2010; Pitesky et al., 2009; W. Teague et al., 2016). Furthermore, N and P-loaded drainage water from largely corn-producing Midwestern states has created a hypoxic zone in the Gulf of Mexico now referred to as the "Dead Zone," which is a site of extensive fish and marine life losses (Rabalais et al., 2002; W. Teague et al., 2016). Predictions suggest that this hypoxic zone will grow to 24,901 km^2 by 2017, roughly equal to the land area of Vermont (Turner, 2017). Eutrophication values used vary by study based on assumptions (primarily N-leaching factors), but have ranged from 2.48 g P-eq and 79.4g N-eq per kg of carcass weight for potential freshwater and marine eutrophication in the Northern Great Plains to 189g PO_4^{3-}eq in Iowa, respectively (Lupo et al., 2013; Pelletier et al., 2010). These studies also suggest that grass-fed beef production may reduce marine and freshwater eutrophication by 2-17% in non-fertilized pastures due to overall lower N inputs, however ammonia volatilization from manure also leads to terrestrial soil acidification, which is the reduction in soil pH resulting from the atmosphere-soil transfer of environmental pollutants. Because generally less N is contained in grass-fed systems, this type of management may reduce potential terrestrial acidification by approximately 9% (Lupo et al., 2013).

1.5. Differences among LCAs

Differences in results among the beef production LCAs in the literature are attributable to three main areas: regionality, pasture and manure management, and accounting methods. In several studies examined, discrepancy in estimated N_2O emissions were found between grass and feedlot-finishing methods. Because of the very high GWP associated with N_2O (nearly 300 times that of CO_2), it is highly impactful on overall GHG footprint when expressed on a CO_2-eq basis. Three studies (Cardoso et al., 2016; Lupo et al., 2013; Stewart et al., 2009) concluded that grass-finishing resulted in lower overall N_2O emissions; whereas two studies reported the contrary (Capper, 2012; Pelletier et al., 2010). Outcomes suggesting lower N_2O from grass-finishing were the result of lower N inputs on pasture versus cropland used for production of concentrate feeds, and lower manure-contributing N volatilization and leaching resulting from the forage-based diets. In some cases, N_2O emissions were more than two times greater for feedlot-finishing than grass-finishing strategies (Cardoso et al., 2016). The alternative, according to Pelletier (2010), was attributable to a low forage utilization rate, which ultimately required a large area of managed pasture. However, the high leaching factor and pasture fertilization assumptions, in combination with double counting of manure leaching (once in the feedlot and again for land application) are likely to have contributed to this conclusion (Lupo et al., 2013). Similarly, Capper (2012) concluded that N_2O emissions in the grass-fed scenario were 54%

greater than the feedlot-finishing scenario; however, there is great difficulty in assessing why, based on their stated study assumptions. In general, worth noting is the 50 to 100% uncertainty associated with the utilization of IPCC default N_2O emission factors, which certainly plays a significant role in strongly contrasting results (IPCC, 2006).

Regionality and climate differences also play a part in discrepancies among LCAs. In the tropical climates such as Brazil, forages grow year-round and dramatically reduce the need for off-farm forage inputs in grass-finishing systems (Cardoso et al., 2016). Alternatively, in temperate climates such as the Upper Midwest, the long winter season requires feed to be mechanically harvested during the growing season or imported from other regions for some part of the year, which may contribute to a higher GHG impact (Pelletier et al., 2010). These discrepancies are exacerbated further by temporal and seasonal weather differences, such as precipitation. This was stated by Cardoso (2016), who suggested that N leaching can vary significantly not only between wet and dry regions in the Cerrado region of Brazil, but also between fecal and urine excreta, although a single emission factor is suggested by IPCC (2006).

Perhaps even more responsible for reported differences is choice of accounting method between studies. Some use IPCC Tier I methods due to lack of regional or specific data, as was the case with enteric CH_4 estimates (Dudley et al., 2014), manure N_2O and CH_4 emissions (Capper, 2012; Dudley et al., 2014; Lupo et al., 2013; Pelletier et al., 2010), and field level feed production emissions (Capper, 2012; Pelletier et al., 2010). Furthermore, some studies opted to use IPCC default emissions factors (EFs) where others substituted site-specific data. For example, although Lupo (2013) utilized IPCC Tier 1 accounting for manure emissions, the fraction of N leached ($Frac_{leach}$) was calculated using regional potential evapotranspiration data. Additionally, several different modeling tools and databases were utilized to compile and calculate emissions, including the Integrated Farm System Model (Stackhouse-Lawson et al., 2012), EcoInvent (Lupo et al., 2013), SimaPro (Lupo et al., 2013; Pelletier et al., 2010) and @Risk (Dudley et al., 2014) as well as a variety of literature.

1.6. Missing pieces contributing to uncertainty

1.6.1. IPCC modeling limitations

There are limitations to the IPCC guidelines with respect to the real-life complexity of agroecosystems, especially for livestock production systems. For example, IPCC does not set specific guidelines for estimating emissions from all processes, such as feed mineral production (Lupo et al., 2013). Additionally, the influence of feed properties (i.e., digestible energy, energy for maintenance and gain) on Y_m, and their impact on overall CH_4 production is largely absent from IPCC calculations (IPCC, 2006). Moreover, literature suggests that SOC can play a vital role in nutrient cycling (Ontl, 2012). Because IPCC calculations do not consider the effects of ranging SOM on N leaching, which is further influenced by soil erosion and water infiltration, it likely does not accurately depict the impacts of beef production on N cycling.

1.6.2. GHG mitigation and ecosystem services through improved grazing

Arguably, the lack of representation of diverse grazing system management in LCAs is an area of major concern. All grass-fed production systems included in the LCAs thus far published were largely characteristic of a single type of grazing- extensive continuous grazing. Grazing management became commonplace with “rational grazing” and “holistic planned grazing” in the mid to late 20th century (Savory, 1998). Currently, grass-fed beef production systems operate under a variety of different management strategies, such as mob grazing (Chiavegato et al., 2015a), rotational grazing, and management intensive grazing (MIG) (UGA, 2015) which is also referred to as adaptive multi-paddock grazing (AMP), or various integrations of these.

Besides food-provisioning, recent literature has suggested that improved grazing management may contribute other societal benefits, such as improved C cycling, reduced competition for arable cropland, increased biodiversity, and reduced energy use (Herrero et al., 2013; Rivera-Ferre et al., 2016; Vries et al., 2015). Particularly, adaptive multi-paddock (AMP) grazing, can result in a myriad of environmental benefits when compared with continuous grazing (Conant, 2010; W. Teague et al., 2016; W. R. Teague et al., 2011). Because continuously grazed cattle often exhibit chronic, intensive selection of specific species within a pasture, species composition is negatively impacted which can further reduce productivity by continuously removing photosynthetic leaf area (Briske et al., 2008). This is largely defined as overgrazing, which exposes the pasture to extensive soil erosion, nutrient runoff, reduced SOC, depletion of root biomass, and reduction of aboveground biomass productivity, even leading to desertification in arid environments (Rayburn, 2000; W. Teague et al., 2016; W. R. Teague et al., 2011; Wang et al., 2015). For these reasons, it is widely accepted that continuous excessive grazing can be detrimental to plant communities (Conant, 2010). However, unilateral use of continuous grazing to model “grass-fed” beef production in LCAs largely ignores the differing environmental impacts that may arise from AMP grazing.

Adaptive multi-paddock grazing is different from continuous grazing in several aspects. This grazing strategy involves high animal stocking on adaptive short-duration grazing intervals where recovery periods are implemented for forage re-growth. These characteristics allow the pastures to be grazed more uniformly due to less species selection resulting from high stocking densities, which in turn prevents invasive, woody, and lignified species with poor palatability and digestibility by cattle from outcompeting high-quality grasses (G. Oates, Jackson, R., 2015; W. Teague et al., 2016; W. R. Teague et al., 2011; UGA, 2015). By ensuring that enough leaf area is left on forages to photosynthesize, rotation decisions can be made on-farm to prevent overgrazing and ensure pasture rest and recovery (W. Teague et al., 2016; W. R. Teague et al., 2011; UGA, 2015). Further, reintroducing cattle into a given paddock before forages reach a reproductive phase (during which lignin is added to the cell wall) can significantly improve digestibility and reduce concomitant enteric CH_4 production (L. G. Oates et al., 2011; Stewart et al., 2009). In addition to overgrazing prevention and improved animal performance, AMP grazing has been correlated with improved soil C sequestration, with the reported capability of sequestering between 0.33 to 1.76 Mg-C ha^{-1} yr^{-1} (Franzluebbers, 2010; McSherry and Ritchie, 2013; Wang et al., 2015). Considering that soils are the greatest land-based C reservoir (with approximately 2,300 Pg of SOM globally), C sequestration represents a considerable strategy for GHG mitigation in beef production systems and elsewhere (Machmuller et al., 2015). According to Silveira (2015), "Reports indicate an increase (or loss) of only 1% of the soil C in the top 4 inches of grazing-land soils is equivalent to the total C emissions from all U.S. cropland agriculture."

Managed grazing can also provide regulating and supporting ecosystem services through soil organic matter (SOM) (Bruce et al., 1999; Hancock, 2016; Machmuller et al., 2015). Because SOM is approximately 58% SOC, increasing the rate of soil C sequestration plays a vital role in overall soil health and vitality (Hancock, 2016; Machmuller et al., 2015; NRCS, 2009; W. R. Teague et al., 2011). Soil that is high in SOM has increased microbial biodiversity, increased water-holding capacity, and improved drought tolerance (Bruce et al., 1999; Machmuller et al., 2015). Higher SOM contents in the soil results in greater cation exchange capacities (CEC), which defines nutrient availability for plant uptake and reduces nutrient leaching and runoff (Brown, 2016; Kaiser et al., 2008). Soil organic matter also directly impacts the water-holding capacity (WHC) of the soil; every 1% increase in SOM can hold approximately the equivalent of an additional acre-inch of water, or 16,500 gallons (Gould, 2015; Hancock, 2016). Increased WHC directly impacts the resiliency of the system against drought and mitigates runoff, nutrient loss and soil erosion during heavy rainfalls (Conant, 2010; Russelle et al., 2007). One study reported CEC and WHC increases of 95 and 34%, respectively, after five-year conversion of cropland to intensively grazed pastures, indicating potential for significant soil quality improvements (Machmuller et al., 2015).

The potential of AMP grazing as a GHG mitigation strategy through soil C sequestration is a growing body of study and knowledge. Compared with continuously grazed pastures, Conant (2003) indicated that pastures in the Southeast U.S. converted to AMP grazing for at least three years had 22% greater total soil C, with sequestration rates averaging 0.41 Mg C ha-^{1} yr^{-1}. Similarly, sequestration rates of 0.60 Mg C ha-^{1} yr^{-1} in the Northern Great Plains (Liebig et al., 2010) and 0.44 Mg C ha-^{1} yr^{-1} in the Southern Great Plains (Wang et al., 2015) under AMP grazing have been reported. However, significantly greater sequestration rates have been reported for AMP-managed pastures previously converted from other land uses, averaging 8.0 Mg C ha^{-1} yr^{-1} after five years of conversion from cropland in Georgia (Machmuller et al., 2015). When comparing different grazing management transitions, (Wang et al., 2015) reported that conversion from light-continuous grazing and heavy-continuous grazing to AMP grazing generated 0.660 and 3.530 Mg C ha-^{1} yr^{-1} sequestered over 10 years and 0.330 and 1.765 Mg C ha-^{1} yr$^{-1,}$ over 20 years, respectively.

