ADVANCING BEEF SUSTAINABILITY: LONG-TERM IMPACTS OF ADAPTIVE MULTIPADDOCK GRAZING AND SOIL CARBON SEQUESTRATION ON BEEF'S CARBON FOOTPRINT IN THE MIDWESTERN USA

By

Lautaro Garcia

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ABSTRACT

Beef cattle are the largest source of greenhouse gas (**GHG**) emissions in the livestock sector and the broader agri-food system. Adaptive multipaddock (**AMP**) grazing, an intensive form of rotational grazing aimed at improving forage and animal productivity, has been proposed as a rare, low-input, low-cost strategy to potentially offset beef emissions by boosting soil carbon (**C**) sequestration. However, the long-term (>10 years) impact of AMP on soil C sequestration in grazinglands and net GHG emissions in the entire cow-calf to finish beef production system, using longitudinal soil C monitoring, remains highly uncertain. To study these impacts, we conducted a life cycle assessment (**LCA**) using eleven years of on-farm input data and soil C measurements from the Lake City AgBioResearch Center in the Upper Midwest. We compared two beef production systems: AMP-only and AMP+FL (where FL stands for feedlot). Both systems employed AMP grazing during the cow-calf and stocker phases, differing only in the finishing strategy: grass-finishing under AMP for AMP-only and FL-finishing for AMP+FL. Our impact scope included emissions from enteric methane, manure management, feed production, on-farm energy use and transportation, and potential soil C sequestration. Soil C sequestration in AMP-managed pastures averaged 1.70 Mg C ha^{-1} yr^{-1} over eleven years. Including soil C into the GHG balance, AMP-only emissions were reduced from 29.80 to -10.04 kg CO_2-e/kg CW (133%), and AMP+FL emissions from 25.85 to -2.41 kg CO_2-e/kg CW (109%). These data indicate that both systems could become net C sinks over the eleven years. However, the observed drop in soil C sequestration from years 4 to 11 in the top 30 cm suggests that the sequestration rate is slowing, thus impacting offsetting potential. Despite this, AMP grazing may already offer a solution for significant long-term (>10 years) climate benefits while maintaining beef production and farm resilience in current beef grazing systems in the Upper Midwest.

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LIST OF ABBREVIATIONS

AMP	Adaptive multipaddock
AU	Animal unit
BD	Bulk density
BPSs	Beef production systems
C	Carbon
CH_4	Methane
CO_2	Carbon dioxide
CL	Clay loam soil
CO_2-e	CO2-equivalent
CP	Crude protein
CW	Carcass weight
DDGS	Dried distillers grains with solubles
DMI	Dry matter intake
EF	Emission factor
ESM	Equivalent soil mass
GEI	Gross energy intake
GHG	Greenhouse gas
IPCC	Intergovernmental Panel on Climate Change
LCA	Life cycle assessment
Mg	Megagram or ton
MMS	Manure management system
N_2O	Nitrous oxide

Pg	Petagrams
S	Sandy soil
SL	Sandy loam soil
SOC	Soil organic carbon
TDN	Total digestible nutrients
Ym	Enteric methane conversion factor

CHAPTER 1

LITERATURE REVIEW: ADAPTIVE MULTIPADDOCK GRAZING AS A CLIMATE SOLUTION FOR BEEF PRODUCTION - A 5W1H PERSPECTIVE

1.1 Introduction

Grasslands are essential for ruminant production, soil carbon (**C**) storage, and providing many other ecosystem services (Bengtsson et al., 2019). However, chronic mismanagement has degraded around 50% of current grasslands, as assessed by changes in vegetation greenness (Bardgett et al., 2021), and caused significant historical losses of soil C stocks. This degradation presents an opportunity for implementing ecological-guided grazing approaches that can enhance pasture productivity and resilience and potentially restore soil C levels (throughout this thesis, the term “soil C” will refer to soil organic carbon).

Recently, the interest in improving grazing management has mainly been due to its potential to offset ruminant greenhouse gas (**GHG**) emissions, particularly from beef cattle. The beef sector alone contributes 5.9% of global anthropogenic GHG emissions (Gerber et al., 2013). Adaptive multipaddock (**AMP**) grazing, an intensive form of rotational grazing with long recovery periods, has shown promise in enhancing soil C sequestration compared to other grazing methods (Mosier et al., 2021; Wang et al., 2015). Furthermore, it presents a rare opportunity as it addresses the three pillars of sustainability–social, economic, and environmental (Gosnell et al., 2020)–offering low cost, low input, and scalability across millions of hectares grazed worldwide.

Despite its recognized benefits as a natural climate solution (Griscom et al., 2017), the practical implementation of grazing management practices to optimize soil C storage and sequestration remains underutilized. This delay is partly due to a lack of clarity regarding the interconnected dynamics of soil C, grazing practices, and animal GHG emissions.

