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Impacts of soil carbon sequestration on life cycle greenhouse gas emissions in Midwestern USA beef finishing systems

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ABSTRACT

Beef cattle have been identified as the largest livestock-sector contributor to greenhouse gas (GHG) emissions. Using life cycle analysis (LCA), several studies have concluded that grass-finished beef systems have greater GHG intensities than feedlot-finished (FL) beef systems. These studies evaluated only one grazing management system – continuous grazing – and assumed steady-state soil carbon (C), to model the grass-finishing environmental impact. However, by managing for more optimal forage growth and recovery, adaptive multi-paddock (AMP) grazing can improve animal and forage productivity, potentially sequestering more soil organic carbon (SOC) than continuous grazing. To examine impacts of AMP grazing and related SOC sequestration on net GHG emissions, a comparative LCA was performed of two different beef finishing systems in the Upper Midwest, USA: AMP grazing and FL. We used on-farm data collected from the Michigan State University Lake City AgBioResearch Center for AMP grazing. Impact scope included GHG emissions from enteric methane, feed production and mineral supplement manufacture, manure, and on-farm energy use and transportation, as well as the potential C sink arising from SOC sequestration. Across-farm SOC data showed a 4-year C sequestration rate of 3.59 Mg C ha^{-1} yr^{-1} in AMP grazed pastures. After including SOC in the GHG footprint estimates, finishing emissions from the AMP system were reduced from 9.62 to −6.65 kg CO_2-e kg carcass weight $(CW)^{-1}$, whereas FL emissions increased slightly from 6.09 to 6.12 kg CO_2-e kg CW^{-1} due to soil erosion. This indicates that AMP grazing has the potential to offset GHG emissions through soil C sequestration, and therefore the finishing phase could be a net C sink. However, FL production required only half as much land as AMP grazing. While the SOC sequestration rates measured here were relatively high, lower rates would still reduce the AMP emissions relative to the FL emissions. This research suggests that AMP grazing can contribute to climate change mitigation through SOC sequestration and challenges existing conclusions that only feedlot-intensification reduces the overall beef GHG footprint through greater productivity.

1. Introduction

Beef production can be an environmentally deleterious process, leading to high GHG emissions and land degradation, along with feed-food competition. Depending on the accounting approach and scope of emissions included, estimates by various sources (IPCC, FAO, EPA and others) place the contribution of livestock as a whole to global anthropogenic GHG emissions at 7–18%. The United States (U.S.) is the leading beef producer (19% of world production) and among top beef consumers globally (an average of 25 kg per person per year in 2017) (OECD, 2016). In addition, beef consumption is growing globally as the nutrition transition towards greater meat consumption continues in many countries (OECD, 2016). Therefore, producing beef with less GHG emissions (reducing GHG intensity) is of interest both globally and domestically. Life cycle assessment (LCA), the most common approach to GHG emissions accounting, has been used to estimate environmental impacts of beef production.

In previous beef LCA literature, grass-fed (over the entire life cycle) or grass-finished (referring exclusively to the finishing stage) systems are often modeled using simplified grazing parameters typically

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representative of continuous grazing, a simplistic management strategy in which cattle graze the same pasture continuously through an entire grazing season (Crosson et al., 2011; de Vries et al., 2015). This grazing management approach, while still the most common, can negatively impact plant regrowth and recovery, as well as plant communities, and has low productivity (Oates et al., 2011). Grazing management techniques vary greatly, however, ranging from continuous to light rotational to intensively managed. Accordingly, the land, ecosystem, and GHG emission impacts resulting from beef production are highly dependent on the type of grazing management system utilized (Brilli et al., 2017; Rowntree et al., 2016). Additionally, because grass-fed beef production has been increasing in response to United States consumer demand in recent years (Stone Barn Center, 2017), it would be useful to explore the environmental impacts of alternative grass-finishing systems. Some literature has identified beneficial ecosystem services resulting from the adoption of a carefully managed system known as adaptive multi-paddock (AMP) grazing. This approach applies an adaptive strategy that incorporates short grazing intervals with relatively high animal stocking densities, which are designed to allow plant recovery, promoting optimal plant communities and protecting soils (Conant et al., 2003; Teague and Barnes, 2017). These principles were conceptualized by Voisin (1959) as "rational grazing" and have also been embraced within grazing systems such as "holistic planned grazing" (Savory and Butterfield 1998) and "management-intensive grazing" (Gerrish, 2004). Potential AMP grazing benefits include reductions in overgrazing and soil erosion, improved forage utilization and animal productivity, and increased soil carbon (C) sequestration, which might reduce net GHG emissions (Teague et al., 2016).

Soil C sequestration is a critical ecosystem service of grasslands, which can be maximized using best management practices for livestock grazing (Griscom et al., 2017; Liebig et al., 2010; McSherry and Ritchie, 2013; Wang et al., 2015). However, there remains substantial uncertainty about soil C change over time in managed grasslands (Desjardins et al., 2012; Olson et al., 2017; Paustian et al., 2016), with possible limitations to soil C storage related to C and N cycling, including soil N limitations (van Groenigen et al., 2017). Additionally, protecting long-term soil C storage is contingent upon preventing land-use change (Petersen et al., 2013). For these reasons, beef LCAs often assume soil C equilibrium. Given critical relationships between agricultural management and the terrestrial C pool (Olson et al., 2017), as well as the extensiveness of grazing lands (~336 million ha of land in the United States; (Chambers et al., 2016)) and their importance to livelihoods (Asner et al., 2004; Briske et al., 2015; Desjardins et al., 2012), grassland C sinks might represent a significant GHG mitigation strategy that should be included in beef production models. The few studies that considered low rates of soil C sequestration in GHG accounting for beef production indicated potential emissions decreases of 24–535% (Beauchemin et al., 2010; Lupo et al., 2013; Nguyen et al., 2010; Pelletier et al., 2010). Although many used modeled C sequestration from beef simulation studies (Alemu et al., 2017; Beauchemin et al., 2011; Nguyen et al., 2010, 2012), such estimates might not represent actual, on-farm changes in SOC (Petersen et al., 2013; Teague et al., 2011). This need for on-farm SOC data was discussed by Griscom et al. (2017), who identified AMP grazing as a potentially important climate change mitigation strategy, but were unable to include it in their analysis due to lack of robust data.