Overall, grassland C sequestration literature indicates that intensively managed grazing systems may serve to mitigate GHG emissions from beef production. Comparing grass to feedlot finishing, Lupo (2013) and Pelletier (2010) did not did not include C sequestration in their respective LCA boundaries, however sensitivity analyses were conducted with respect to GHG mitigation response to C sequestration. With a C sequestration rate of 0.41 Mg C ha^{-1} yr^{-1}, Lupo (2013) illustrated that cradle-to-gate grass-fed emissions were reduced by 24%, but ultimately still had a larger GHG footprint that feedlot-finished beef. On the contrary, Pelletier (2010) indicated that with C sequestration rates of 0.12 Mg C ha^{-1} yr^{-1} during the cow-calf phase and 0.4 Mg C ha^{-1} yr^{-1} during the grass-finishing phase, grass-finished beef would be 15% less GHG intensive than feedlot-finished beef. Some studies even suggest that AMP grazing systems may be an overall CO_2-e sink (after including animal emissions), with values ranging from -0.310 to -3.180 Mg C ha^{-1} yr^{-1} on a land basis (Liebig et al., 2010; Wang et al., 2015) and -26 to -145 kg CO_2-e per kg of weight gain on an animal production basis (Liebig et al., 2010); negative values presenting actual C sequestration in soils. Across grazing management systems in Europe, Soussana et al. (2007) reported average net sink of -0.880 kg CO_2-e m^{-2} $year^{-1}$ (-240 g C m^{-2} $year^{-1}$). Lastly, Machmuller (2015) demonstrated that rapid increases in soil C following crop-pasture conversion yielded an overall net GHG sink of -1.6 Mg CO_2-e ha^{-1} yr^{-1} in an intensively grazed dairy system. In addition to C sequestration, improved exchange of CH_4 with methanotropic bacteria within the soil can provide a minor CH_4-CO_2-e sink in some grazing systems (Chiavegato et al., 2015a; Jones et al., 2005; Liebig et al., 2010; Soussana et al., 2007). These studies establish a connection between grazing management, C sequestration, water conservation, and a systems-perspective net GHG impact.

Despite the mounting evidence in support of soil C sequestration, little is known about the potential for these rates to be maintained over time. Some have found that C can be sequestered for many years in lower soil depths, whereas others argue that there is a C saturation limit of soils across depths. In a study by Soussana et al. (2007) that contrasted full GHG accounting of soils across Europe, results did not support the common literature assumption of C sink saturation. Instead, results showed that even unmanaged grasslands tended to sequester C for many years, although location and management practices of managed grasslands varied in their sequestration rates and storage amounts.

Others indicate diminishing rates of SOC sequestration over time, but have not necessarily observed a saturation limit. Franzeluebbers (2010) commented that after 25 years of pasture establishment, approximately 80% of the maximum C storage will be reached. Similarly, Machmuller (2015) found that rates can decline at about 6.5 years following pasture conversion from cropland, although neither study argued for a C saturation limit. Instead, they theorize that rates decline as C takes longer to reach lower soil depths (Franzluebbers, 2010; Machmuller et al., 2015). At minimum, the potential for grassland C sequestration to act as a short-term GHG mitigation for beef production must be carefully and critically evaluated.

The vast majority of beef LCAs to-date have operated under the assumption of soil C equilibrium (zero net C sequestration). Therefore, much of the potential GHG mitigation via C sequestration in grass-based beef production system is overlooked, which consequently misrepresents overall net GHG impacts between grass and feedlot finishing systems. To better account for overall system GHG flux, and therefore provide more robust conclusions, a regional, systems-based approach will be crucial in the future (Janzen, 2011).

Animal performance characteristics are also improved under AMP grazing compared with continuous grazing systems, which contribute to lower GHG impact. In continuously grazed systems, reduced forage quality can contribute to a significantly longer finishing time (greater than 200 d) compared with what is possible in feedlot-finishing systems (Pelletier et al., 2010). However, Cardoso (2016) demonstrated that, compared with continuously grazed pastures in Brazil, AMP grazing on improved pastures reduced overall land area needed by two-thirds. Additionally, enteric CH_4 emissions were reduced by 54%, while only increasing N_2O emissions by 1.08 kg CO_2-e per kg of carcass weight. In contrast, N_2O emissions were increased by 5 kg CO_2-e per kg of carcass weight in the feedlot finishing system compared with the continuous grazing system (Cardoso et al., 2016). Overall, the AMP grazing scenario showed a 50% decrease in GHG footprint compared with the continuously grazed system, resulting in an overall equal footprint to the feedlot-finishing system. These benefits were most likely the result of increased productivity (and a concomitant reduction in finishing time), as indicated by Oates (2011), who demonstrated significant increases in both forage quality and quantity under AMP grazing versus continuous grazing (L. G. Oates et al., 2011).

As human population growth continues to increase in the future, a reduction in competition for arable cropland in livestock production may be considered an ecosystem service and an important indicator of sustainability (Machmuller et al., 2015). If current meat consumption and production patterns continue, 6% more arable cropland and 33% more cattle (1.39 to 1.85 billion cattle) will be needed, and agricultural GHG emissions will increase 27% by the year 2050 (Schader et al., 2015). Considering that cattle, at present, are the end-users of one-third of global arable grain production and contribute significantly to agriculture GHG footprint, there are, and will continue to be, significant trade-offs for food production. Schader et al. (2015) indicated that if livestock feed components that compete with direct human-food crop production (food-competing feedstuffs) were reduced to 0%, benefits would include a 22% reduction in use of arable cropland, a 22% reduction in N-surplus and a 5% reduction in overall agriculture GHG footprint. Overall, ruminant numbers would only decrease marginally, if grassland-based production was employed versus cropland for concentrate finishing. Despite the greater GHG footprint of grass-fed beef production systems currently indicated by the literature (Capper, 2012; Lupo et al., 2013; Pelletier et al., 2010; Stackhouse-Lawson et al., 2012), benefits of grass-land based beef production due to reduced use of arable cropland may outweigh the productivity losses (Schader et al., 2015). West et al. (2014) reported that this reduced competition for arable cropland through grass-fed beef production could provide enough land to grow food for an additional four billion people.

1.7. Areas for improvement

This review identified three areas of improvement for future beef production LCA research: 1) improved assumption transparency, site-specificity, and consistency; 2) inclusion of alternative ecosystem services as relevant impact categories; and, 3) expanding grass-fed beef models to reflect different management and more accurately depict environmental impacts, especially as related to the great potential for soil C sequestration in properly managed systems.

Above, we provided relevant literature that indicates many other environmental impacts associated with beef production aside from GHG footprint. These include soil erosion, desertification, water-holding capacity, cation-exchange capacity, and competition for arable land. While these impact categories cannot all be reflected on a CO_2-e basis, they too are indicative of a system's sustainability and should be addressed in future research.

Secondly, unclear assumptions of LCAs make it difficult to identify how results materialized. Furthermore, differing assumptions, regionality and lack of consistency make it very difficult to compare results across different LCAs. For example, levels of GHG inclusion from secondary and tertiary processes vary widely in LCA GHG accounting. A "true" and complete LCA would ideally account for all the GHGs associated with beef production, including far removed processes and energy needed to create inputs for supporting processes (i.e. energy needed to make the tractor to till the land to grow the corn to feed the cattle). However, this all-inclusive accounting is not practical. It has been suggested that designating LCAs with a numerical suffix indicating the "degree of separation" would be a helpful method. As described originally by Lal (2004), denoting LCAs by primary, secondary, and tertiary would help identify how remote the GHG accounting is in each model (Lal, 2004).

Also, management varies by region with climate and cultural nuances, for example, leading to different levels of fertilizer application, manure management and subsequent land application, feed components, and handling of cattle (i.e. inclusion of a stockering/backgrounding phase in some regions). Therefore, these production systems must be studied independently to better understand the site-specific environmental consequences (Pitesky et al., 2009; Stackhouse-Lawson et al., 2012). Lastly, representation of resulting emissions is inconsistent across LCAs. Although the most common functional unit is kg CO_2-e per kg of carcass weight, others have expressed emissions on a live-weight basis (Dudley et al., 2014), or tonnes of CO_2-e per 1.0×10^9 kg of beef (Capper, 2012), amplifying the difficulty in comparing LCA results.

In the last section of this review, the divergence of alternative grazing mechanisms such as AMP grazing from continuous grazing was explored. Because continuous grazing is largely the only grazing strategy explored by LCAs, this represents a narrowly focused approach. Developing LCAs for different kinds of grazing systems is an important and critical area for study and improvement in the future. It is possible that modeling AMP grazing and its contribution to both GHG emissions and other ecosystem services will result in different outcomes than indicated in previous LCAs utilizing a continuous grazing model.

1.8. Conclusions

The GHG emissions from the livestock sector vary widely by management practice, regional climate, and LCA accounting methods. While it is clear that promising and significant mitigation techniques and technologies exist, including AMP grazing, their widespread adoption and overall potential have yet to be thoroughly studied or realized. According to the literature, this is due to a number of things, including adoption barriers, cost of adoption of sustainable methods, lack of investment, and lack of political action to incentivize sustainability and promote healthy levels of consumption of livestock products (Herrero et al., 2016). To promote the movement into more sustainable livestock practices, effective, long-term agricultural policies should aim to support ecologically resilient and pastoral ecosystems, mitigate GHG emissions, address the interface among agriculture, society, the economy, and culture, and avoid unintended consequences from production (W. Teague et al., 2016).

The expanding population presents an additional complexity to the realm of sustainable livestock production. The combination of mitigation techniques with a reduction in overall meat consumption in diets seems to be necessary in developed countries. However, in developing and low-income countries where meat and milk products are main sources of income and calories alike, reduction in consumption and production does not seem a likely option. Currently, however, "The continuing trend of increasing global consumption of meat is not compatible with reducing GHG emissions from agriculture," (Herrero et al., 2016).

Some have argued that attention would be better allocated to reducing emission from the cow-calf stage of production rather that the feedlot/end-stage of cattle (Stackhouse-Lawson et al., 2012). Arguably, the findings from Capper (2012), Lupo (2013), and Pelletier (2010) would support this conclusion. However, others have shown that not only are there potential mitigation opportunities to be realized within the finishing stage, but beef finishing could become a C sink rather than an emission source- helping to offset emissions from the cow-calf stage. One such mitigation strategy, as explored by this review, is AMP grazing. Additionally, another option to be considered is a dual purpose meat-milk production system (Gerber, 2013).

What can be stated confidently after a thorough review of the literature is that cattle can be both an environmental detriment and a key in environmental stewardship. At any given time in any given place, which position they take depends on management practices. To ensure food security, the production of healthy protein products, and environmental quality in the near future, actions must be taken to not only limit GHG emissions from the livestock sector, but also repair and restore the ecosystems that have already been damaged.

LITERATURE CITED

LITERATURE CITED

Brilli, L., Bechini, L., Bindi, M., Carozzi, M., Cavalli, D., Conant, R., . . . Farina, R. (2017). Review and analysis of strengths and weaknesses of agro-ecosystem models for simulating C and N fluxes. *Science of the Total Environment, 598*, 445-470.

Briske, D. D., Derner, J., Brown, J., Fuhlendorf, S., Teague, W., Havstad, K., . . . Willms, W. (2008). Rotational grazing on rangelands: reconciliation of perception and experimental evidence. *Rangeland Ecology and Management, 61*(1), 3-17.

Broucek, J. (2014). Production of Methane Emissions from Ruminant Husbandry: A Review. *Journal of Environmental Protection, 5*(15), 1482-1493.

Brown, K., Lemon, J. (2016). Cations and Cation Exchange Capacity. Retrieved from http://www.soilquality.org.au/factsheets/cation-exchange-capacity

Bruce, J. P., Frome, M., Haites, E., Janzen, H., Lal, R., and Paustian, K. (1999). Carbon sequestration in soils. *Journal of Soil and Water Conservation, 54*(1), 382-389.

Capper, J. L. (2012). Is the Grass Always Greener? Comparing the Environmental Impact of Conventional, Natural and Grass-Fed Beef Production Systems. *Animals: an open access journal from MDPI, 2*(2), 127-143.

Cardoso, A. S., Berndt, A., Leytem, A., Alves, B. J. R., de Carvalho, I. d. N. O., de Barros Soares, L. H., . . . Boddey, R. M. (2016). Impact of the intensification of beef production in Brazil on greenhouse gas emissions and land use. *Agricultural Systems, 143*, 86-96.

Casey, J. W., and Holden, N. M. (2006). Greenhouse gas emissions from conventional, agri-environmental scheme, and organic Irish suckler-beef units. *Journal of Environmental Quality, 35*(1), 231-239.