This review aims to comprehensively analyze AMP grazing as a climate solution for beef production by conducting an integrative assessment of the literature on grasslands soil C, AMP grazing, and beef GHG emissions. The primary objective is to explore key questions through the 5W1H or Kipling framework–Who, What, Why, Where, How, and When–adapted specifically to assess AMP grazing's potential in mitigating the climate impact of beef production (**Table 1.1**).

Table 1.1. 5W1H framework for studying AMP grazing's potential as a climate solution.

5W1H	Adaptation of the framework
WHO	AMP grazing definition
WHAT	What is the potential climate mitigation with AMP grazing in beef production? Case studies
WHY	Beef sector contribution to GHG emissions, and other environmental and socioeconomic reasons to improve grazing management
WHERE	Role of grazing systems in the beef supply chain
HOW	Mechanisms of AMP grazing to reduce beef climate impact
WHEN	Potential readiness of AMP grazing as a climate solution

By addressing these interconnected questions and related challenges, this review seeks to inform the design of sustainable grazing practices, facilitate evidence-based climate action, and identify future research needs to ensure the long-term sustainability of grazing beef systems.

1.2 Who: Origin and definition of AMP grazing

The concept and practice of AMP grazing have gained notable attention among producers, scientists, and the media in recent years (Clifford et al., 2020). Rooted in the pioneering work of influencers such as André Voisin (Voisin, 1959) and John Acocks (Acocks, 1966), AMP encompasses various grazing management principles that have evolved into similar practical methodologies such as Holistic Planned Grazing (Savory & Butterfield, 1999), Management Intensive Grazing (Martz et al., 1999), and Voisin Rational Grazing (Machado, 2010), among others. More recently, Teague et al. (2011) introduced the term "adaptive multipaddock" grazing, which has emerged as the most used term within the scientific community.

AMP is a non-prescriptive grazing approach characterized by short grazing events and extended plant recovery periods post-grazing (Teague et al., 2013). This approach aims to maintain or improve livestock performance by enhancing pasture productivity and ecosystem function. In practice, this is achieved by grazing a large herd of livestock in multiple small, fenced paddocks for hours to days at moderate to high densities. Higher densities promote more uniform forage utilization, even distribution of urine and feces, and trampling of forage, all of which help cycle above-ground nutrients back into the soil (Thompson et al., 2023). Additionally, AMP producers frequently leave enough trampled and/or standing forage residue to support plant recovery and maintain continuous soil coverage, allowing plants and their root systems to stay healthy (Thompson et al., 2023). Although AMP grazing typically requires minimal fossil fuel inputs, it demands significant knowledge and hands-on management (Garnett et al., 2017; Wang et al., 2020). Producers frequently move cattle between paddocks, making informed decisions about grazing duration and intensity through proactive planning, experimentation, and continuous monitoring of plant and animal performance (Stanley et al., 2024).

The main difference between AMP and other grazing methods lies in its ability to prevent overgrazing while harnessing the ecological and economic benefits of grazing, which would be diminished or nonexistent under undergrazing or no-grazing scenarios. Grazing, defined as the act of animals defoliating plant tissue (Stanley et al., 2024), is inherent to all grazing systems. However, since overgrazing occurs when an individual plant is subjected to repeated and severe grazing without sufficient time for physiological recovery (Teague et al., 2013)–particularly due to excessive grazing duration or insufficient recovery time between grazing events–the occurrence of this phenomenon highly depends on the degree of temporal control within the grazing practice. In continuous grazing, animals graze continuously in one pasture without rotation throughout the season. This results in preferential and repeated defoliation of the most palatable plants while less palatable ones may be left untouched, creating a mosaic of overgrazed and undergrazed patches (Stanley et al., 2024); or in extreme cases of animals' nutritional deficiencies, it can lead to overgrazing of the entire pasture. Conventional rotational systems, although they allow some recovery time between grazing events and reduce patchy defoliation, can still lead to overgrazing. If rotations and stocking rates are not well-adapted to plant growth and the farm's carrying capacity, grazing periods may be too long, or recovery periods too short, resulting in the defoliation of plants that have not fully recovered (Stanley et al., 2024). This is why conventional rotational systems tend to be more prescriptive and focused on short-term animal and pasture productivity, sometimes using synthetic inputs. In contrast, AMP systems emphasize adaptability and usually have more paddocks to prevent overgrazing and enhance long-term farm resilience, which often results in lower dependency on synthetic inputs.

AMP farms have documented several economic and resilience benefits, including boosted animal productivity per unit of land and enhanced pasture productivity. Additionally, AMP grazing can improve overall ecosystem function by increasing plant diversity, reducing bare ground and soil erosion, improving water-holding capacity, and promoting the growth of perennial species (Gosnell et al., 2020; Teague et al., 2013; Teague et al., 2011). Moreover, a growing body of literature suggests it can sequester SOC (Machmuller et al., 2015; Mosier et al., 2021; Rowntree et al., 2020; Stanley et al., 2018; Teague et al., 2011).