Previous LCAs have compared feedlot to grass-finishing strategies. Worth noting is that both feedlot- and grass-finishing systems follow similar management practices in the two previous phases of production (cow-calf and backgrounding). A majority of GHG emissions are attributed to the cow-calf sector (Beauchemin et al., 2010). However, most of the differences in beef production environmental impact arise from the finishing strategy employed. An estimated 97% of cattle are feedlot-finished in the U.S., while the remaining 3% are broadly "grass-finished," irrespective of management (Stone Barns Center, 2017). Many studies indicate that feedlot finishing systems have lower cradle-to-gate GHG emissions per kg of carcass weight because grass-fed systems have greater enteric methane (CH_4) emissions (due to microbial ruminal fermentation), attributed to the more fibrous diet and longer finishing times, and lower overall carcass weights (Capper, 2012; Desjardins et al., 2012; Lupo et al., 2013; Pelletier et al., 2010; Stackhouse-Lawson et al., 2012; Swain et al., 2018). However, as noted above, many of these studies did not consider the potential for soil C sequestration in well-managed grasslands, and emissions from feedlot finishing might be underestimated due to a lacking representation of soil changes during feed production, such as soil erosion (Janzen, 2011). From 1982 to 2012, 6.07 million hectares of "prime farmland" in the U.S. were lost due to soil erosion, and currently 4.2 Mg ha^{-1} yr^{-1} are still lost from cropland (USDA, 2015; 2012). Because soil organic matter (SOM) consists of 40–75% C, erosion constitutes a significant loss of soil fertility and water-holding capacity and can contribute to GHG emissions. Furthermore, livestock consume about one-third of all grain produced globally and in the U.S. (FAO, 2012; Schader et al., 2015). For these reasons, soil erosion on land used to produce feed crops is an important indicator of sustainability and should be incorporated into beef LCA accounting, but has generally been excluded. Additionally, emissions from grass-fed systems vary greatly due to differences in regional and on-farm practices. For example, different assumptions about fertilization rates on pasture have resulted in a 5-fold difference in N_2O emissions (Lupo et al., 2013; Pelletier et al., 2010). Studies have identified these gaps and have called for more robust research and inclusion of soil C in future LCA models (Lupo et al., 2013; Pelletier et al., 2010).

Considering the variability in grazing strategies and research gaps in soil C dynamics, the goal of the present study was to estimate the system GHG impacts associated with feedlot (FL) finishing and compare them with finishing using an alternative grazing technique, AMP grazing, including soil C accounting. Additionally, we aimed to answer the call for more robust data of the impacts of AMP grazing on soil C sequestration, as it may contribute to a natural climate solution (Griscom et al., 2017). To do this, an ISO-compliant partial LCA was conducted for the finishing phase of cattle production in the Upper Midwest, U.S., and combined with soil C sequestration results from 4 years of on-farm data collection in the AMP grazing scenario.

2. Materials and methods

All data were combined using a deterministic environmental impact model created in MS Excel. Emissions and land use occupation were calculated for the two comparative beef production finishing systems: FL and AMP grazing.

2.1. System boundaries

Because most management differences and much of the variability among beef production systems are concentrated within the finishing phase, system boundaries were limited to this phase only, thus excluding cow-calf and backgrounding stages. Two different finishing strategies in the Upper Midwest, FL (> 97% of all production) and AMP grazing, were modeled using a combination of on-farm data and scientific literature information. All major GHGs (CH_4, carbon dioxide (CO_2), and nitrous oxide (N_2O)), including those from enteric ruminal fermentation, manure storage and handling, feed production, and on-farm energy use, were included. Tertiary emissions, including those from manufacture of machines, equipment, and infrastructure were excluded, based on their assumed minor contributions (Lupo et al., 2013). Gasses were converted to CO_2 equivalents (CO_2-e) using their 100-year global warming potentials (CO_2 = 1, CH_4 = 34, N_2O = 298) (IPCC, 2014). For continuity and comparison with previous beef LCAs, the functional unit was 1 kg of carcass weight (CW).

Table 1
Temporal grazing and animal data for the adaptive multi-paddock grazing finishing system for each year (2012–2016).

Characteristic	2012	2013	2014	2015	2016	Mean	St. dev
Grazing began	19 May	13 May	14 May	11 May	18 May	15 May	
Grazing ended	6 Nov	4 Nov	7 Nov	19 Nov	15 Nov	11 Nov	
LW[a] at start (kg)	323	342	368	344	358	347	15
LW[a] at slaughter (kg)	560	530	499	548	513	530	22
Carcass weight (kg)	299	283	271	287	261	280	13

[a] LW = live weight.

2.2. Finishing systems

2.2.1. Adaptive multi-paddock grazing

Five-year (2012–2016) on-farm data from Michigan State University (MSU) Lake City AgBioResearch Center (Lake City, Michigan) were used for the AMP grazing finishing system. Red Angus steers (n = 210) were weaned onto grass at 7 months of age in November, with exact dates varying slightly year-to-year depending upon frost occurrence, weather and forage quality. Steers were then AMP grazed from mid-May until November, at which point off-farm alfalfa hay was purchased and offered with stockpiled forage until slaughter in early-to-mid December. On average, steers finished at 530 kg (σ = 47 kg) live weight in 200 days (average of 180 days AMP grazing and 20 days of alfalfa hay feeding). Steers were grazed on high-quality, predominately alfalfa, cool-season mix pastures at a mean stocking rate of 2.7 steers/ha and rotated among paddocks according to forage availability and quality (Table 1). More details on pasture composition is given in Appendix A.1. Rotation frequency focused on preventing overgrazing and assuring forage recovery, allowing appropriate regrowth before being grazed again. Pastures were not fertilized, irrigated or treated for pests for > 7 years prior to this management implementation in 2010. Precipitation and mean temperature for each month of the 2012–2016 grazing seasons are given in the Appendix (Table A.2.1). Average daily gain (ADG) was calculated from actual animal performance data, whereas dry matter intake (DMI) and net energy for maintenance were calculated using equations derived from the National Research Council's (NRC) *Nutrient Requirements of Beef Cattle* (National Academies of Sciences, 2016), and feed conversion ratio (= DMI/ADG) was calculated following Cassady et al. (2016). Average dressing percentage calculated from carcass data was 53% among all 5 years.