Cheung, R., McMahon, P. (2017). *Back to Grass: The Market Potential for U.S. Grassfed Beef.* Retrieved from http://www.stonebarnscenter.org/images/content/3/9/39629/Grassfed-MarketStudy-F.pdf

Chiavegato, M. B., Rowntree, J., Powers, W. J., and Carmichael, D. (2015a). Pasture-derived greenhouse gas emissions in cow-calf production systems. *Journal of Animal Science, 93*(3), 1350.

Church, D. C. (1993). *The Ruminant Animal: Digestive Physiology and Nutrition* (2 ed.): Prentice-Hall.

Conant, R. T. (2010). *Challenges and opportunities for carbon sequestration in grassland systems*: FAO.

Dudley, Q. M., Liska, A. J., Watson, A. K., and Erickson, G. E. (2014). Uncertainties in life cycle greenhouse gas emissions from US beef cattle. *Journal of Cleaner Production, 75*, 31-39.

EPA. (2013). Overview of Greenhouse Gases. *Climate Change.* Retrieved from http://www3.epa.gov/climatechange/ghgemissions/gases/n2o.html

EPA. (2016a). Greenhouse Gas Emissions- Understanding Global Warming Potentials. Retrieved from https://www.epa.gov/ghgemissions/understanding-global-warming-potentials

EPA. (2016b). *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2014*. Retrieved from https://www.epa.gov/ghgemissions/inventory-us-greenhouse-gas-emissions-and-sinks

FAO. (2012). *Livestock and Landscapes*. Food and Agriculture Organization of the United Nations.

FAO. (2016). Animal Production. Retrieved from http://www.fao.org/animal-production/en/?%DC%98%E0%07=

FAO, Steinfeld, H., Gerber, P., Wassenaar, T. D., Castel, V., Rosales M, M., and Haan, C. d. (2006). *Livestock's long shadow: environmental issues and options*.. Rome: Food and Agriculture Organization of the United Nations.

Franzluebbers, A. J. (2010). *Grassland Carbon Sequestration: Management, Policy and Economics* Vol. 11. *Chapter VIII: Soil organic carbon in managed pastures of the southeastern United States of America*

Gerber, P. J., Steinfeld, H., Henderson, B., Mottet, A., Opio, C., Dijkman, J., Falcucci, A. and Tempio, G. (2013). *Tackling climate change through livestock: a global assessment of emissions and mitigation opportunities*.

Gould, M. C. (2015). Compost increases the water holding capacity of droughty soils. Retrieved from http://msue.anr.msu.edu/news/compost_increases_the_water_holding_capacity_of_droughty_soils

Gustavo, G. T. C., Ryan, M. R., and Richard, T. L. (2013). Energy Use and Greenhouse Gas Emissions from Crop Production Using the Farm Energy Analysis Tool. *BioScience, 63*(4), 263-273.

Halberg, N., van der Werf, H. M. G., Basset-Mens, C., Dalgaard, R., and de Boer, I. J. M. (2005). Environmental assessment tools for the evaluation and improvement of European livestock production systems. *Livestock Production Science, 96*(1), 33-50.

Hancock, D. (2016). Storing Carbon (Presentation at the Grassfed Exchange ed.). Presntation at the 2016 Grassfed Exchange Conference, Perry, GA.

Herrero, M., Havlík, P., Valin, H., Notenbaert, A., Rufino, M. C., Thornton, P. K., . . . Obersteiner, M. (2013). Biomass use, production, feed efficiencies, and greenhouse gas emissions from global livestock systems. *Proceedings of the National Academy of Sciences of the United States of America, 110*(52), 20888-20893.

Herrero, M., Henderson, B., Havlík, P., Thornton, P. K., Conant, R. T., Smith, P., . . . Gill, M. (2016). Greenhouse gas mitigation potentials in the livestock sector. *Nature Climate Change*.

IPCC. (2006). *IPCC Guidelines for Greenhouse Gas Inventories*. Retrieved from http://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/4_Volume4/V4_10_Ch10_Livestock.pdf

ISO. (2006). 14040 *Environmental management- Life cycle assessment principles and framework*: International Organization for Standardizations.

Janzen, H. H. (2011). What place for livestock on a re-greening earth? *Animal Feed Science and Technology, 166*, 783-796.

Johnson, K. A., and Johnson, D. E. (1995). Methane emissions from cattle. *Journal of Animal Science, 73*(8), 2483-2492.

Jones, S. K., Rees, R. M., Skiba, U. M., and Ball, B. C. (2005). Greenhouse gas emissions from a managed grassland. *Global and Planetary Change, 47*(2), 201-211.

Kaiser, M., Ellerbrock, R. H., and Gerke, H. H. (2008). Cation exchange capacity and composition of soluble soil organic matter fractions. *Soil Science Society of America Journal, 72*(5), 1278-1285.

Kebreab, E., Johnson, K. A., Archibeque, S. L., Pape, D., and Wirth, T. (2008). Model for estimating enteric methane emissions from United States dairy and feedlot cattle. *Journal of Animal Science, 86*(10), 2738-2748.

Lal, R. (2004). Carbon emission from farm operations (Vol. 30, pp. 981-990). OXFORD: Elsevier Ltd.

Liebig, M. A., Gross, J. R., Kronberg, S. L., Phillips, R. L., and Hanson, J. D. (2010). Grazing management contributions to net global warming potential: a long-term evaluation in the Northern Great Plains. *Journal of Environmental Quality, 39*(3), 799-809.

Lupo, C. D., Clay, D. E., Benning, J. L., and Stone, J. J. (2013). Life-Cycle Assessment of the Beef Cattle Production System for the Northern Great Plains, USA. *Jounral of Environmental Quality, 42*(5), 1386-1394.

Machmuller, M. B., Kramer, M. G., Cyle, T. K., Hill, N., Hancock, D., and Thompson, A. (2015). Emerging land use practices rapidly increase soil organic matter. *Nature Communications, 6*, 6995.

McAllister, T., Cheng, K.-J., Okine, E., and Mathison, G. (1996). Dietary, environmental and microbiological aspects of methane production in ruminants. *Canadian Journal of Animal Science, 76*(2), 231-243.

McSherry, M. E., and Ritchie, M. E. (2013). Effects of grazing on grassland soil carbon: a global review. *Global Change Biology, 19*(5), 1347-1357.

NRCS, USDA. (2009). Soil Quality Indicators.

Oates, G., Jackson, R. (2015). Potential carbon sequestration and forage gains with management-intensive rotational grazing. Retrieved from https://www.cias.wisc.edu/potential-carbon-sequestration-and-forage-gains-with-management-intensive-rotational-grazing-research-brief-95/

Oates, L. G., Undersander, D. J., Gratton, C., Bell, M. M., and Jackson, R. D. (2011). Management-Intensive Rotational Grazing Enhances Forage Production and Quality of Subhumid Cool-Season Pastures. *Crop ScienceE, 51*(2), 892-901.

Ontl, T. A. S., L. A. (2012). Soil Carbon Storage. *Nature Education Knowledge, 3*(10).

Pelletier, N., Pirog, R., and Rasmussen, R. (2010). Comparative life cycle environmental impacts of three beef production strategies in the Upper Midwestern United States. *Agricultural Systems, 103*(6), 380-389.

Pimentel, D., Pimentel, M. (2008). *Food, Energy, and Society* (Third ed.): CRC Press; Taylor and Francis Group.

Pitesky, M. E., Stackhouse, K. R., and Mitloehner, F. M. (2009). Clearing the Air: Livestock's Contribution to Climate Change (Vol. 103, pp. 1-40).

Powers, W., B. Auvermann, A. Cole, C. Gooch, R. Grant, J. Hatfield, P. Hunt, K. Johnson, A. Leytem, W. Liao, J. M. Powell. (2014). *Chapter 5: Quantifying Greenhouse Gas Sources and Sinks in Animal Production Systems. In Quantifying Greenhouse Gas Fluxes in Agriculture and Forestry: Methods for Entity-Scale Inventory.* Retrieved from Office of the Chief Economist, U.S. Department of Agriculture.

Rabalais, N. N., Turner, R. E., and Wiseman Jr, W. J. (2002). Gulf of Mexico hypoxia, AKA" The dead zone". *Annual Review of Ecology and Systematics*, 235-263.

Rayburn, E. (2000). *Overgrazing Can Hurt Environment, Your Pocketbook*. Retrieved from West Virginia Farm Bureau News: https://www.wvu.edu/~agexten/forglvst/overgraz.htm

Ripple, W. J., Smith, P., Haberl, H., Montzka, S. A., McAlpine, C., and Boucher, D. H. (2014). Ruminants, climate change and climate policy. *Nature Climate Change, 4*(1), 2.

Rivera-Ferre, M. G., López-i-Gelats, F., Howden, M., Smith, P., Morton, J. F., and Herrero, M. (2016). Re-framing the climate change debate in the livestock sector: mitigation and adaptation options: Mitigation and adaptation options in the livestock sector. *Wiley Interdisciplinary Reviews: Climate Change*.

Rowntree, J. E., Ryals, R., DeLonge, M., Teague, W. R., Chiavegato, M., Byck, P., . . . Xu, S. (2016). Potential mitigation of midwest grass-finished beef production emissions with soil carbon sequestration in the United States of America. *Future of Food: Journal on Food, Agriculture and Society, 4*(3), 31-38.

Russelle, M. P., Entz, M. H., and Franzluebbers, A. J. (2007). Reconsidering integrated crop-livestock systems in north America. *Agronomy Journal, 99*(2), 325-334.

Savory, A., Butterfield, J. (1998). *Holistic Management: A New Framework for Decision Making* (2nd ed.): Island Press.

Schader, C., Muller, A., Scialabba, N. E.-H., Hecht, J., Isensee, A., Erb, K.-H., . . . Niggli, U. (2015). Impacts of feeding less food-competing feedstuffs to livestock on global food system sustainability. *Journal of the Royal Society, Interface / the Royal Society, 12*(113).

Soussana, J. F., Allard, V., Pilegaard, K., Ambus, P., Amman, C., Campbell, C., . . . Valentini, R. (2007). Full accounting of the greenhouse gas (CO 2, N 2O, CH 4) budget of nine European grassland sites. *Agriculture, Ecosystems and Environment, 121*(1), 121-134.

Stackhouse-Lawson, K. R., Rotz, C. A., Oltjen, J. W., and Mitloehner, F. M. (2012). Carbon footprint and ammonia emissions of California beef production systems. *Journal of Animal Science, 90*(12), 4641-4655.

Stewart, A. A., Little, S. M., Ominski, K. H., Wittenberg, K. M., and Janzen, H. H. (2009). Evaluating greenhouse gas mitigation practices in livestock systems: an illustration of a whole-farm approach. *The Journal of Agricultural Science, 147*(4), 367-382.

Teague, W., Apfelbaum, S., Lal, R., Kreuter, U., Rowntree, J., Davies, C., . . . Wang, T. (2016). The role of ruminants in reducing agriculture's carbon footprint in North America. *Journal of Soil and Water Conservation, 71*(2), 156-164.

Teague, W. R., Dowhower, S. L., Baker, S. A., Haile, N., DeLaune, P. B., and Conover, D. M. (2011). Grazing management impacts on vegetation, soil biota and soil chemical, physical and hydrological properties in tall grass prairie. *Agriculture, Ecosystems and Environment, 141*(3), 310-322.

Turner, R. E., Rabalais, R.N. (2017). *2017 Forecast: Summer Hypoxic Zone Site Northern Gulf of Mexico*. Louisiana State University Department of Oceanography and Coastal Services.

UGA. (2015). What Is Management-Intensive Grazing? Retrieved from http://www.caes.uga.edu/topics/sustainag/grazing/

USDA. (2014). *Cattle Final Estimates 2009-2013*. Retrieved from http://usda.mannlib.cornell.edu/usda/nass/SB1034/sb1034.pdf.

USDA. (2015). *Summary Report: 2012 National Resources Inventory*. Center for Survey Statistics and Methodology.

Vries, d. M., Middelaar, v. C. E., and Boer, d. I. J. M. (2015). Comparing environmental impacts of beef production systems: A review of life cycle assessments. *Livestock Science, 178*, 279-288.

Wang, T., Teague, W. R., Park, S. C., and Bevers, S. (2015). GHG Mitigation Potential of Different Grazing Strategies in the United States Southern Great Plains. *Sustainability, 7*(10), 13500-13521.