1.3 Why: Environmental and socioeconomic considerations to improve grazing management in beef operations

Given that beef cattle production predominantly relies on grazing systems globally, managing grazinglands judiciously is critical for the sustainability of the beef sector. A potential working definition of sustainable beef production emerging from scientists and supply chain stakeholders includes beef production systems (**BPSs**) that are environmentally, socially, and economically sound (GRSB, 2018). These systems feature cattle that are predominantly grass-based and grazed on pastures to minimize feed-food competition and maintain high animal welfare standards (EIP-AGRI, 2021; Eshel et al., 2018; NRC, 2010). They are producer-centered, low-input, durable, capable of balancing trade-offs, and incorporate agroecological principles (Smith et al., 2024; Thompson et al., 2023). Moreover, they provide safe, nutritious, and affordable beef and byproducts while maintaining or improving the natural resource base (NRC, 2010). This definition, adaptable to local conditions, could serve as a benchmark for holistically evaluating BPSs' sustainability and the suitability of GHG mitigation strategies, preventing unintended consequences from a carbon-myopic approach (Harrison et al., 2021).

Beef plays a major role globally in the economy and society, providing employment across the supply chain and contributing to food security. As the third most consumed meat worldwide (Greenwood, 2021), it is a highly desired source of animal protein. However, the environmental impacts of beef production, particularly its contribution to climate change, are highly debated and challenge the sector's sustainability, warranting further exploration as detailed in the subsequent sections.

1.3.1 Current state of grassland soil C, grazing, and beef GHG emissions

The research history of soil C, improved grazing management, and beef GHG emissions spans several decades. Extensive research on soil C and climate change mitigation began in the 1990s (Barnwell et al., 1992; Paustian et al., 1997), leading to the first syntheses summarizing soil C sequestration rates in grazinglands with improved grazing practices in the early 2000s (Conant et al., 2001; Follett et al., 2001). In 2006, FAO released the report "Livestock's Long Shadow" on the GHG impacts of animal agriculture, which significantly promoted research on GHG emissions in beef production (Steinfeld et al., 2006). Later, in the 2010s, AMP grazing gained modest research interest for its potential environmental benefits, including soil C sequestration, compared to other forms of grazing (Teague et al., 2011).

Research on soil C, grazing, and GHG emissions has substantially increased in recent years, but typically in isolation. Moreover, among experimental grazing studies, AMP grazing is rarely applied due to its non-prescriptive and adaptive nature, which makes it unsuitable for traditional experimental designs (Teague et al., 2013). Not surprisingly, even fewer studies have integrated AMP with soil C change monitoring into a comprehensive farm-to-gate GHG balance assessment (Rowntree et al., 2020; Stanley et al., 2018). This limited integration has hindered a full understanding of AMP grazing's effectiveness in improving soil C and the net GHG balance of beef production (**Figure 1.1**).

Nevertheless, previous work has established some key consensus related to beef grazing cattle: historical mismanagement of grazinglands has led to significant soil C losses (Sanderman et al., 2017); judicious grazing management can improve soil C stocks (Smith, 2014); and beef cattle significantly contribute to global GHG emissions, primarily through methane (**CH_4**) and nitrous oxide (**N_2O**) emissions (Gerber et al., 2013).