2.2.2. Feedlot

The FL finishing system was based on 2-year data (2015–2016) from a research trial at the MSU Beef Center (East Lansing, Michigan), where Red Angus steers (n = 16; subsampled from the same herd at the Lake City AgBioResearch Center as those in the AMP grazing system) were finished in the FL for 90 days. These steers were finished in 90 days due to logistics for another research project. Because 90 days is not a representative timeline for FL finishing, the observed data were used to simulate an extension from 90 to 170 days on-feed (Table A.3.1), a more common time-to-finish in the Upper Midwest and in FLs (National Academies of Sciences, 2016). While it was a trade-off not to have complete 170-day field data, we chose to use the partial data from these animals because they represent actual on-farm data from animals of the same genetic background. The 90-day on-farm data and NRC report (National Academies of Sciences, 2016) were used to predict FL performance, including DMI, ADG, and energy partitioning. Feed required for maintenance was calculated to determine net energy required for gain (NE_g) using Table 12-1 of National Academies of Sciences (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. Mean modeled finishing weight was 654 kg, similar to the 2015 average reported by USDA (2016a). Calculated DMI, ADG, and feed conversion ratio were within 0.1 kg of the values observed by Cassady et al. (2016), indicating that the simulation method was sound. Dressing percentage was assumed to be 63%.

2.3. Land occupation

The total area (ha) needed to produce feed to finish one steer in each system was calculated using their respective feed ingredient amounts. The total mixed ration of the FL was broken down into the dry matter (DM) weight of each feed ingredient fed to each steer, and the needed land area was calculated retrospectively by summing the land area (ha) needed per feed ingredient based on USDA crop yields for Michigan (USDA, 2016b). Similarly, land area was calculated for AMP grazing using average Michigan alfalfa yield for grazed pastures (USDA, 2016b) and locally sourced off-farm alfalfa hay, and a 70% grazing forage utilization rate based on on-farm observations and records (Doug Carmichael, manager, MSU Lake City AgBioResearch Center, pers. comm.).

2.4. Enteric CH_4, manure CH_4 and N_2O emissions

Enteric CH_4 emissions were modeled using IPCC (2006) Tier 2 methods. Gross energies (bomb calorimeter-derived total heat of combustion energy contents) were calculated using feed ingredients and animal characteristics from on-farm data for both systems. Default CH_4 conversion factors (Y_m) of 3.0% and 6.5% (± 1%) for FL and AMP grazed cattle, respectively, were applied. Despite the high uncertainty associated with IPCC CH_4 accounting, specifically the use of default Y_m values (Rowntree et al., 2016; Stackhouse-Lawson et al., 2012), we used this method for consistency with other studies. However, a sensitivity analysis was conducted comparing the impact of IPCC default methods to results based on SF_6 tracer gas data from the Lake City AgBioResearch Farm. Sensitivity of enteric CH_4 emissions to a reduction of IPCC default Y_m from 6.5 to 5.5. to represent an increase in forage quality, was also considered (see Appendix, section A.5.).

Manure CH_4 and N_2O emissions were estimated using IPCC (2006) Tier 2 methods. During FL finishing at the MSU Beef Center, manure was collected beneath the slatted floors of confinement pens for approximately 1 year before being pumped 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 FL, and for the land application phase. Although wheat was not a feed ingredient in the FL diet, manure spread on these fields still offsets synthetic fertilizers. To calculate these offsets, nitrogen (N) losses from manure were subtracted from excreted N. According to these calculations and current synthetic N application rates, manure-derived N application was calculated to reduce needed N inputs by 31.4%. Additional details on fertilizer offset calculations are given in the Appendix. All manure during AMP grazing finishing was deposited on pasture, and therefore emissions were coordinated using IPCC (2006) models for emissions from managed soils. Soil N dynamics for the pasture itself were not modeled, under the assumption that these emissions do not significantly change in response to animal production. Additionally, because no N fertilizer was applied in the AMP grazing system and manure was not collected and applied to other cropland, no fertilizer offsets were assumed. Soil N emissions from feed production (i.e., from synthetic fertilizer) were accounted for in feed emissions.

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 FL and the AMP grazing finishing systems. Because both on-farm and regional data were available, direct and indirect manure N_2O emissions were calculated using IPCC (2006) Tier 2 methods. Using actual weight gain, dietary NE_g and crude protein, excreted N (N_{ext}) was calculated and used to determine direct N_2O emissions from manure. Volatized N, which contributes to indirect N_2O emissions, was calculated using the default emissions factor (EF_4) of 0.010 kg N_2O-N because of the relatively small uncertainty (± 0.002–0.05) (IPCC, 2006). Indirect N_2O emissions from leaching and runoff (NH_3 and NO_x) were calculated using the regression equation of Rochette et al. (2008), using a relationship between precipitation (P) and potential evapotranspiration (PET). P and PET data were collected from the MSU Enviro-weather database and averaged for the 2015 and 2016 growing seasons (May 15 - Oct 15). The proportion of N leached from manure ($Frac_{leach}$) was consequently calculated as 0.28 kg N leached per kg N excreted, which is double that calculated by Lupo et al. (2013) using the same method, likely because of the greater P and PET in the Upper Midwest than in the Northern Great Plains.

Manure CH_4 emissions and direct and indirect N_2O emissions were converted to a CO_2-e basis for both finishing strategies and summed across the two manure management stages (slatted floor barn and land application) for the FL-finished cattle.

2.5. Feed production

GHG emissions from FL-feed inputs were estimated using the nutrient and energy composition of diet ingredients and their proportions in the rations fed at the MSU Beef Center. The five ration ingredients were corn grain, high moisture corn (HMC, also known as high moisture maize, HMM), corn silage, alfalfa hay, and dried distillers grains with solubles (DDGS, from corn) (Table A.3.2.). Crop yields were derived from Michigan-specific crop data according to USDA (2016b), except for that of HMC, which came from Schroeder (2012) because of data availability. The ration varied between years, with more DDGS included in 2015 and more HMC in 2016. To account for the difference in needed land area in 2015 and 2016, the area was averaged between years to obtain a more standard representation.