CHAPTER 2

IMPACTS OF CARBON SEQUESTRATION ON LIFE CYCLE EMISSIONS IN MIDWESTERN USA BEEF FINISHING SYSTEMS

2.1. Introduction

Beef production can be an environmentally deleterious process. While livestock as a whole contribute about 14.5% of all global anthropogenic greenhouse gas (GHG) emissions, beef cattle alone account for more than half of this total, more than any other livestock category (Gerber, 2013). The United States is the leading beef producer and is among the top beef consumers globally, producing 19% of the world's total and consuming an average 25 kg of beef per person per year in 2017 (Beef2Live, 2017; OECD, 2016). Therefore, beef production is an area of interest for both global and domestic reductions in GHG intensity. Life cycle analysis (LCA) modelling is the most common approach to GHG emissions accounting, and has been used to estimate the environmental impacts of beef production.

In previous beef LCA literature, grass-fed or grass-finished systems are often modeled using continuous grazing parameters, a simplistic management strategy in which cattle are left to graze in a pasture continuously through a grazing season. This approach, while still the most common strategy, disallows plant regrowth and recovery and is low in terms of production. However, grazing management techniques are highly variable, ranging from continuous to light rotational to intensively managed. Accordingly, the land, ecosystem, and GHG impacts resulting from beef production are highly dependent on the type of management system utilized for finishing (Brilli et al., 2017; Rowntree et al., 2016). Additionally, because grass-fed beef production has been increasing as a response to consumer demand in recent years, it would be useful to examine the differing environmental impacts among alternative grazing systems. Some literature has pointed to beneficial ecosystem services resulting from the adoption of adaptive multi-paddock (AMP) grazing (Conant et al.). This type of grazing was originally characterized by Andre Voisin in the 1950's (Voisin, 1959). Then termed "rational grazing," it has since undergone iterations as "holistic planned grazing" (Savory, 1998) and "management-intensive grazing (Gerrish, 2004). Potential AMP benefits include reduction in overgrazing and soil erosion, improved forage utilization and animal production, and soil carbon sequestration, which may reduce overall net emissions. Soil carbon sequestration is an important ecosystem service in grasslands, which can be maximized using AMP livestock grazing according to some studies (Liebig et al., 2010; McSherry and Ritchie, 2013; Wang et al., 2015). However, few studies have captured the results of soil C change over time, and existing data are variable (Olson et al., 2017). For these reasons, soil C equilibrium is often assumed in beef LCAs. However, considering that the terrestrial C pool is three times greater than the atmospheric pool, and since grazing animals occupy much of this land, grassland sinks may represent a significant GHG mitigation strategy and should be included in beef production models (Olson et al., 2017).

Previous LCAs that have compared grass-fed to feedlot finishing models have unilaterally concluded that feedlot finishing systems have lower cradle-to-gate GHG emissions due to high enteric methane emissions attributed to the higher fiber diets and concomitant longer finishing time in grass-fed systems (Pelletier et al., 2010; Capper, 2012; Stackhouse et al., 2012; Lupo et al., 2013). However, some have suggested that emissions arising from feedlot finishing may be underestimated due to the lack of representation of soil characteristics during feed production, such as soil erosion (Janzen, 2011). Even though soil erosion declined by 44% from 1982-2012, 15 million acres of "prime farmland" were lost over the same time frame, and 4.60 tons of soil acre^{-1} are still lost annually from cropland (USDA, 2012, 2015b). Because soil organic matter (SOM) consists of 58% C, soil erosion also contributes to GHG emissions. Furthermore, cattle consume one-third of all grain produced in the U.S. (FAO, 2012; Schader et al., 2015). For these reasons, soil erosion is an important indicator of sustainability and should be incorporated into LCA accounting. Additionally, emissions arising from grass-fed systems may be highly variable due to regional and on-farm practices. For example, assumptions about fertilization rate on pasture have resulted in a five-fold difference in N_2O emissions (Lupo et al., 2013; Pelletier et al., 2010). Studies have identified these gaps and variation and researchers have called for more robust research and soil C inclusion in future LCA models (Lupo et al., 2013; Pelletier et al., 2010).

Considering the variability in grazing strategies and research gaps in soil C presented above, the goal of this research was to develop a model to estimate environmental impacts associated with feedlot finishing compared with an alternative grazing technique, AMP, with the inclusion of robust soil C accounting. To do this, an ISO compliant partial life cycle analysis was conducted for the finishing stage of cattle production in the Upper Midwest and combined with soil C sequestration rates, which were determined from four-year on-farm soil data, to determine net system GHG emissions.

2.2. Materials and Methods

2.2.1. System boundaries

Because much of beef production system variability is concentrated within the finishing stage, this model was confined to the finishing stage only. Two different finishing schemes common to the Upper Midwest, feedlot and AMP grazing, were modeled using a combination of on-farm data and literature information. All common emissions categories, including enteric methane (CH_4), manure, feed production, and on-farm energy use were included. Tertiary emissions, including those emitted from production of machines, equipment, and infrastructure were not included based on their assumed minor contribution (Lupo et al., 2013). Gasses were converted to carbon dioxide equivalents (CO_2-e) using their 100-year global warming potentials as reported by the IPCC 2014 Fifth Assessment Report (CO_2=1, CH_4=34, N_2O=298) (IPCC, 2014). For continuity and comparison with previous beef LCAs, the functional unit was set at kg CO_2-e per kg of carcass weight (kg CO_2-e kg CW^{-1}).

2.2.2. Finishing systems

2.2.2.1 Adaptive multi-paddock grazing

Five year (2012-2016) on-farm data from MSU Lake City AgBioResearch center were used for the AMP grazing finishing system. Red Angus steers (n=210) were weaned on-farm in mid-May and AMP grazed until mid-November, with exact dates varying year-to-year depending on occurrence of frost, weather and forage quality. Off-farm hay was offered with stockpiled forage until slaughter in early-to-mid December. On average, steers finished at 528 kg (σ=47 kg) live weight in 200 days (average 180 days grazing and 20 days fed hay), which is 150 days shorter than the grass-finishing time assumed by Lupo et al. (2013). Steers were grazed on high-quality cool season grass and alfalfa mix pastures at an average stocking rate of 2.7 steers/hectare (1.1 steers/acre) and rotated according to forage availability and quality. Detailed botanical composition is given by Chiavegato et al. (2016b). Rotation frequency was focused on prevention of overgrazing and assurance of forage recovery; therefore, allowing appropriate regrowth before being grazed again. Pastures were not fertilized, irrigated or treated for pests for greater than seven years prior to this management implementation in 2010. The five-year average annual rainfall during the months of study was 34.52 inches ± 2.40 inches. Average daily gain (ADG) was calculated from actual animal data, while dry matter intake (DMI) and net energy for maintenance (NE_m) were calculated using equations derived from the beef cattle NRC (NRC, 2016), and feed conversion efficiency was calculated following Cassady et al. (2016).

Average dressing percentage calculated from steer carcass data was 53% across all five years and was calculated with steer carcass data.

2.2.2.2 Feedlot

The feedlot (FL) finishing system was based on two-year data (2015-2016) from a research trial at the MSU Beef Center, where Red Angus steers (n=16) were feedlot finished for 90 days (Minasny et al., 2017). Steers were subsampled from the same herd at Lake City AgBioResearch Center, and therefore had the same genetic background as steers in the AMP grazing system. Because 90 days is not a representative timeline for actual on-farm feedlot-finishing, the research data were used to simulate an extension from 90-days on-feed to 170-days on-feed (**Table A.1**.). This 170-day period is a common time-to-finish in the Upper Midwest and is similar to the average time spent in feedlot (NRC, 2016). The beef cattle NRC (2016) and on-farm ration data were used to simulate feedlot performance, including DMI, ADG, and energy partitioning. Feed required for maintenance was calculated to determine net energy for gain (NE_g) required for production using table 12-1 (NRC, 2016). Production measures were calculated at three, 57-day intervals to allow for fluctuations in DMI and ADG due to changing energy partitioning as the steers grew larger with time. Mean finishing weight was 654 kg, similar to the 2015 average reported by (USDA, 2016b). Calculated DMI, ADG, and feed conversion efficiency were all within +/- 0.1 kg of the actual study values reported by (Cassady et al., 2016), indicating that the simulation method was justified for the feedlot system. Dressing percentage was assumed to be 63% based on average yield grade of 3. Because the system boundary was limited to the finishing stage only, cow-calf and backgrounding stages were not included.

2.2.3 Enteric methane, manure methane and nitrous oxide emissions

Although on-farm data using SF_6 tracer gas for enteric CH_4 emissions from the Lake City AgBioResearch center AMP grazing system were available (Chiavegato et al., 2015a), enteric CH_4 emissions were modelled using IPCC (2006) Tier 2 for consistency with other LCA literature. Gross energies (bomb calorimeter-derived total combustion energy contents) were calculated using actual feed components and animal characteristics from farm data for both systems. Default methane conversion factors (Y_m) of 3% and 6.5% for feedlot and grazed cattle, respectively, were applied.

Manure CH_4 and N_2O were estimated using IPCC (2006) Tier 2 methodologies. During feedlot finishing at the MSU Beef Center, manure was collected beneath the slatted floors in the confinement pens for approximately one year before being pumped out and spread on nearby wheat fields in late July-early August. Therefore, emissions were calculated and summed for the liquid/slurry phase during manure management at the feedlot, and for the land application phase. All manure during AMP grazing finishing at the MSU Lake City AgBioResearch center was deposited on pasture, and therefore emissions were coordinated using IPCC models for emissions from managed soils (IPCC, 2006). Soil N dynamics for the pasture itself were not modeled under the assumption that these emissions are not directly attributable to the animal production. Nitrogen soil dynamics from feed production (i.e. from synthetic fertilizer) were accounted for in feed emissions. Fertilizer offsets were not considered in this model because all collected manure from the feedlot was land applied to crops not used for the livestock feed. However, considering animal manure as a fertilizer offset would reduce the overall N_2O emissions from feed production.

Manure CH_4 emissions were calculated as a function of volatile solids composition in the manure according to dietary gross and digestible energies from both feedlot and the AMP grazing finishing system. Because on-farm and regional data were available, direct and indirect manure N_2O were calculated using IPCC (2006) Tier 2 methods. Using actual weight gain, dietary NEg and crude protein, excreted nitrogen (N_{ext}) was calculated and used to determine direct N_2O emissions from manure. Volatized N, which contributes to indirect N_2O, was calculated using the default EF_4 of 0.10 kg N_2O-N because of the relatively low range of uncertainty (+/- 0.002-0.05) (IPCC, 2006). Indirect N_2O emissions arising from leaching and runoff (NH_3 and NO_x) were calculated using the regression equation proposed by Rochette et al. (2008), using a relationship between precipitation (P) and potential evapotranspiration (PE). Data for P and PE were collected from the MSU Enviro-weather database and were averaged for the 2015 and 2016 growing seasons (May 15 - Oct 15). $Frac_{leach}$ was therefore calculated as 0.28 kg N leached per kg N excreted, which is double the value calculated by (Lupo et al., 2013), likely because of the higher P and PE values for the Upper Midwest compared with the Northern Great Plains.

Manure CH_4 and direct and indirect N_2O were converted to a CO_2-e basis for both finishing strategies, and summed across the two manure management systems for the feedlot finished cattle.

2.2.4 Feed Production

Emissions arising from FL-feed inputs were estimated using the actual ration components' nutrient and energy composition and dietary proportions from the Michigan State University Beef Center. Accordingly, five agricultural products were included: corn grain, high moisture corn (HMC), corn silage, alfalfa hay, and dried distillers grains with solubles (DDGs)

(**Table A.2.**). Crop yields were derived from Michigan specific crop data according to (USDA, 2015a), except for HMC, which originated from (Schroeder, 2012) because of data availability. The ration varied between years, with more corn grain included in 2015 and more HMC in 2016. Because HMC is a high yielding commodity crop, the acreage needed was much smaller in 2016. Therefore, the acreage was averaged between years to obtain a more standard representative.