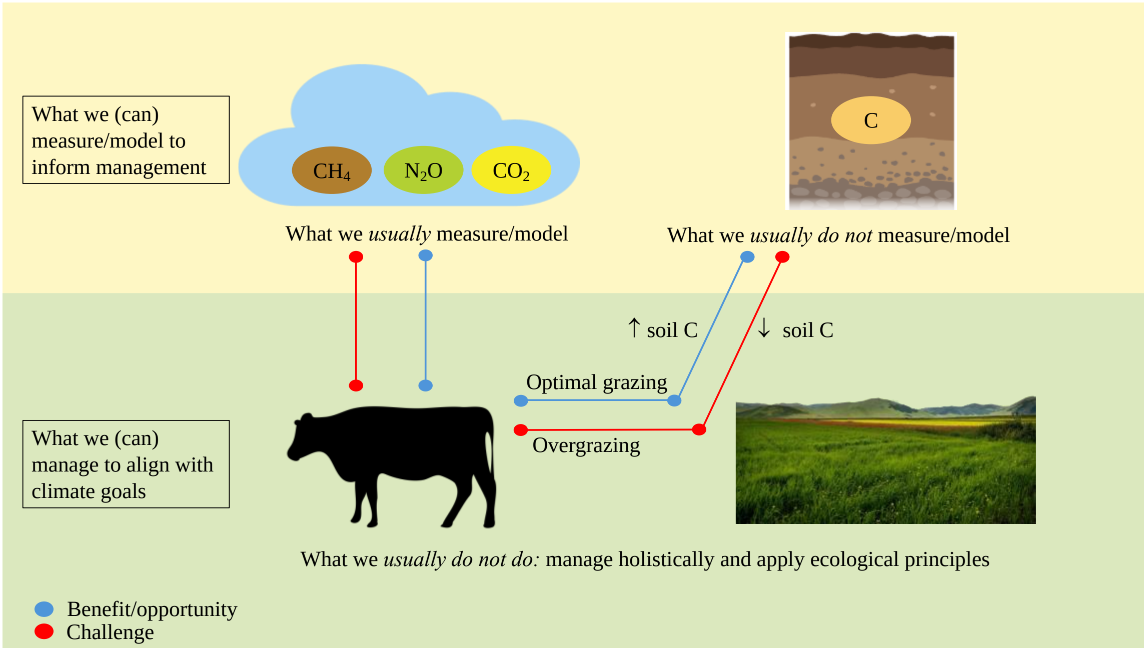

Figure 1.1. Simplified illustration of the nexus between beef cattle and grazinglands in the context of GHG emissions and soil C change.

1.3.1.1 Current state and management of grasslands for grazing beef systems

Grasslands, defined as ecosystems primarily dominated by grass or grass-like vegetation (Dondini et al., 2023), play a crucial role in the global and ruminant C cycle, covering ~40.5% of Earth's land surface (Bai & Cotrufo, 2022). These grasslands, including both natural grasslands and pastures, account for nearly 2 billion hectares used for livestock grazing (Dondini et al., 2023), predominantly by beef cattle, and store ~20% of the world's soil C stocks (Dondini et al., 2023).

However, grasslands globally are threatened by human management, with about half of them having experienced some degree of degradation (Bardgett et al., 2021), largely due to both undergrazing and overgrazing (Bai & Cotrufo, 2022; FAO, 2016b). This degradation impacts both dry and wet climates and leads to substantial reductions in soil C stocks (Dlamini et al., 2016). However, the historical loss of grasslands' soil C can be thought of as the technical potential for soil C sequestration (Sanderman et al., 2017). A recent global meta-analysis found that over the past six decades, livestock grazing has reduced soil C stocks by 17 ± 4% at a 1-meter depth, resulting in an absolute loss of 46 ± 13 Pg C across ~24 million km^2 of grazinglands (Ren et al., 2024).

There is broad agreement on several key points regarding the management of grasslands soil C stocks: 1) preventing grassland conversion to cropland is key to avoid potential losses of 30%–50% of topsoil C stocks (Paustian et al., 2016; Smith, 2014); 2) continuous grazing tends to reduce soil C stocks (Bai & Cotrufo, 2022); and 3) improved grassland and grazing management practices can increase soil C stocks (Conant et al., 2017; Griscom et al., 2017). Thus, managing grasslands to prevent soil C loss and ideally transform them into C sinks is essential.

Grazing practices vary widely in input use. In terms of management intensity, they can broadly be classified into continuous and rotational grazing. Continuous grazing lacks rotations and is generally indicated as the most common practice globally (McDonald et al., 2019; Wang et al., 2015). This is supported by a U.S. Department of Agriculture (USDA) survey that found 60% of U.S. cow-calf operations use continuous grazing, while the remaining 40% use variations of rotational systems (Whitt & Wallander, 2022).

1.3.1.2 Overview of beef GHG emissions globally

The beef industry is the largest source of GHG emissions within the agricultural sector globally (Gerber et al., 2013). Most beef emissions occur during the production phase (Poore & Nemecek, 2018), including both indirect emissions from the manufacturing and distribution of inputs (e.g., seed, animal feed, fertilizers) and direct emissions from beef cattle. The primary GHGs associated with these processes are CH_4, N_2O, and carbon dioxide (**CO_2**).

While beef usually has the highest C intensity compared to other animal and plant proteins (Semba et al., 2021), it also shows the greatest variability (Gerber et al., 2013; Poore &

Nemecek, 2018). The global average C intensity of beef, including both pre- and post-farm activities, is estimated at 46.2 CO_2-equivalent (**CO_2-e**)/kg carcass weight (**CW**). However, values range from 14 kg to 76 kg CO_2-e/kg CW (Gerber et al., 2013).