Because emissions data from cultivation of feed ingredients and manufacture of feed were not available for our study region (except for DDGS), we used the Farm Energy Analysis Tool, an open-source database GHG emission model calibrated with an extensive crop production literature database (Camargo et al., 2013). GHG emissions resulting from transportation of feed ingredients from farms to the feed mill, grain 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 find a GHG footprint of HMC in the literature, its emissions were assumed to equal those of corn grain. GHG emissions from production of DDGS were taken from a Michigan model (Kim and Dale, 2008), because of the proximity of the study to the Lake City AgBioResearch Center; this model included corn crop production and dry-milling, but not ethanol distillation. It was estimated that each 25.4 kg of corn used in dry-mill ethanol production generates 7.89 kg of DDGS (USDA, 2016a). GHG emissions from the manufacture of mineral supplements for rations were predicted using methodology of Lupo et al. (2013) and were scaled to represent the difference in finishing time for both finishing systems.

GHG emissions from crop irrigation were not included because < 10% of all cropland in Michigan is irrigated (USDA, 2014). Additionally, GHG emissions associated with land-use change, either for feed production or grazing land, were not considered, under the assumption that feed crops were produced on existing cropland and that 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 with synthetic fertilizers, or treated for pests, and therefore no GHG emissions were generated from these processes. GHG emissions resulting from off-farm alfalfa hay production used during the non-grazing season were taken from Camargo et al. (2013).

2.6. On-farm fuel use and transportation

As Pelletier et al. (2010) did, 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) calculation approach. All feed inputs for both systems were assumed to be transported 30 km by truck, except for DDGS, which were assumed to be transported 60 km to account for both transportation between manufacturers and to the FL, with a load capacity of 23,000 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).

2.7. Soil C

2.7.1. Sample collection and C analysis

To determine 4-year soil C sequestration, soil samples were collected at the Lake City AgBioResearch Center. The permanent pastures (grasslands) were established approximately 30 years ago and were continuously grazed and hayed until 2010 when AMP management was initiated. Seventy percent of the LCRC soil-type is sandy-loam (SL), and only SL soils were sampled for soil C 2012, accounting for a majority of farm landscape (Chiavegato et al., 2015b). In 2016, a more robust soil monitoring protocol was implemented and soils were sampled at nine transects, representing three soil types: sandy (S), clay loam (CL) and SL. Soil samples were collected within 50 m of two of the three 2012 sites. We sampled at 100 m distance from the third 2012 site due to an alfalfa seeding which disrupted the original site. Despite the distance in monitoring sites, the soil types were consistent and reflective in soil C change overtime. We acknowledge the locational variation between sampling sites could influence the results. For each site and in both years, 10 soil sub-samples were collected at four 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. To determine soil C stock, analysis was conducted as reported by Chiavegato et al. (2015b). Combined samples from each depth were sieved at 2 mm and then dried at 50 °C separately by depth until reaching a constant weight. Soils were then ground using a ball-mill grinder and analyzed for C using an ECS 4010 CHNSO Elemental Combustion System (Costech, Valencia, CA, U.S.).

To calculate bulk density, three additional sets of samples per site were collected using a 9120-Rap Powerprobe hydraulic soil sampler (AMS, American Falls, ID, U.S.). A 5.08 cm^2 core was drilled the entire 0–30 cm depth and cut into the respective depth fractions. The soils were weighed and then dried at 50 °C 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. Soil C stock was calculated as the mass of C per ha (calculated from the 0–30 cm samples) multiplied by the land area (ha).

Once soil C stocks were calculated for all 3 soil types (S, SL and CL) in 2016, they were compared to the 2012 baseline C stock of the SL soils to estimate C sequestration from 2012 to 2016. The C contents in the 2016 SL soils were slightly greater than the mean of those of all three soil types. Therefore, to estimate SOC sequestration conservatively, we averaged mean soil C stock from all 2016 sites before comparing them to the 2012 (SL) mean soil C stock.

2.7.2. Soil erosion

In lieu of on-farm data for soil erosion and SOM from cropland used to grow feed ingredients, representative literature values were used. For the FL scenario, we assumed an intensive cropping system with no soil C sequestration, as croplands are often eroded (Izaurralde et al., 2007; Olson et al., 2016). According to the USDA (2015), mean sheet and rill soil erosion on cropland in Michigan is 8.12 Mg ha^{-1} yr^{-1}. Characteristic SOM data were taken from long-term ecological research at the MSU Kellogg Biological Station (Syswerda et al., 2011). Mean C content for soils under a 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). Based on research by Lal (2003), 20% of eroded SOC was assumed to be emitted into the atmosphere. The soil erosion rate was multiplied by this factor and the C content to calculate total C loss, which was then converted to a CO_2-e basis (Table A.6.1). The resulting kg CO_2-e from soil erosion was then added to the total GHG emission footprint for FL. Because our data indicated C sequestration in the AMP grazing scenario over the 4-year interval, we assumed zero soil erosion based on Olson et al. (2016).

2.7.3. Soil CO_2-equivalent flux

Corresponding soil C sequestration was combined with emissions from each beef production system (FL or AMP grazing) to estimate net emissions from that system. To achieve this, soil C mass (Mg C ha^{-1}) was combined for the entire 0–30 cm depth for both 2012 and 2016, the increase from 2012 to 2016 was calculated, and these increases were divided by 4 to estimate Mg C sequestered ha^{-1} yr^{-1}, assuming linear sequestration over that time. This value was then divided by the grassland area needed (kg CW ha grassland^{-1}) for animals grazing and production (excluding land needed for supplemental alfalfa hay) and converted to CO_2-e. We assumed that soil C was at a steady state on the land used to grow hay. Net GHG flux is the net summation of CO_2-e arising from beef production and soil C emissions through soil erosion, as well as net reductions through soil C sequestration.

3. Results

3.1. Animal production

Animal production characteristics for FL and AMP resulting from the LCA simulations using averaged 5-year data (Table 1) are presented in Table 2. On average, cattle in the FL finished in 29 less days with CWs 45% greater than those of AMP grazed cattle. Average daily gain was 100% greater and DMI 10% lower for steers in the FL than those in the AMP grazing scenario. This resulted in a feed conversion ratio that was 56% lower for FL than for AMP grazed finishing.

3.2. Land occupation

FL and AMP finishing produced 1655 and 751 kg CW ha feed^{-1}, respectively. For the FL, this represents a 120% greater productivity on a CW basis per ha compared to AMP. In other words, 0.30 and 0.67 ha was required to produce 500 kg beef CW in the FL and AMP grazing systems, respectively.