While it would have been ideal to have emissions data from cultivation and manufacturing of feed products, these data were not available and therefore we elected to use existing feed emissions data from Gustavo et al., (2013) for all feed components except for DDGs. Because this study represented a robust analysis of feed-crop emissions using the Farm Energy Analysis tool (FEAT) calibrated with an extensive literature reviewed database, we believe that it contributed to our model in a representative manner. Emissions resulting from transportation of feed from farm to manufacturing facility, drying, on-farm fuel use, and application of insecticide, herbicide, lime, K_2O, P_2O_5, and synthetic N were included. Because we were unable to identify the GHG footprint of HMC in the literature, emissions were modeled in the same way as corn grain. Emissions from production of DDGs was taken from a Michigan model (Kim and Dale, 2008), because of the proximity of the study to Lake City AgBioResearch center, which included corn crop production and dry-milling, but not ethanol distillation. Because some studies use co-product allocation as a GHG offset for DDGs (because of its dual-purpose for ethanol production), it is likely that these emissions may be underestimated as pointed out by Hall et al. (2011). It was estimated that each 25.4 kg (56 lb) bushel of corn used in dry-mill ethanol production generates 7.89 kg (17.4 lbs) of DDGs (USDA, 2016a).

Emissions from crop irrigation were not included because <10% of all cropland is irrigated in Michigan according to the 2012 USDA Census of Agriculture (USDA, 2014).

However, in the Western US where irrigation is more predominant, irrigation might represent an important source of GHG emissions. According to (Sloggett, 1992), 23% of the on-farm energy used for crop production in the US was for on-farm pumping of water. Additionally, emissions associated from land use change, either for feed production or grazing land, were not considered under the assumption that feed crops were produced from existing cropland and grass-fed cattle were grazed on existing pasture/grassland in Michigan or other regions of the Upper Midwest.

Pastures for the AMP grazing system were not irrigated, fertilized or treated for pests, and therefore no GHG emissions were generated from those processes. Emissions resulting from off-farm alfalfa hay used during the non-grazing season were taken from Gustavo et al. (2013). Emissions from the production of mineral supplements in the ration were simulated using methodology by Lupo et al. (2013), and were scaled to represent the difference in finishing time for both production systems.

Total area (ha) needed to produce feed to finish a steer in each system was then calculated using their respective feed component amounts. The total mixed ration (TMR) of the FL was broken down into the dry matter (DM) weight of each feed component fed to each steer, and land area was calculated retrospectively by determining the hectares needed per feed component based on USDA crop data for Michigan (USDA, 2015a). Similarly, land area was calculated for AMP grazing using alfalfa yield (USDA, 2015a) for grazed pastures and locally sourced off-farm hay, and an assumed 70% grazing forage utilization rate at the MSU Lake City AgBioResearch center, based on on-farm observations (Doug Carmichael, manager, personal communication).

2.2.5 On-farm fuel use and transportation

Similar to Pelletier et al. (2010) on-farm energy use per head of cattle was taken from (Ryan and Tiffany, 1998) in the absence of current or on-farm data. Energy use was converted to GHG emissions via the EPA, (2015) calculations approach. Transportation of inputs (i.e., fertilizer and pesticides) and transportation of feed from farm to manufacturer (if applicable) was accounted for in feed emissions. However, transportation of feed inputs from manufacturer (or from original farm) to beef production farm was accounted for in this section. All feed-inputs for both systems were assumed to be transported 30 km by truck, except for DDGs which was assumed to be 60 km to account for both transportation between manufacturers and to the feedlot, with a load capacity of 36,000 kg/load (Rowntree et al., 2016). Only energy consumed during the delivery load was included to account for an empty return (Lupo et al., 2013).

2.2.6 Soil carbon

2.2.6.1 Sample collection and carbon analysis

To determine four-year soil carbon sequestration, soil samples were collected at the Lake City AgBioResearch center. In 2012, soils in three sandy-loam (SL) replicate sites were sampled for soil C (Chiavegato et al., 2015c). In 2016, soils were also sampled across farm at nine GPS sites, representing three soil types: sandy (S), clay loam (CL) and SL. Sandy loam soils makes up approximately 70% of all soil types across the farm at the Lake City AgBioResearch center. While two 2016 SL sites were adjacent to 2012 sample sites (< 50 m), one SL site was farther (>100 m). For each site, 10 soil sub-samples were collected at four different depths (0-5, 5-10, 10-20, and 20-30 cm), and combined by depth. In both 2012 and 2016, samples were collected in the fall of each year.

For soil C stock determination, analysis was conducted as reported in Chiavegato et al., (2015c). Combined samples from each depth were sieved at 2mm and then dried at 50˚C separately by depth until reaching a constant weight. Soils were then ground using a Ball Mill and analyzed for carbon (C) using an ECS 4010 CHNSO Elemental Combustion System (Costech, Valencia, CA).

During soil collection, to determine bulk density, an additional three 30 cm samples per site were collected using a 9120-Rap Powerprobe hydraulic soil sampler (AMS, American Falls, ID). A 5.08 cm^2 core was drilled the entire 0-30 cm depth and cut into their respective depth fractions to prevent soil compaction. The soils were weighed and then dried to a constant weight and reweighed. Bulk density was calculated by dividing the dry weight by the hydraulic core volume of each soil sample after the removal of rocks.

Once soil C stocks were calculated, we used the 2012 SL as a baseline for sequestration from 2012-2016. For overall SOC sequestration, we compared each 2016 soil type, S, SL and CL to the 2012 baseline to generate an estimated sequestration relative to the 2012 SL soils within each soil type sampled in 2016. For overall SOC sequestration, we aggregated mean soil C stock from the nine replicate sites in 2016, and compared these back to the 2012 SL mean soil C stock. Carbon content in SL soils was centric compared to C content of S and CL soils, therefore this method was conservative relative to comparing C stocks only between SL soil types.

2.2.6.2 Soil erosion

In lieu of on-farm data for soil erosion and SOM from cropland used to grow feed inputs, representative literature values were used. According to the USDA NRCS, average sheet and rill soil erosion on cropland in Michigan equals 3.29 Mg $acre^{-1}$ yr^{-1}. Characteristic SOM data were taken from long-term ecological research at the Michigan State University Kellogg Biological

Station (Syswerda et al., 2011). Average C content for soils under conventionally managed corn-soybean-wheat rotation, which is typical in the Upper Midwest, was 10.4 g C kg^{-1} soil (Syswerda et al., 2011). Soil erosion rate was multiplied by C content to get total C loss, which was then converted to a CO_2-e basis (**Table A.4.**). The resulting kg CO_2-e from soil erosion was then added to the total GHG emission footprint.

2.2.6.3. Soil carbon equivalent flux

To combine soil C sequestration with emissions of the beef production systems to estimate net emissions, soil C mass (Mg C/ha) was composited for the entire 0-30 cm depth, differences were taken from 2012-2016, and then divided by four years to get Mg C sequestered ha^{-1} yr^{-1}. This value was then divided by the land area needed (kg CW/ha feed) for animal production (which included grazed land and land needed for supplemental hay), and converted to CO_2-e. The net C sequestered, on a CO_2 basis, was then subtracted from total emissions from beef production.

2.3. Results and Discussion

2.3.1. Animal production

Animal production characteristics resulting from the simulation and five-year averaged data are presented for FL and AMP in **Table 2.1**, respectively. On average, cattle in the FL system finished in 30 less days and with 31% greater carcass weights than AMP grazed cattle. Average daily gain was 49% greater in the FL scenario, whereas DMI and feed conversion ratios were 8% and 228% lower in the FL scenario compared with the AMP scenario. Average finishing weights for the FL cattle were similar to those reported in other literature (Capper, 2012; Lupo et al., 2013; Pelletier et al., 2010), whereas AMP finishing weights were 10-20% heavier and finished in 51% less time than their grass-fed models (Capper, 2012; Lupo et al., 2013; Pelletier et al., 2010). This difference may be explained by improved forage utilization and quality in the AMP grazing system versus those that are conventionally grazed. Studies using AMP grazing suggest that it can result in more ideal pasture species composition (more digestible forages with greater digestible energy) due to prevention of consistent species selection and overgrazing, pasture recovery, and re-grazing before lignification (Oates et al., 2011; W. Teague et al., 2016; W. R. Teague et al., 2011; UGA, 2015).

Table 2.1. Animal production characteristics

	FL[1]	**AMP**[2]
Time (days)	171.50	200.80
Beg-End LW (kgs)	361-654	362-528
Dressing %	0.62	0.53
Avg ending carcass weight (kgs)	405.75	280.16
ADG (kg)	1.80	0.88
DMI (kg/d)	9.86	11.04
Feed conversion ratio[3]	5.70	13.04

[1] Feedlot (FL); taken from the averages of three, 57-day simulation intervals using (2015-2016) MSU Beef Center finishing data
[2] Adaptive Multi-paddock (AMP) Grazing; taken from five-year (2012-2016) Lake City Research Center finishing data
[3] Feed conversion ratio was calculated by dividing DMI by ADG (Cassady et al., 2016)

2.3.2 GHG Emissions

Total GHG emissions for each finishing strategy are reported in **Table 2.2,** and are broken down by category in **Figure 2.1**. The estimated GHG emissions associated with FL and AMP finishing in the Upper Midwest were 7.03 and 9.62 kg CO_2-e kg CW^{-1}, respectively. Generally, values were within the range reported in other studies in the Northern Great Plains (Lupo et al., 2013), Upper Midwest (Pelletier et al., 2010), and California (Stackhouse-Lawson et al., 2012). Compared with Pelletier et al., (2010), GHG emissions from the AMP grazing finishing system in our study were estimated to be 45% lower than estimated in their grass-finishing model. This was most likely the result of a shorter finishing time and greater animal productivity, and the high pasture fertilizer application in their model.

Similar to other studies, enteric CH_4 contributed the most to overall emissions (**Figure 2.1**) in AMP. Compared with FL finishing, the AMP grazing scenario generated double the enteric CH_4 emissions, highlighted in **Figure 2.1**. This emphasizes the benefits of higher energy feeds and shorter lifetime, resulting in productivity gains in our FL model. Despite the heavy uncertainty associated with IPCC CH_4 accounting, specifically with regards to the use of default Y_m values (Stackhouse-Lawson et al., 2012; Rowntree et al., 2016), we used this method for consistency with other literature. If, however, we had used the results of Chiavegato et al. (2015b) estimated from data collected at the MSU Lake City AgBioResearch center, the enteric CH_4 footprint in our AMP grazing model would have been reduced by 35%, representing a decrease from 1,434 to 1,030 kg CO_2-e steer^{-1} (**Table A.3.)**. Similarly, if Y_m had been reduced from the default of 6.5+/-1 to 5.5 to reflect improved forage quality, enteric CH_4 emissions from the AMP scenario would have been reduced by 15% (**Table A.3.**). Therefore, it is likely that the use of IPCC (2006) enteric CH_4 accounting methodology overestimates actual enteric CH_4 emissions, especially when high quality forages are grazed. This overestimation has also been reported in other studies (Rowntree et al., 2016; Stackhouse-Lawson et al., 2012), and reflects the uncertainty embedded in the use of IPCC methodology to represent a wide variety of diets.

Although less total manure emissions were estimated for the AMP grazing system than for the FL scenario (6.9 vs 7.3 Mg CO_2-e), when reflected on an animal productivity basis (CO_2-e kg CW^{-1}), manure emissions were actually greater than the FL scenario. This is the result of an elevated DMI and lower overall carcass weight. Manure CH_4 and N_2O for the FL scenario estimated by this study were greater than reported by others (Lupo et al., 2013; Pelletier et al., 2010). Likely, this is attributable to the use of IPCC Tier 2 methodology in the current study instead of Tier 1 employed in other studies, greater precipitation in our study area compared with others, and handling of manure in the FL as a liquid instead of a solid.

Our feed production emissions in the FL were slightly greater than feed production emissions reported by Lupo et al. (2013) and Pelletier et al. (2010), likely because our feedlot ration did not include a high proportion of legume forages and contained a larger portion of DDGs and corn grains, which according to Gustavo et al. (2013) have greater emissions compared with other feedstuffs. Additionally, the model employed by Gustavo et al., (2013) included additional inputs and processes not included in conventional beef production LCAs; for example, pest management and transportation of inputs. Fertilizer offsets from animal manure is a common assumption in many beef LCAs, however, they were not included in this study because all manure collected from the FL was spread on non-feed component cropland. Considering fertilizer offsets would have reduced our overall GHG footprint from feed production, and may consequently account for some of the deviation compared with other studies. Emissions from on-farm energy use and transportation and mineral supplement production represented a very small portion of total emissions, accounting for <2.5% and <0.05% of total GHG footprint, respectively.