Several factors can account for the differences in C intensity, including beef origin, feed digestibility and production efficiency, and land-use change. First, beef origin plays a major role. The lowest regional values are in Western and Eastern Europe and the Russian Federation because most beef is “dairy beef” (Berry, 2021), where beef is a co-product of milk production, resulting in significant emissions allocated to the dairy industry (Gerber et al., 2013). Second, specialized BPSs in industrialized countries such as those in Western Europe, North America, and Oceania have lower C intensities due to high feed digestibility and production efficiency. In contrast, developing regions like South Asia, sub-Saharan Africa, Latin America, and East and Southeast Asia have lower feed digestibility, resulting in lower slaughter weights, higher age at slaughter, and higher C intensities (Gerber et al., 2013). In Latin America, land-use change emissions from converting forests to pastures also add significantly to the C footprint. Third, GHG profiles differ between industrialized and developing regions. Developing areas, which rely heavily on grazing, mainly emit GHGs from enteric CH_4, whereas industrialized regions, reliant on feed imports, fertilizers, and mechanization, show higher emissions of CO_2 and N_2O, with less dominance of enteric CH_4 (Gerber et al., 2013). Finally, C intensities also differ among countries and even within ecoregions. For example, the means of U.S. cradle-to-gate emissions for specialized BPSs across ecoregions range from 20.2 to 28.9 kg CO_2-e/kg CW. In the Southern Plains, the primary beef-producing ecoregion in the U.S., individual production systems range from 16.6 to 30.1 kg CO_2-e/kg CW (Rotz et al., 2019).

1.3.1.3 Current suggested mitigation strategies in beef life cycle assessments

The wide range of emissions within the beef supply chain shows that there is considerable room for improvement. Gerber et al. (2013) estimated that emissions from the livestock sector could be cut by 30% if producers followed the practices of the 10% of producers with the lowest emission rates in the same regions and production systems. Recently, Cusack et al. (2021) conducted a global meta-analysis evaluating strategies to reduce and offset emissions in beef production. They reviewed 57 comparative beef management life cycle assessments (**LCAs**) published between 2006 and 2018, covering 292 comparisons of improved versus conventional management. Beef mitigation strategies generally fall into two categories: (1) improving efficiency to reduce GHG emissions per unit of beef and (2) enhancing land-based C capture to offset emissions. The study found that land-based C sequestration (e.g., silvo-agroforestry, organic compost, intensive rotational grazing) reduced net emissions by 46% per unit of beef, while efficiency improvements (e.g., better feed quality, supplements, breed selection) reduced net emissions by 8% (Cusack et al., 2021). However, the effectiveness varied by region. Latin America showed the greatest potential for emission reductions through both mitigation categories. In Australia, efficiency improvements were effective, but there was a lack of land-based C sequestration studies. In the U.S., C sequestration showed potential, whereas efficiency strategies had no significant effect. Western Europe, Canada, and Asia did not exhibit clear effective mitigation strategies (Cusack et al., 2021).

In summary, efficiency improvements alone may not suffice to reduce aggregated emissions due to the expected growth of the industry. Thus, exploring land-based C sequestration strategies, which generally show significant potential, is crucial for meaningful mitigation.

1.4 Where: Distribution and characteristics of current beef production systems

Understanding where and how beef is produced globally is important to assess the scalability of mitigation practices. Between 2020 and 2022, 71 million metric tonnes (Mt) of beef were produced annually, with ~44% from developed and ~56% from developing countries (OECD/FAO, 2023). Two-thirds of the production comes from only seven countries: the U.S., Europe, and Australia among developed countries, and Brazil, China, Argentina, and India among developing countries (Greenwood, 2021). Global beef production is projected to rise by 9% by 2032, mostly driven by middle-income countries (OECD/FAO, 2023). These projections are important because the net mitigation effect of beef's climate impact depends on both emission intensity reductions and total beef output (Gerber et al., 2013).

Specialized BPSs contribute 57% of the total beef output, mainly in North America, Latin America, and Australia. Dairy systems, prevalent in Europe, New Zealand, and India, make up the remaining 43% (Gerber et al., 2013; Greenwood, 2021). Specialized BPSs are typically divided into cow-calf, stocker, and finishing phases, with the first two primarily grassfed-based (Endres & Schwartzkopf-Genswein, 2018). Production stock (i.e., non-breeding stock) is either grass-finished, particularly in Latin America and Australia (ABIEC, 2023), or confined in feedlots, predominantly in North America (Greenwood, 2021).

In conclusion, BPSs worldwide are diverse, with a significant percentage of the beef output from only a few countries, both developed and developing. Grazing systems are a component most BPSs share (although absent in feedlots and some dairy systems). Therefore, there is both an opportunity and a need for beef GHG mitigation in grazing systems globally.