Table 2
Animal production characteristics from feedlot and adaptive multi-paddock (AMP) grazing finishing strategies.

Characteristic	Feedlot[a]	AMP[b]
Days in the finishing phase	171.5	200.8
Beginning-Ending live weight (kg)	361–654	362–528
Dressing percentage[a]	63%	53%
Mean carcass weight (kg)	406	280
Average daily gain (kg)	1.8	0.9
Dry matter intake (kg/d)	9.9	11.0
Feed conversion ratio[b]	5.7	13.0

[a] The percentage of live weight kept as in carcass weight, after removal of selected parts.
[b] Feed conversion ratio was calculated by dividing DMI by ADG (Cassady et al., 2016).

Table 3
Greenhouse gas (GHG) emissions (kg CO_2-e) per steer in the feedlot and adaptive multi-paddock grazing finishing strategies by impact category, and their percentages of total emissions.

Impact category by production system	GHG emissions (kg CO_2-e steer^{-1})	% of total
Feedlot	2470.4	
Enteric CH_4	777.0	31%
Manure emissions[a]	732.7	30%
Feed emissions[b]	905.4	37%
Mineral supplement emissions[c]	1.3	0.05%
On-farm energy and transportation	54.1	2%
Carcass weight (kg)	405.8	
Total (kg CO_2-e kg carcass weight)	**6.09**	
Adaptive multi-paddock grazing	2694.6	
Enteric CH_4	1434.1	53%
Manure emissions[a]	688.3	26%
Feed emissions[b]	512.0	19%
Mineral emissions[c]	0.8	0.03%
On-farm energy and transportation	59.5	2%
Carcass weight (kg)	280.2	
Total (kg CO_2-e kg carcass weight)	**9.62**	

[a] Predominately indirect N_2O, but also includes manure-derived CH_4.
[b] Calculated from respective feed components combined with fertilizer using the FEAT model (Camargo et al., 2013).
[c] Calculated using data from Lupo et al. (2013).

3.3. GHG emissions

Total GHG emissions for each finishing scenario are reported in Table 3 and Fig. 1. After accounting for the 31.4% fertilizer offset (the percentage of synthetic fertilizer N that was replaced by land-applied manure) in FL finishing, estimated GHG emissions associated with FL and AMP finishing were 6.09 and 9.62 kg CO_2-e kg CW^{-1}, respectively. Enteric CH_4 was the largest source for AMP and intermediary for FL finishing, contributing 53% and 31%, respectively. Feed GHG emissions were the largest source in the FL, 77% greater than those in AMP finishing and a greater percentage of total emissions (37% and 19%, respectively). Manure emissions (comprised of both N_2O and CH_4) represented 30% of FL emissions and 26% of AMP emissions. GHG emissions from mineral supplement production and on-farm energy and transportation together contributed $<$ 2.5% to overall emissions in both finishing scenarios.

3.4. Soil C sequestration

When compared to the 2012 baseline soil C stock of the SL soil (33.97 $\pm$ 0.71 Mg C ha^{-1} in the top 30 cm), 2016 soil C stocks were 15.18, 8.16, and 19.75 Mg C ha^{-1} greater in SL, S, and CL soils, respectively. These data indicate a 4-year sequestration rate of 3.79 Mg C ha^{-1} yr^{-1} for the SL soils. If the S and CL soil types had had a similar 2012 baseline, however, their sequestration rates would have been 2.04 and 4.94 Mg C ha^{-1} yr^{-1}, respectively. Averaging data for all three soil types (for S, SL and CL) led to an estimated sequestration rate of 3.59 Mg C ha^{-1} yr^{-1} (Table 4). To be conservative for the purposes of the LCA, we used this lower sequestration rate for the AMP grazing scenario.

3.5. Net GHG flux

Soil erosion due to feed production in the FL scenario contributed 22.76 kg CO_2-e (0.03 kg CO_2 kg CW^{-1}). Total C sequestration was 3.59 Mg C ha^{-1} yr^{-1}, equally $-$17.54 kg CO_2-e CW^{-1}, assuming this C sequestration was possible for all the land in the AMP grazing finishing scenario. However, after subtracting the land used for off-farm alfalfa

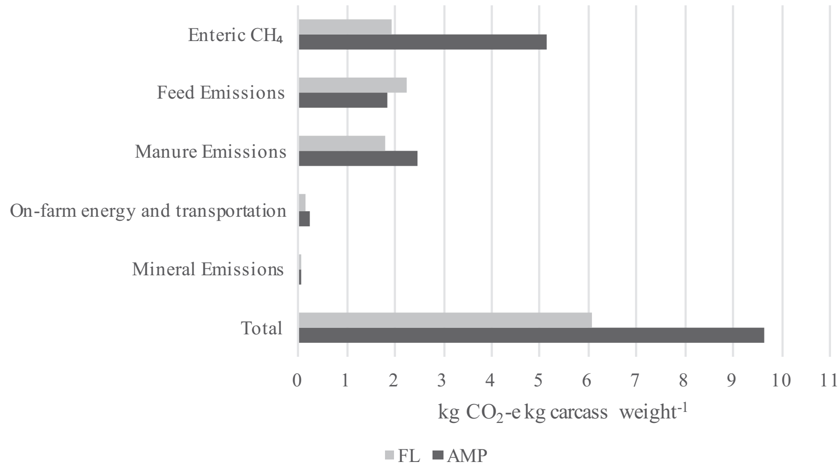

Fig. 1. Greenhouse gas (GHG) emissions (kg CO_2-e kg CW^{-1}) by emissions category for feedlot (FL) and adaptive multi-paddock (AMP) grazing systems.