Table 2.2. Greenhouse gas emissions (kg CO_2-e) associated with one steer for both adaptive multi-paddock (AMP) grazing and feedlot (FL) finishing stages for all impact categories and their percentage of total emissions.

	GHG emissions (kg CO2-e steer^{-1})	**% of total**
AMP	2694.64	
Enteric CH_4	1434.18	53.22%
Manure Emissions [1]	688.27	25.54%
Feed Emissions [2]	511.95	19.00%
Mineral Supplement Emissions [3]	0.76	0.03%
On-farm energy and transportation	59.47	2.21%
kg CW	280.16	
Total (kg CO_2-e kg CW^{-1})	**9.62**	
FL	2853.22	
Enteric CH_4	776.96	27.23%
Manure Emissions [1]	732.66	25.68%
Feed Emissions [2]	1288.20	45.15%
Mineral Emissions [3]	1.34	0.05%
On-farm energy and transportation	54.06	1.89%
kg CW	405.75	
Total (kg CO_2-e kg CW^{-1})	**7.03**	

[1] Predominately indirect N_2O, but also contains manure derived CH_4
[2] Calculated from respective feed components using Gustavo et al., (2013)
[3] Calculated using data from Lupo et al., (2013).

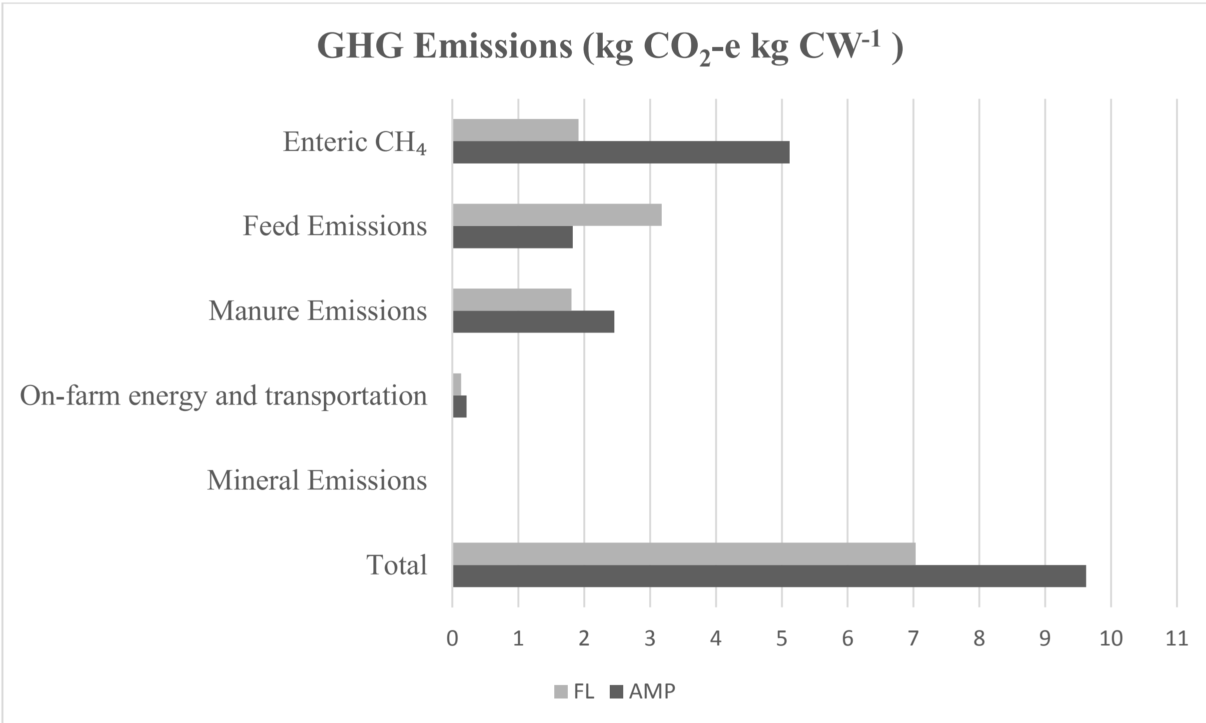

Figure 2.1. GHG emissions (kg CO_2-e kg CW^{-1}) presented by emissions category for feedlot (FL) and adaptive multi-paddock (AMP) grazing systems.

2.3.3. Soil C sequestration

When compared with the 2012 baseline (33.97 +/- 0.71 Mg C ha^{-1} in the top 30 cm, measured in SL soils), the 2016 soil C stock increased by 15.18, 8.16, and 19.75 Mg C ha^{-1} in SL, S, and CL, suggesting a four-year sequestration rate of 3.79, 2.04, and 4.94 Mg C ha^{-1} yr^{-1} within the three soil types, respectively. When S, SL and CL soil C stocks were aggregated, and compared back to 2012 baseline, the mean sequestration rate is 3.59 Mg C ha^{-1} yr^{-1} (**Table 2.3**). Ideally, we would have preferred to have SOC across each soil type in 2012, but those data were not available. However, given that the 2016 SL mean C contents are centric to overall on-farm SOC stocks when comparing to S and CL, we assumed for the purposes of this analysis that the SL soil C contents from 2012 were representative of the overall farm. Further, estimating sequestration rates using average values across all soil types from 2016 yielded a slightly lower rate than when using the 2016 SL data only. Therefore, we determined that using the 2012 SL samples as a baseline for the full field is more conservative and robust for our estimations.

Our estimated sequestration rate (3.59 Mg C ha^{-1} yr^{-1}) is considerably greater than the mean C sequestration rate of 0.41 Mg C ha^{-1} yr^{-1} for improved grazing reported by Conant et al., (2003). However, as indicated by Minasny et al. (2017), these numbers are more than 14 years old and may not accurately represent new understanding of soil C based on more dynamic soil and grazing models. Similar to the management in our study, Wang et al. (2015) reported 3.53 Mg C ha^{-1} yr^{-1} C sequestration for pastures transitioned from heavy continuous to AMP grazing in Northern Texas. This value is more consistent with our four-year sequestration rate of 3.59 Mg C ha^{-1} yr^{-1}.

Studies indicating projected saturation rates based on older soil C stock data suggest that our soils in the Upper Midwest are approaching saturation based on existing soil C stocks. However, because soils that are farther away from soil C saturation will accumulate C more quickly than soils near saturation, and because our estimated soil C sequestration rate is much higher than the 0.41 Mg C ha^{-1} yr^{-1} indicated by Conant et al. (2003), we hypothesize that our soils will continue to sequester C efficiently for an extended period before reaching saturation (Minasny et al., 2017; Stewart et al., 2007).

Alternatively, for the FL scenario, there was no C sequestration because intensively cropped soils associated with feed production are often heavily eroded (Olson et al., 2017). However, there is considerable opportunity for sustainability improvements on these croplands through different agronomic practices such as leaving crop residues, increased perennials in rotations, conservation and no-till, cover crops, and increased use of organic amendments such as compost (Chambers et al., 2016; Minasny et al., 2017). If adopted, these practices may also promote soil C sequestration (Chambers et al., 2016). Nevertheless, studies indicate that in general, grasslands continue to sequester C at higher rates than improved cropland, and that transitions from cropland to grassland show the greatest rates of C sequestration, in some cases as high as 8.0 Mg C ha^{-1} yr^{-1} (Machmuller et al., 2015; Mathew et al., 2017; Minasny et al., 2017; Powlson et al., 2011).

Table 2.3. Differences in soil C stock by year and soil type (top) and 4-year soil C sequestration rates by and across soil types (bottom).

	Soil C Stock (Mg C ha[1])			
	2012[2]		2016[3]	
Soil Type[4]	Mean	Std Error	Mean	Std Error
S	-	-	42.13	1.59
SL	**33.97**	0.71	49.15	6.55
CL	-	-	53.72	7.33
Across Type	-	-	**48.33**	**1.95**
	Soil C Sequestration (Mg C ha^{-1} yr^{-1})			
Soil Type[5]	4-year difference	Sequestration rate	Mean	Std Error
S	8.16	2.04	-	-
SL	15.18	3.79	-	-
CL	19.75	4.94	-	-
Across Type	14.36	**3.59**	3.59	0.84

[1] Represents the average composited 0-30 cm depth of all three replicates, for which ten subsamples were taken for each replicate and depth

[2] All samples collected in 2012 were from sandy loam soils. Average was taken for 2012 Period 2, SysA (data not shown) from (Chiavegato et al., 2015c)

[3] Farm samples from all three soil types were collected

[4] SL represents sandy loam, S represents sandy, and CL represents clay loam soil types

[5] Compared with the soil stock of sandy loam soils in 2012

2.3.4. Net GHG flux

Soil erosion contributed an additional 113.81 kg CO_2-e (0.17 kg CO_2 kg CW^{-1}) to the FL model. Zero soil erosion was assumed for the AMP scenario because our data indicated active C sequestration over the four-year interval measured in the land associated with this management. According to (Olson et al., 2016) soil that is not part of a stable aggregate and is eroding cannot be actively sequestering C. Although soil erosion only contributed modestly to GHG emissions, soil erosion magnified by intensive cropping systems has important negative impacts on other ecosystem services that are vital to food production, water quality, nutrient cycling and habitat support (Olson et al., 2017).

On a CO_2-e basis, total sequestration equaled -17.54 kg CO_2-e for the AMP grazing finishing scenario. Combined with 9.62 kg CO_2-e kg CW^{-1} in emissions, this resulted in a net GHG sink of -7.92 kg CO_2-e kg CW^{-1} (**Figure 2.2**). Subsequently, the 7.03 kg CO_2-e kg CW^{-1} of FL emissions combined with 0.17 kg CO_2-e kg CW^{-1} from erosion resulted in a net emission of 7.20 kg CO_2-e kg CW^{-1} for the FL scenario (**Figure 2.2**). In beef LCA studies from the Northern Great Plains and the Upper Midwest where soil C sensitivity was included, sequestration rates of 0.41 and 0.12 Mg C ha^{-1} yr^{-1} reduced emissions of grass-fed beef production by 24 and 30%, respectively (Lupo et al., 2013; Pelletier et al., 2010). While these studies were models generated from the literature, our model used five years of actual finishing data derived from Lake City AgBioResearch center, and four years of soil C data in paddocks where cattle were managed by AMP grazing. Therefore, long-term grazing-finishing projections in the Upper Midwest could be considerably more sustainable than previously thought. This calls into question the common assumption that intensification reduces overall GHG footprint through higher productivity. This study shows that when full consideration of land impacts is given to the GHG emissions in both grain and grazing-based production, the environmental benefits may outweigh the productivity losses.