1.5 How: Mechanisms of AMP grazing to reduce beef net GHG emissions

1.5.1 AMP grazing for soil C sequestration: evaluating impacts on pathways of soil C change under a conceptual framework

The literature evaluating the effects of grazing on soil C shows inconsistent results, ranging from significant gains to substantial losses (FAO, 2019). This inconsistency partly stems from the complexity and variability of grazing practices, often oversimplified in research (Teague et al., 2013). Studies have typically relied on dichotomous comparisons such as "presence or absence" of grazing or "light versus heavy" grazing (Abdalla et al., 2018), leading to interpretations such as higher stocking rates (animal units/total grazed area) equating to greater defoliation, and potentially less soil C. Therefore, some studies recommend reducing grazing intensity by reducing farm animal units (**AUs**) to increase soil C (McSherry & Ritchie, 2013). However, while this approach may be appropriate in certain contexts, it overlooks the diverse nuances of grazing patterns. This lack of comprehensive metrics has led to mixed outcomes across the literature, impeding a cohesive understanding of the impacts of grazing on soil C changes (Stanley et al., 2024).

Recognizing these limitations, Stanley et al. (2024) introduced a holistic framework to understand the importance of adaptively managing specific grazing levers or metrics for optimal soil C sequestration. This framework defines three grazing patterns–undergrazing, “optimal” grazing, and overgrazing (**Figure 1.2**)–based on four levers: intensity, duration, frequency, and timing (**Figure 1.3**). These patterns influence ecosystem function over time. They defined undergrazing as the ecological and soil C scenarios where the benefits of grazing are not fully utilized due to insufficient animal numbers (poor utilization) and/or infrequent grazing (over-rest). Overgrazing, on the other hand, is defined as grazing in excess (i.e., excessive number of animals and/or grazing too frequently), which can lead to detrimental ecosystem function and soil C loss. They argue that the potentially greater success of AMP grazing in soil C sequestration over other grazing practices might be attributed to AMP producers' ability to achieve more consistently “optimal” grazing patterns in the long term, limiting overgrazing or undergrazing scenarios.

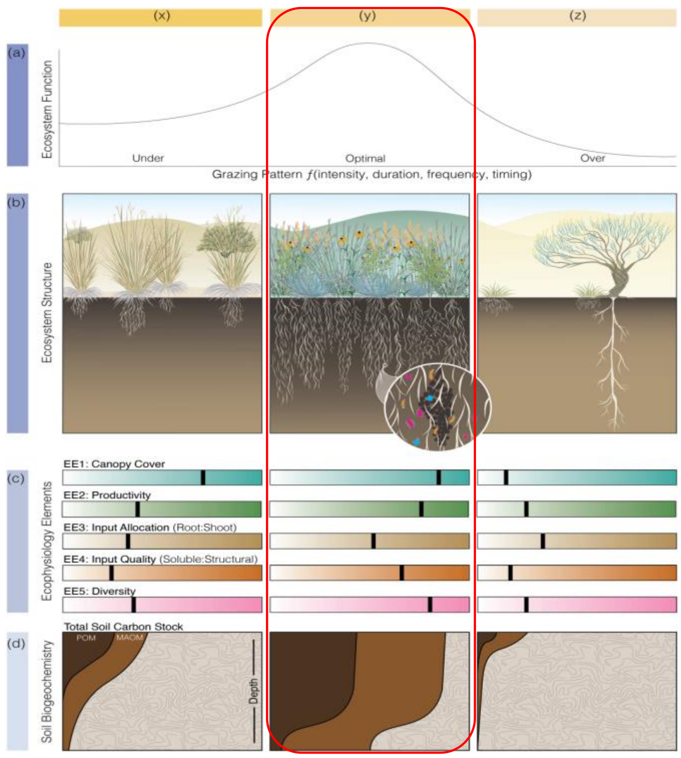

Figure 1.2. Illustration of grazing effects on the ecosystem and soil C. Reproduced with modification from Stanley et al. (2024) with permission from the authors. Conceptual illustration outlining how grazing patterns—undergrazing (x), optimal grazing (y), and overgrazing (z)—affect ecosystem function (a) and structure (b) by regulating plant ecophysiological elements (EEs; c), which subsequently impact soil biogeochemical outcomes (d), particularly the total soil organic carbon stock (represented by particulate (POM) and mineral associated organic matter (MAOM)). EEs (c) are shown either as continuous gradients or as ratios of two components. The color gradient from left to right indicates low-high or left:right, respectively, and the black markers denote the projected outcomes for each grazing pattern. The red square represents where AMP grazing would potentially fit among the grazing patterns.

In practice, AMP producers apply the four management levers (see **Table 1.2** for definitions) in their day-to-day practices. Smaller paddock sizes in AMP allow and require producers to observe the land's conditions more frequently and in greater detail. These paddocks are grazed only sporadically throughout the year (frequency lever). AMP producers make informed decisions about when to graze a pasture (the timing lever), typically guided by the key species' growth stage (i.e., a preferred species susceptible to overgrazing; Xu et al., 2019) and/or a grazing plan (Rowntree et al., 2020), while also adapting to other on-the-ground observations and external factors. Using smaller paddocks naturally increases stocking density (i.e., AUs/hectare), requiring quicker cattle rotations to fresh paddocks (duration lever) and preventing overgrazing of regrowing plants. The manipulation of frequency, timing, and duration culminates in a specific amount of biomass utilized during a grazing window (intensity lever) (Stanley et al., 2024). It is common for AMP producers to leave a substantial amount of plant residue in grazed pastures.