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

Soil type	Soil C stock (Mg C ha^{-1})			
	2012		2016	
	Mean	Std. error	Mean	Std. error
Sandy	–	–	42.13	1.59
Sandy loam	**33.97**	0.71	49.15	6.55
Clay loam	–	–	53.72	7.33
All	–	–	**48.33**	**1.95**

Soil type	Soil C sequestration (Mg C ha^{-1})			
	4-year increase	Mean annual increase	Mean	Std. error
Sandy	8.16	2.04	–	–
Sandy loam	15.18	3.79	–	–
Clay loam	19.75	4.94	–	–
All	14.36	**3.59**	3.59	0.84

The bolded numbers represent those used for numerical comparison.

hay production (7.2% of total land required in the AMP grazing scenario), assuming no net SOC changes on that land, the final net value associated with C sequestration was −16.27 kg CO_2-e CW^{-1}. Combined with 9.62 kg CO_2-e kg CW^{-1} in emissions, this resulted in a net GHG sink of −6.65 kg CO_2-e kg CW^{-1} (Fig. 2). The 6.09 kg CO_2-e kg CW^{-1} of FL emissions combined with 0.03 kg CO_2-e kg CW^{-1} from erosion resulted in a net emission of 6.12 kg CO_2-e kg CW^{-1} for the FL scenario (Fig. 2).

3.6. Enteric CH_4 sensitivity analysis

Our sensitivity analysis indicates that using the on-farm SF_6 tracer gas data instead of IPCC default Y_m would reduce the enteric CH_4 footprint in the AMP grazing scenario by 36%, representing a decrease from 42 to 27 kg CH_4 $steer^{-1}$, and a reduction in overall GHG footprint of 1.81 kg CO_2-e kg CW^{-1} (19% decrease) (Fig. 3, Table A.5.1). Similarly, if Y_m had been reduced from the default of 6.5 (± 1.0) to 5.5 to reflect improved forage quality, enteric CH_4 emissions from the AMP grazing scenario would have been reduced by 15% (Table A.5.1).

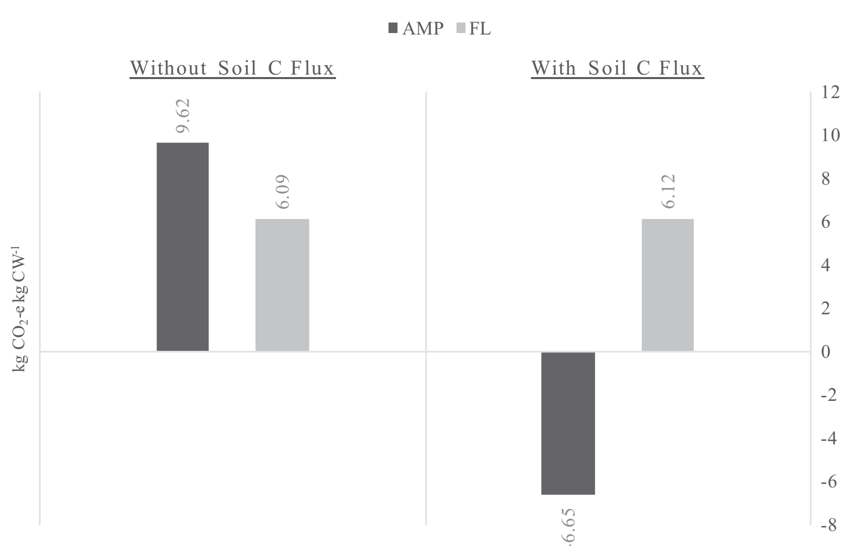

Fig. 2. Estimated emissions (kg CO_2-e kg CW^{-1}) for each finishing strategy − feedlot (FL) and adaptive multi-paddock (AMP) grazing − before (left) and after (right) net C flux from soils (sequestration and erosion) is incorporated.

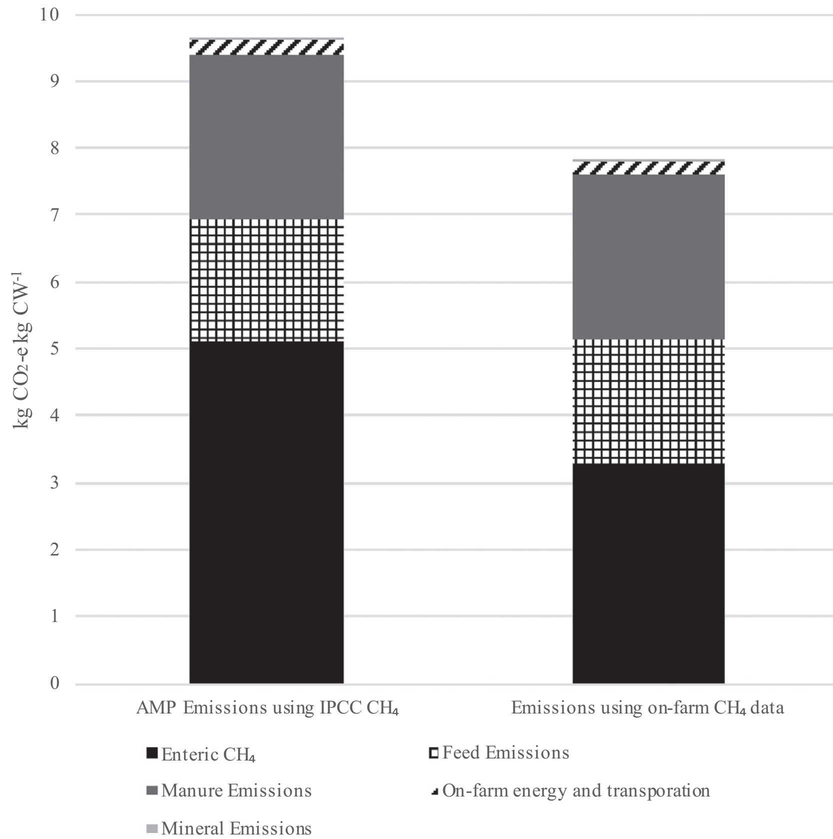

Fig. 3. Impact of methane (CH_4) emission sensitivity analysis on overall emissions for the adaptive multi-paddock (AMP) grazing system. Enteric CH_4 emissions estimated with the commonly used IPCC method (Y_m = 6.5 for grazing systems) is compared to CH_4 data collected on-farm using SF_6 tracer gas (Chiavegato et al., 2015a).