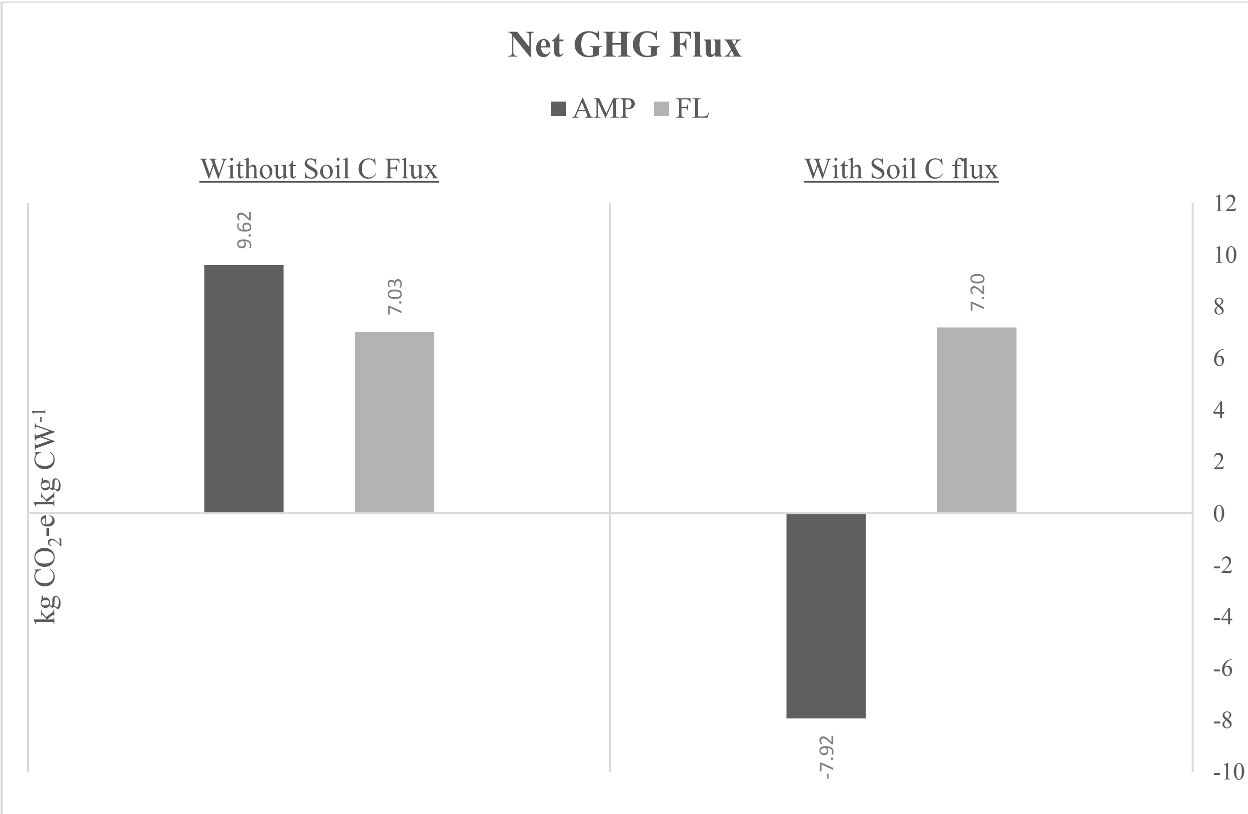

Figure 2.2. Emissions for each finishing strategy, adaptive multi-paddock (AMP) grazing and feedlot (FL), are reported for before and after soil C flux is considered. Bars on the left represent all emissions calculated through the LCA. Bars on the right represent net GHG flux after incorporation of soil carbon sequestration sinks and soil erosion additions on a CO_2-e basis.

Generally, because of the large enteric CH_4 footprint, grazing systems have been pointed to as the greatest area for attention in terms of GHG footprint in beef production. However, measurements of actual SOC have not been factored into these outcomes. In fact, if soil C sequestration is achieved through best management practices, enteric CH_4 from the finishing stage can be substantially mitigated. We demonstrate this based on soil C collection from Lake City AgBioResearch center from 2012-2016, which indicates a -7.92 kg CO_2-e kg CW^{-1} sink. Therefore, our results in **Figure 2.2** are considerably different than current literature and document the need to consider emissions and sinks from the entire system, including the soil ecosystem, when modelling beef production systems.

2.4. Conclusions

Using a standard LCA approach in combination with inclusion of soil C accounting, this study calculated net GHG emissions of two beef finishing systems in the Upper Midwest: feedlot and adaptive multi-paddock grazing. There are several important literature contributions arising from this. First, the incorporation of environmental influence of AMP grazing compared with continuous grazing, the common grass-fed management system utilized by most LCAs, yields considerably greater food provisioning with different environmental outcomes including a significant C-sink, which can offset enteric CH_4. Second, integrating on-farm soil C data within the studied management system contributes significantly to the current gap in the literature. To date, this is the first study that aims to address this gap. In doing so, our results show that not only can adoption of improved grazing management support soil C sequestration, but that this production system may serve as an overall GHG sink, likely for years into the future.

There are considerable implications emerging from this study, both for beef production and for climate change policy. According to West et al., (2014), an additional four billion people could be fed globally if livestock did not compete for arable cropland that could be used directly to produce human-edible food. However, existing beef LCAs suggest movement into grass-based beef production would drastically reduce energy efficiency and food production, while significantly increasing CH_4 emissions and needed land-area (Capper, 2012). Our results are contrary to this view. Potentially, an AMP grazing system can be a win-win scenario with marked increased in beef production per unit land when compared to LCAs in the existing literature, while also improving ecosystem services. Therefore, AMP has great potential to meet food production challenges as population projections increase while potentially improving the natural resource base human life depends on.

Continued investigation of AMP grazing in large-scale landscape trials across multiple ecoregions differing in climate and plant species mixes is warranted and necessary to best apply this strategy to local and regional conditions.

APPENDIX

A.1. Feedlot Simulation Calculations

An extension from a 90-day finishing research experiment to an industry representative 170-day finishing period was simulated (**Table A.1)**. Dry matter intake (DMI) was calculated from NRC (2016) Eq. 10-4 as

$$DMI, \frac{kg}{d} = 3.830 + 0.0143\ x\ ISBW$$

where *ISBW* is initial shrunk body weight (kg), calculated from live weight upon entrance into the FL. This equation is used for steers fed high-grain diets with NE_g greater than 1.4 Mcal kg NE_g^{-1}. This equation was chosen for our methods because the NE_g used in our feedlot ration was 1.46 Mcal kg NE_g^{-1}. NE_m required was calculated following NRC (2016) Eq. 11-1 as

$$NEm = 0.077\ SBW^{0.75}$$

where SBW is shrunk body weight. This was divided by actual NE_m consumed in the ration to determine feed required for maintenance. Thus, NE_g required was calculated as

$$NEg\ required\ for\ gain = (DMI - feed\ required\ for\ maintenance)\ x\ NEg\ consumed$$

where feed required for maintenance is in kg DM. Average daily gain (ADG) was extrapolated from NRC (2016) table 12-1 using the NRC (1984) equation

$$NEg = 0.0493\ x\ W^{0.75} x\ LWG^{1.097}$$

where W is weight (kg) and LWG is live weight gain.

Therefore, for each 57-day interval, weight gain was calculated, added to the weight at the beginning of the interval, and used as the beginning weight for the next interval. Feed composition is represented in **Table A.2**.

Table A.1. Feedlot simulation, broken down into three, 57-day intervals for both 2015 and 2016.

	F1: Days 1-57		F2: Days 58-114		F3: Days 115-171	
Production Measure	2015	2016	2015	2016	2015	2016
Entry SBW[1]	351.41	343.57	458.00	451.87	560.60	554.47
NE_g consumed[2] (Mcal kg d^{-1})	1.45	1.48	1.45	1.48	1.45	1.48
DMI[3] (kg d^{-1})	8.41	8.29	9.93	9.84	11.40	11.31
NE_m required[4] (Mcal kg d^{-1})	6.25	6.14	7.62	7.55	8.87	8.80
NE_m consumed[2] (Mcal kg DM^{-1})	2.13	2.16	2.13	2.16	2.13	2.16
Feed req. for maintenance (kg DM)	2.93	2.85	3.57	3.50	4.16	4.08
NE_g required[5] (Mcal d^{-1})	7.97	8.05	9.25	9.38	10.53	10.69
Shrunk ADG[6] (kg)	1.87	1.90	1.80	1.80	1.70	1.70
Weight gained (kg)	106.59	108.30	102.60	102.60	96.90	96.90
End weight (kg)	458.00	451.87	560.60	554.47	657.50	651.37

[1] Represents liveweight at entry into the feedlot (on-farm data) minus 4% conversion to shrunk body weight (SBW)
[2] Energy composition of actual TMR
[3] Calculated using NRC (2016) Eq. 10-4
[4] Calculated using NRC (2016) Eq. 11-1
[5] Calculated using NRC (2016), pg. 190
[6] Calculated using NRC (2016) table 12-1

Table A.2. Actual on-farm feed components. Feed composition and weights fed during the 90-day trial were scaled up based on the projected feed needed for each interval of the simulation and multiplied by the percentage that each feed component represented in the original ration.

	Year	
Feed component (kg DM)	2015	2016
Corn silage	158.56	282.38
DDGs	760.98	208.22
HMC	311.00	1144.65
Corn grain[1]	399.59	
Alfalfa hay	64.55	43.03
Total	1694.67	1678.29

[1] Corn grain was only fed in 2015. In 2016, this was compensated with increased corn silage and HMC.

A.2. Enteric CH_4 Sensitivity

Table A.3. Enteric CH_4 sensitivity analysis. Enteric CH_4 emissions calculated using IPCC default Y_m (6.5) was compared with emissions resulting from a 1.0 reduction to Y_m= 5.5. Additionally, enteric CH_4 from default IPCC methods was compared with on-farm enteric CH_4 data collected in 2012 using SF_6 tracer gas.

	Kg CH_4 steer^{-1}	% reduction
IPCC Y_m=6.5	42.18	
IPCC Y_m=5.5[1]	35.69	15.38%
2012 SF_6 Tracer[2]	27.24	35.41%

[1] Uncertainty for Y_m (IPCC, 2006) is +/- 1.0. Therefore, to reflect a higher forage quality, Y_m was reduced to 5.5.

[2] Data from (Chiavegato et al., 2015b). A 0.85 metabolic body weight conversion was used to reflect a difference between cow and steer enteric CH_4 production following (Rowntree et al., 2016). Emissions were also temporally scaled from a 395-day production cycle to our 172-day finishing period.

A.3. Soil Erosion Calculations

Table A.4. Conversions and calculations of soil erosion (tons soil acre^{-1} yr^{-1}) into GHG emissions (CO_2-e).

Conversions	Value	Source
tons soil acre^{-1} yr^{-1}	3.29	USDA NRCS (2012)
tonnes soil acre^{-1} yr^{-1}	2.98	
Average carbon %	0.01	Syswerda et al. (2011)
C loss (tonnes acre^{-1})	0.03	
C loss (kg acre^{-1})	31.04	
CO_2-e addition	113.81	
kg CO2e kg CW^{-1}	0.17	

LITERATURE CITED

LITERATURE CITED

Beef2Live. (2017). World Beef Production: Ranking Of Countries. Retrieved from http://beef2live.com/story-world-beef-production-ranking-countries-0-106885

Brilli, L., Bechini, L., Bindi, M., Carozzi, M., Cavalli, D., Conant, R., . . . Farina, R. (2017). Review and analysis of strengths and weaknesses of agro-ecosystem models for simulating C and N fluxes. *Science of the Total Environment, 598*, 445-470.

Capper, J. L. (2012). Is the Grass Always Greener? Comparing the Environmental Impact of Conventional, Natural and Grass-Fed Beef Production Systems. *Animals: an open access journal from MDPI, 2*(2), 127-143.

Cassady, C. J., Felix, T. L., Beever, J. E., and Shike, D. W. (2016). Effects of timing and duration of test period and diet type on intake and feed efficiency of Charolais-sired cattle. *Journal of Animal Science, 94*(11), 4748.

Chambers, A., Lal, R., & Paustian, K. (2016). Soil carbon sequestration potential of US croplands and grasslands: Implementing the 4 per Thousand Initiative. *Journal of Soil and Water Conservation, 71*(3), 68A-74A.

Chiavegato, M. B., Rowntree, J., Powers, W. J., and Carmichael, D. (2015a). Pasture-derived greenhouse gas emissions in cow-calf production systems. *Journal of Animal Science, 93*(3), 1350.

Chiavegato, M. B., Rowntree, J., Powers, W. J., and Carmichael, D. (2015b). Enteric methane from lactating beef cows managed with high- and low-input grazing systems. *Journal of Animal Science, 93*(3), 1365-1375.

Chiavegato, M. B., Rowntree, J., Powers, W. J., and Carmichael, D. (2015c). Carbon flux assessment in cow-calf grazing systems. *Journal of Animal Science, 93*(8), 4189-4199.

Conant, R. T., Six, J., and Paustian, K. (2003). Land use effects on soil carbon fractions in the southeastern United States. I. Management-intensive versus extensive grazing. *Biology and Fertility of Soils, 38*(6), 386-392.