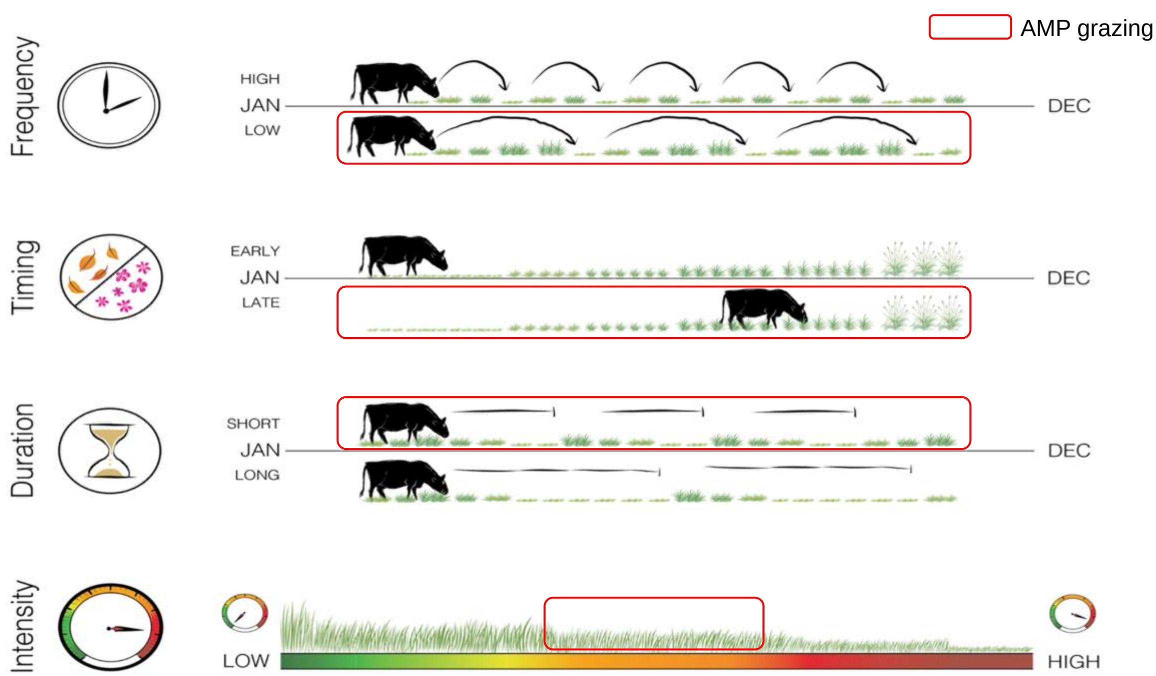

Figure 1.3. Illustration summarizing critical elements of a grazing pattern. Reproduced with modification from Stanley et al. (2024) with permission from the authors. Frequency, timing, and duration are displayed on an annual timeline, offering contrasting examples of how they vary in "high" and "low" scenarios. Intensity is depicted by the biomass consumed during a specific period, which is influenced by the preceding three elements. Red squares indicate how AMP grazers tend to apply these four elements.

Stanley et al. (2024) propose that the long-term impacts of grazing on soil C depend on how grazing influences five ecophysiological elements (EEs): ground and canopy cover (EE1), plant productivity (EE2), input allocation (EE3), aboveground input quality (EE4), and plant diversity (EE5). If these EEs are positively stimulated by the management levers, they could account for soil C increases via three pathways or proximal controls on the soil C balance: adding C inputs, reducing C losses, and catalyzing below-ground transformations (Stanley et al., 2024). Although detailed theories on how AMP and the five EEs impact the three pathways are beyond the scope of this review, we summarized in Table 1.2 the potential effects of AMP on EEs (grazing → EEs) and the subsequent effect of each EE on the pathways that could ultimately explain increases in soil C (EEs → Ps). The authors argue that by having more spatial and temporal control to adapt grazing timing, duration, frequency, and intensity in response to these EEs, AMP producers may be able to achieve more "optimal" grazing patterns over the long term.