4. Discussion

4.1. Animal production

Average finishing weights for the FL steers (Table 2) were similar to those reported by others (Capper, 2012; Lupo et al., 2013; Pelletier et al., 2010). However, our 4-year AMP animal performance and days to finish were considerably different than that of previous studies (Nguyen et al., 2010; Picasso et al., 2014). Our AMP-grazed steers finished in 150 days shorter and 99 kg heavier than the continuously grazed steers in Lupo et al. (2013). This difference can be explained by improved forage quality and utilization in the AMP grazing system versus the more conventional continuous grazing system. Studies using AMP grazing indicate that it can result in more ideal pasture species composition (more digestible forages with greater digestible energy content) due to prevention of consistent species selection and overgrazing, providing pasture recovery, and re-grazing before lignification (Oates et al., 2011; Teague et al., 2016; Teague et al., 2011; UGA, 2015).

4.2. Land occupation

The result that FL requires less than half the agricultural land to produce beef versus grass-fed (in this case, AMP) is typical of assumptions or results in other studies (Capper, 2012; Lupo et al., 2013; Nguyen et al., 2010; Pelletier et al., 2010). Our findings speak to the complexity of addressing beef production systems. Should a society focus on increasing the overall production and efficiency of protein production, limiting land occupation and using greater energy inputs, or produce at a lower rate with less fossil fuel-associated emissions from feed production? Likewise, for the AMP grazing system to produce comparable amounts of beef, either more cows would be needed to produce additional animals for the system, or the cattle would have to remain in the system for a longer period of time. Either scenario would increase the overall emissions and land requirement.

Utilization of AMP grazing as a regenerative agriculture management system can improve soil ecological function, lessening damage from tillage and inorganic fertilizers and can improve biodiversity and wildlife habitat (Teague et al., 2016). While not as productive as FL based on yields, the AMP grazing system produced considerably greater amounts of beef on a land basis as compared to continuous grazing, showing that improved management can increase the output of grass-fed beef. Ultimately, in a closed system, this implies somewhat lower per capita beef consumption, but greater environmental benefits from what is consumed.

4.3. GHG emissions

Generally, GHG emissions (per kg CW) were within the range of those reported in other studies for the Northern Great Plains (Lupo et al., 2013), Upper Midwest (Pelletier et al., 2010), and California (Stackhouse-Lawson et al., 2012) within the U.S. Compared to results of the grass-finishing model of Pelletier et al. (2010), GHG emissions from finishing with AMP grazing in our study were about 45% lower. This was most likely due to their longer finishing time, lower animal productivity, and high application rate of fertilizer on pasture. Levels of N application and related assumptions about associated emissions from pasture and cropland are primary contributors to discrepancies among beef production LCA models. According to Stewart et al. (2009), reduction in fertilizer use resulted in the largest decrease (30%) in modeled overall emissions from beef production. Additionally, in the western U.S., where irrigation is more predominant, irrigation might represent an important source of GHG emissions not included in this model. According to Sloggett (1992), 23% of the on-farm energy used for crop production in the U.S. was for on-farm pumping of water.

Similar to other studies, enteric CH_4 in our study contributed the most to overall GHG emissions (Fig. 1) in the AMP grazing system. Compared to FL finishing, AMP finishing generated more than twice the enteric CH_4 emissions per kg CW (Fig. 1). This emphasizes the major benefits of utilizing more highly digestible, higher energy feeds, and a shorter finishing phase, resulting in productivity gains in our FL model. However, as shown in the sensitivity analysis (Fig. 3), it is likely that the use of IPCC (2006) enteric CH_4 accounting methodology overestimates actual enteric CH_4 emissions in the AMP grazing system, especially when more digestible, higher 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 and different management strategies in beef finishing systems.

Although estimated total manure emissions were lower for the AMP grazing scenario than the FL scenario (6.9 vs 7.3 Mg CO_2-e, respectively), the opposite was true when reflected on an animal productivity basis (CO_2-e kg CW^{-1}) (2.46 vs. 1.81 CO_2-e kg CW^{-1} for AMP grazing and FL, respectively; Fig. 1). This is the result of greater DMI and lower final CW in the AMP grazing scenario than in the FL scenario. Manure CH_4 and N_2O for the FL scenario estimated in this study were greater than those in other studies (Lupo et al., 2013; Pelletier et al., 2010). This is likely due to use of the IPCC Tier 2 methods in the current study instead of the Tier 1 methodology used in prior studies, greater precipitation in our study area than in those of the others, and handling manure in the FL as a liquid instead of a solid.

Feed production emissions in the FL finishing scenario were slightly greater than those reported by Lupo et al. (2013) and Pelletier et al. (2010), likely because our FL ration included only a small proportion of legume forages and a larger proportion of DDGS and corn grain which have greater emissions per kg than other feedstuffs (Camargo et al., 2013). Additionally, the LCA model of Camargo et al. (2013) included production of inputs and processes not included in conventional beef production LCAs, such as feed-associated pest management and transportation of inputs. Application of manure produced in the FL in the present study offset 31.4% of N fertilizer application, which is lower than the 69% used in the study of Lupo et al. (2013). This, in part, is due to the greater average fertilization rate reported by USDA (2017) used in the present study than in the 2010 USDA data used by Lupo et al. (2013). Emissions from on-farm energy use and transportation and mineral supplement production represented small percentages of total GHG emissions (< 2.5% and < 0.05%, respectively).

4.4. Soil C sequestration

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 "management intensive grazing" in the Southeastern U.S. cited by Conant et al. (2003). However, we are unsure of the forage productivity, stocking rate, and rotation intensity used in their study, but hypothesize that potentially higher stocking rates and greater forage production and utilization in our study may partially explain the higher C sequestration rate as reported by Minasny et al. (2017) and Paustian et al. (2016). Assuming management similar to that in our study, for pastures transitioned from heavy continuous grazing to AMP grazing in northern Texas, U.S., Wang et al. (2015) reported 3.53 Mg C ha^{-1} yr^{-1} of C sequestration, a value close to our rate (3.59 Mg C ha^{-1} yr^{-1}).

The high rate of C sequestration observed in our study may be the result of the recent management intervention from continuous grazing to AMP in 2010 (Smith, 2014). However, we are unsure for how long this high rate of C sequestration may continue. Based on the 3.53 Mg C ha^{-1} yr^{-1} of C sequestered over a 9-year period in Wang et al. (2015), we expect that our soils could continue to sequester at this rate for several years. However, because soils that are further from C saturation will accumulate C faster than soils near saturation, and because our estimated soil C sequestration rate is much greater than the 0.41 Mg C ha^{-1} yr^{-1} indicated by Conant et al. (2003), we expect continued sequestration, likely to diminish over time (Minasny et al., 2017; Stewart et al., 2007). Therefore, we caution about extrapolating the reported rates for an extended period. Continued collection of soil C data and monitoring of AMP grazing systems in the Upper Midwest will shed more light on the ultimate C sequestration and storage potential.