Cook, R., and Weller, D. (2004). *In defense of crop monoculture.* Paper presented at the Proceedings 4th International Crop Science Congress'.(The Regional Institute Ltd: Gosford, NSW) Available at: www.cropscience.org.au/icsc2004/symposia/2/1/1128_cookrj. htm.

FAO. (2012). *Livestock and Landscapes*. Food and Agriculture Organization of the United Nations.

Gerber, P. J., Steinfeld, H., Henderson, B., Mottet, A., Opio, C., Dijkman, J., Falcucci, A. and Tempio, G. (2013). *Tackling climate change through livestock: a global assessment of emissions and mitigation opportunities*. Retrieved from Rome.

Gerrish, J. (2004). *Management-Intensive Grazing: The Grassroots of Grass Farming*: Green Park Press.

Gustavo, G. T. C., Ryan, M. R., and Richard, T. L. (2013). Energy Use and Greenhouse Gas Emissions from Crop Production Using the Farm Energy Analysis Tool. *BioScience, 63*(4), 263-273.

Hall, C. A. S., Pimentel, D., and Dale, B. E. (2011). Seeking to Understand the Reasons for Different Energy Return on Investment (EROI) Estimates for Biofuels. *Sustainabilitty, 3*(12), 2413-2432.

IPCC. (2006). *IPCC Guidelines for Greenhouse Gas Inventories*. Retrieved from http://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/4_Volume4/V4_10_Ch10_Livestock.pdf

IPCC. (2014). Anthropogenic and Natural Radiative Forcing. In G. Myhre, D. Shindell, F.-M. Bréon, W. Collins, J. Fuglestvedt, J. Huang, D. Koch, J.-F. Lamarque, D. Lee, B. Mendoza, and A. R. T. Nakajima, G. Stephens, T. Takemura and H. Zhang (Eds.), *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*: Cambridge University Press.

Janzen, H. H. (2011). What place for livestock on a re-greening earth? *Animal Feed Science and Technology, 166*, 783-796.

Kim, S., and Dale, B. E. (2008). Life cycle assessment of fuel ethanol derived from corn grain via dry milling. *Bioresource Technology, 99*(12), 5250-5260.

Liebig, M. A., Gross, J. R., Kronberg, S. L., Phillips, R. L., and Hanson, J. D. (2010). Grazing management contributions to net global warming potential: a long-term evaluation in the Northern Great Plains. *Journal of Environmental Quality, 39*(3), 799-809.

Lupo, C. D., Clay, D. E., Benning, J. L., and Stone, J. J. (2013). Life-Cycle Assessment of the Beef Cattle Production System for the Northern Great Plains, USA. *Journal of Environmental Quality, 42*(5), 1386-1394.

Machmuller, M. B., Kramer, M. G., Cyle, T. K., Hill, N., Hancock, D., and Thompson, A. (2015). Emerging land use practices rapidly increase soil organic matter. *Nature Communications, 6*, 6995.

Mathew, I., Shimelis, H., Mutema, M., and Chaplot, V. (2017). What crop type for atmospheric carbon sequestration: Results from a global data analysis. *Agriculture Ecosystems & Environment, 243*, 34-46.

McSherry, M. E., and Ritchie, M. E. (2013). Effects of grazing on grassland soil carbon: a global review. *Global Change Biology, 19*(5), 1347-1357.

Minasny, B., Malone, B. P., McBratney, A. B., Angers, D. A., Arrouays, D., Chambers, A., . . . Winowiecki, L. (2017). Soil carbon 4 per mille. *Geoderma, 292*, 59-86.

National Academies of Sciences, E. a. M. (2016). *Nutrient Requirements of Beef Cattle* (Vol. Eighth Revised Edition). Washington, DC: National Academies Press.

Oates, L. G., Undersander, D. J., Gratton, C., Bell, M. M., and Jackson, R. D. (2011). Management-Intensive Rotational Grazing Enhances Forage Production and Quality of Subhumid Cool-Season Pastures. *Crop Science, 51*(2), 892-901.

OECD. (2016). Meat Consumption. Retrieved from https://data.oecd.org/agroutput/meat-consumption.htm

Olson, K. R., Al-Kaisi, M., Lal, R., and Cihacek, L. (2016). Impact of soil erosion on soil organic carbon stocks. *Journal of Soil and Water Conservation, 71*(3), 61-67A.

Olson, K. R., Al-Kaisi, M., Lal, R., and Morton, L. W. (2017). Soil ecosystem services and intensified cropping systems. *Journal of Soil and Water Conservation, 72*(3), 64A-69A.

Pelletier, N., Pirog, R., and Rasmussen, R. (2010). Comparative life cycle environmental impacts of three beef production strategies in the Upper Midwestern United States. *Agricultural Systems, 103*(6), 380-389.

Powlson, D. S., Whitmore, A. P., and Goulding, K. W. T. (2011). Soil carbon sequestration to mitigate climate change: a critical re-examination to identify the true and the false. *European Journal of Soil Science, 62*(1), 42-55.

Rochette, P., Worth, D. E., Lemke, R. L., McConkey, B. G., Pennock, D. J., Wagner-Riddle, C., and Desjardins, R. (2008). Estimation of N2O emissions from agricultural soils in Canada. I. Development of a country-specific methodology. *Canadian Journal of Soil Science, 88*(5), 641-654.

Rowntree, J. E., Ryals, R., DeLonge, M., Teague, W. R., Chiavegato, M., Byck, P., . . . Xu, S. (2016). Potential mitigation of midwest grass-finished beef production emissions with soil carbon sequestration in the United States of America. *Future of Food: Journal on Food, Agriculture and Society, 4*(3), 31-38.

Ryan, B., and Tiffany, D. G. (1998). Energy use in Minnesota agriculture. *Minnesota Agricultural Economist*(693), 1, 4-7.

Savory, A., Butterfield, J. (1998). *Holistic Management: A New Framework for Decision Making* (2nd ed.): Island Press.

Schader, C., Muller, A., Scialabba, N. E.-H., Hecht, J., Isensee, A., Erb, K.-H., . . . Niggli, U. (2015). Impacts of feeding less food-competing feedstuffs to livestock on global food system sustainability. *Journal of the Royal Society, Interface / the Royal Society, 12*(113).

Schroeder, J. W. (2012). Weights and Measures of Common Feeds. Retrieved from https://www.ag.ndsu.edu/pubs/ansci/livestoc/as1282.pdf

Sloggett, G. (1992). Estimating Energy Use in World Irrigation. In R. Fluck (Ed.), *Energy in Farm Production* (Vol. 6): Institute of Food and Agricultural Sciences.

Stackhouse-Lawson, K. R., Rotz, C. A., Oltjen, J. W., and Mitloehner, F. M. (2012). Carbon footprint and ammonia emissions of California beef production systems. *Journal of Animal Science, 90*(12), 4641-4655.

Stewart, C. E., Paustian, K., Conant, R. T., Plante, A. F., and Six, J. (2007). Soil Carbon Saturation: Concept, Evidence and Evaluation. *Biogeochemistry, 86*(1), 19-31.

Syswerda, S. P., Corbin, A. T., Mokma, D. L., Kravchenko, A. N., and Robertson, G. P. (2011). Agricultural Management and Soil Carbon Storage in Surface vs. Deep Layers. *Soil Science Society of America Journal, 75*(1), 92-101.

Teague, W., Apfelbaum, S., Lal, R., Kreuter, U., Rowntree, J., Davies, C., . . . Wang, T. (2016). The role of ruminants in reducing agriculture's carbon footprint in North America. *Journal of Soil and Water Conservation, 71*(2), 156-164.

Teague, W. R., Dowhower, S. L., Baker, S. A., Haile, N., DeLaune, P. B., and Conover, D. M. (2011). Grazing management impacts on vegetation, soil biota and soil chemical, physical and hydrological properties in tall grass prairie. *Agriculture, Ecosystems and Environment, 141*(3), 310-322.

UGA. (2015). What Is Management-Intensive Grazing? Retrieved from http://www.caes.uga.edu/topics/sustainag/grazing/

USDA. (2012). *Sheet, Rill and Wind Erosion*.

USDA. (2014). *2012 Census of Agriculture*.

USDA. (2015a). *Crop Production*. National Agricultural Statistics Service.

USDA. (2015b). *Summary Report: 2012 National Resources Inventory*. Center for Survey Statistics and Methodology.

USDA. (2016a). Corn Background. Retrieved from http://www.ers.usda.gov/topics/crops/corn/background.aspx

USDA. (2016b). *Overview of the United States Cattle Industry*.

Voisin, A. (1959). *Grass Productivity*: Island Press.

Wang, T., Teague, W. R., Park, S. C., and Bevers, S. (2015). GHG Mitigation Potential of Different Grazing Strategies in the United States Southern Great Plains. *Sustainability, 7*(10), 13500-13521.

West, P. C., Gerber, J. S., Engstrom, P. M., Mueller, N. D., Brauman, K. A., Carlson, K. M., . . . Siebert, S. (2014). Leverage points for improving global food security and the environment. *Science (New York, N.Y.), 345*(6194), 325-328.

CHAPTER 3

CONCLUSIONS AND FUTURE RESEARCH

The results of this study indicate that AMP grazing can potentially sequester SOC and offset GHG emissions for the finishing stage of beef production in the Upper Midwest. However, other stages of cattle production also emit GHGs and can have negative environmental impacts. Namely, the cow-calf stage contributes most to cradle-to-gate emissions. Although we only examined the impacts of AMP grazing compared with the feedlot system in the finishing stage, this production strategy can also be implemented in the cow-calf stage and therefore may result in further GHG mitigation. Studies examining the environmental impacts for the entire cradle-to-gate lifespan of cattle using AMP grazing throughout would be warranted.

In this study, we were lucky to have one of the few longer term SOC data sets including baseline SOC stocks for our perusal. Moving forward, to continue to include SOC as an environmental indicator into beef-cattle LCAs in other eco-regions, more robust SOC accounting and datasets will be absolutely vital. It is possible that the sequestration rate observed in this study may not apply in other climates (i.e., in more arid regions), other soil types, and different plant species combinations where SOC sequestration capacity is limited. Results of such information could influence the regionality of beef production in the future, and possibly be instrumental in creating production systems in which beef production is only sustained in areas where its environmental impacts may be more optimally compensated for with SOC sequestration.

Another important area of further research is the study of additional ecosystem services that are underrepresented by the current LCA literature. Aside from GHG emissions, beef production can result in environmental impacts that include deforestation and land-use change, soil health impacts such as water holding capacity, biodiversity and cation exchange capacity, marine and freshwater eutrophication, and soil erosion and sedimentation. These ecosystem services are often hard to understand and may have a variety of effects, making them both difficult to study and to include in LCA. However, considering that these ecosystem services have a multitude of both environmental and economic benefits (i.e., food provisioning, nutrient cycling, water infiltration and ecosystem resiliency), they represent an important area of needed study in the future. Full-cost accounting is one method to incorporate these impacts. Monetization of externalized costs borne by beef production and antecedent feed production may present an opportunity to examine these impacts on a dollar basis, which may generate a holistic indicator of sustainability further than what can be deduced currently based only on GHG emissions.

Lastly, the environmental improvements that may be generated through AMP grazing are strictly hypothetical if this practice is not more widely adopted in the future. Even though we have some evidence, scientifically, what these practices may contribute to environmental preservation and GHG mitigation, policy implications are hard to understand and are often not the focus of implicative research. It is vital for scientific findings, such as those presented in this study, to influence environmental and agricultural policy in the future. Currently, continuous grazing is the most common "grass-fed" technique, globally, while conventional feedlot finishing is the most common production technique domestically. However, if one goal is to increase the scale of adoption of regenerative and less environmentally harmful practices, such as AMP grazing, we must better understand two things: 1) how these research findings may translate into policy; and, 2) how these policies will impact food security, consumer perception, and farmer adoptability.

Knowledge gained from more robust soil C datasets and ecosystem service dynamics will help formulate sound policy regarding beef production practices. Additionally, future research may give light to cropping mechanisms and systems that may result in greater soil formation and SOC sequestration. Even if soil C saturation does limit the time-scale that AMP grazing will provide a GHG sink, the possibility that it may serve as a short-run mitigation technique while better renewable energy technologies are developed, remains a critically important consideration and potentially viable alternative.