Although we assumed in this study that the croplands used for the FL scenario experience soil erosion rather than carbon sequestration, there is considerable opportunity for sustainability improvements of croplands as well, through different agronomic practices such as leaving crop residues, increased use of perennials or cover crops in rotations, conservation tillage and no-till, 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, well-managed grasslands continue to sequester C at greater rates than improved cropland, and transitions from cropland to grassland show some of the greatest rates of C sequestration, ranging from 0.22 to 8.0 Mg C ha^{-1} yr^{-1} (Machmuller et al., 2015; Mathew et al., 2017; Minasny et al., 2017; Powlson et al., 2011).

4.5. Net GHG flux

Although soil erosion contributed negligibly (< 0.5%) to net GHG emissions of FL finishing, soil erosion magnified by intensive cropping systems, such as under conventional feed production in the FL scenario, 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).

In beef LCA studies from the Northern Great Plains and the Upper Midwest in which soil C dynamics were estimated, hypothetical C sequestration rates of 0.41 and 0.12 Mg C ha^{-1} yr^{-1} reduced emissions from continuous 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 5 years of on-farm finishing data and 4 years of comparative soil C data from paddocks under AMP grazing. Therefore, it is possible that long-term AMP grazing finishing in the Upper Midwest could contribute considerably more to climate change mitigation and adaptation than previously thought.

While we did not contrast the more standard continuous grazing to AMP grazing, recent studies indicate that grazing management can be a key driver promoting the relatively high SOC levels that we observed. For instance, Teague et al. (2016) estimated that AMP grazing could induce SOC sequestration rates 10 times greater than the commonly used default value of 0.41 Mg C ha^{-1} yr^{-1} proposed by Conant et al. (2003). This is also demonstrated by Wang et al. (2015) where AMP grazing lands sequestered 3 Mg C ha^{-1} yr^{-1} more than land under continuous grazing. Finally, when converting degraded cropland to AMP grazing, Machmuller et al. (2015) reported mean sequestration values as great as 8.0 ± 0.85 Mg C ha^{-1} yr^{-1} in the top 30 cm of soil for a 7-year duration. Given evidence that effective grazing management can contribute to greater C sequestration rates, higher potential values should also be included in LCAs that consider improved grazing management systems (e.g., AMP grazing). Furthermore, greater rates of C sequestration may be possible for several years. While AMP grazing was adopted at the Lake City AgBioResearch Center just a few years before the current study (in 2012), Machmuller et al. (2015) monitored soils for 7 years during the transition from cropland to pastureland, and Wang et al. (2015) monitored soils for 10 years in ranches where long-term AMP grazing management had been practiced and showed longer periods of greater C sequestration rates. It should be noted that soil C dynamics are directly linked to both land potential and the productivity of both pasture and cattle, and that both stocking rates and productivity influence the net GHG footprints estimated here. Thus, producing the same amount of cattle using land that is less productive would yield lower C sequestration per ha, but also lower stocking rates and reduce beef production (and associated emissions) per ha. Therefore, different regions are likely to demonstrate different outcomes as several variables shift. However, at a minimum, our analyses and results reflect the key importance of geographically localized LCAs to more accurately estimate impacts of grazing on C sequestration rates.

The greater C sequestration and resulting net GHG sink in the AMP grazing system illustrated in this study, compared to continuous grazing systems in other LCAs, calls into question the common assumption that FL intensification reduces the overall GHG footprint through greater productivity. This study shows that when full consideration of land impacts is given to GHG emissions in both FL and AMP grazing-based finishing, environmental benefits may outweigh productivity losses.

Generally, because of their large enteric CH_4 emissions, grazing systems have been pointed to as the greatest area for attention for decreasing the GHG footprint of beef production. However, measurements of SOC have not typically been factored into these outcomes. In fact, if soil C is sequestered through best management practices, our results suggest that enteric CH_4 from the finishing phase can be substantially mitigated. We demonstrated this based on measurements of soil C and cattle productivity at the Lake City AgBioResearch Center from 2012 to 2016, which indicates a sink during the finishing phase of $-6.65\ \mathrm{kg\ CO_2\text{-}e\ kg\ CW^{-1}}$ which is similar to the results of Beauchemin et al. (2011). Yet, our results differ from many of those in the current literature, reflecting the importance of considering emissions and sinks from the entire system in a geographically localized area, including the soil ecosystem, when modeling beef production systems.

5. Conclusions, implications and future projections

Using a standard LCA approach including soil C accounting, this study calculated net GHG emissions of two beef finishing systems in the Upper Midwest, U.S.: FL and AMP grazing. Several important impacts can be derived. Integrating on-farm soil C data within the studied management system contributes significantly to existing LCAs. In doing so, our results show that not only can adoption of improved grazing management facilitate soil C sequestration, but that the finishing phase of the beef production system may serve as an overall GHG sink. While it is unclear how long this effect will be observed, it is reasonable to hypothesize that it would continue, possibly at a reduced rate, for several years into the future.

Studies suggest that intensification of beef production systems would significantly reduce GHG emissions. However, as illustrated by this study in the Upper Midwest, under AMP grazing, more extensive (grass-based) but intensively managed beef finishing can deliver environmental benefits (such as soil C sequestration and other ecosystem services) with less environmental impact per kg CW than intensive FL finishing. While AMP grazing requires twice as much land than FL, if effectively implemented over a large area, total C sequestration in the Upper Midwest could increase substantially. This does imply less overall beef production in the region, albeit with greater environmental benefits from what is produced. Further, before AMP grazing can be realistically implemented across a large landscape, a concerted effort must be implemented to educate livestock producers on its benefits, as a great majority of the United States still employs continuous grazing for grass-fed beef production.

Continued investigation of AMP grazing in large-scale landscape trials across multiple ecoregions differing in climate and plant species mixtures is necessary to have a more nuanced appreciation of the role of ruminants in GHG dynamics coupled with their role in the global food supply.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.agsy.2018.02.003.

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