1.5.2 AMP grazing for enhancing productivity and reducing fossil fuel emissions

The mitigation potential of AMP grazing extends beyond soil C sequestration. Improved grazing management can enhance forage quality and quantity with low or null use of synthetic inputs, boosting cattle performance (in both meat and breeding stock) and reducing fossil fuel emissions (Teague & Barnes, 2017). However, the effectiveness of these strategies depends partly on the efficiency baseline of the beef grazing system. Many BPSs in regions like Sub-Saharan Africa and Latin America operate with low efficiency, largely due to cattle's nutritional constraints. Temporal forage shortages and limited access to feed resources hinder the ability to meet cattle's nutritional needs consistently. This contributes to the fact that ruminants from developing countries account for 75% of global ruminant GHG emissions (Herrero et al., 2013). Grazing systems in developing countries often suffer from low productivity, poor feed quality (Herrero et al., 2013), and infrequent supplementation (Gerber et al., 2013), leading to suboptimal reproductive and growth performance and higher GHG emissions per animal. For instance, South American specialized BPSs produce 31% of global beef but contribute 54% of emissions from the specialized beef sector (Gerber et al., 2013). This is partly due to low weaned calf percentages–56-60% in Brazil and Argentina compared to 82-84% in the U.S. and Canada (Basarab et al., 2024; Lobato et al., 2014; Vázquez et al., 2023). Additionally, Sub-Saharan Africa, with 20% of global pastureland and 17% of the world's beef herd, accounts for only 7% of total beef output (OECD/FAO, 2023).

In industrialized countries with highly efficient BPSs that often rely on fossil fuel inputs like synthetic nitrogen (**N**) fertilizer (e.g., in the U.S. and Europe) (Gerber et al., 2013), AMP grazing can reduce dependency on these inputs. Reducing fertilizer use can significantly decrease the overall emissions of beef operations (Stanley et al., 2018). AMP grazing could potentially also decrease enteric CH_4 emissions via improvements in forage digestibility, however, research findings vary regarding short- and long-term impacts of different grazing practices in animal CH_4 production (Thompson & Rowntree, 2020).

Table 1.2. Conceptual framework applied on AMP grazing to potentially explain AMP's common higher soil C outcomes.*

	Item	Definition	Main use & theoretical effect of AMP grazing
Grazing pattern[1]	Frequency	Number of days of rest a pasture receive before being regrazed [1]	Long, not too unfrequently/frequently [1]
	Timing	Phenological growth stages and season inform plant vulnerability to defoliation [2]	Planned and adjusted to promote desirable species and functional groups
	Duration	Days animals spend grazing per pasture per year [1]	Short [1] (combined with high stock density to promote low selectivity and more evenly distribute grazing utilization) [3]
	Intensity	Amount or % of aboveground biomass consumed by grazers over a given time compared to residual levels [1]	Influenced by frequency, timing and duration* Often 40-60% of biomass utilization
Ecophysiological elements (EE) to observe and adapt management[1]	**EE1**: Ground and canopy cover	% cover [1]	↑ % cover, ↓ bare soil [4]
	EE2: Plant productivity	Megagram (Mg)/ha of net primary production (NPP) [1]	↑ NPP [4]
	EE3: Input allocation	Root:shoot ratio [1]	↑ root:shoot allocation [5], ↑ root exudation [5, 6], ↑ total root biomass [7], ↑ plant species with greater root:shoot ratios [1]
	EE4: Aboveground input quality	C:N, soluble:structural component ratios [1]	↑ plant litter of newer leaves & manure
	EE5: Plant diversity	Plant species richness or functional group [1]	↑ plant diversity [4] (↑ productivity, N cycling, root allocation, and quality plant litter)
Pathways to ↑ soil C via grazing[1]	**P1**: "Increasing overall C fixation in plant biomass and soil C inputs" [1]	grazing → **EE1**→ **EE2**→ **P1** (increased canopy cover and NPP via defoliation) grazing → **EE2**→ **P1** (compensatory re-growth) grazing → **EE1**→ **EE4** → **P1** (removal of senescent tissue) grazing → **EE1**→ **EE5**→ **P1** (removal of senescent tissue)	
	P2: "Reducing SOC losses, via slower decomposition or erosion prevention" [1]	grazing → **EE1**→ **P2** (reduction of bare soil) grazing → **EE1**→ **P2** (removal of senescent tissue) grazing → **EE3**→ **EE4** → **P2** + **P3** (diverse functional types, low frequency and variable severity of defoliation) grazing → **EE5**→ **P2** ('insurance value' of plant diversity)	
	P3: "Increasing the efficiency of below ground transformations" [1]	grazing → **EE2**→ **P3** (recycle of ↑ quality inputs) grazing → **EE3** → **EE4** → **P1** + **P3** (↑ root exudation → ↑ microbial transformations of organic material) grazing → **EE4** → **P3** (recycle of ↑ quality inputs) grazing → **EE4** → **P1** + **P3** (defer plant senescence keep ↑ quality inputs longer) grazing → **EE5**→ **P3** (diverse litter inputs → diverse microbial community) grazing → **EE5**→ **P1** + **P2** + **P3** (maintain diversity and avoid invasive species encroachment)	

Table 1.2. (cont'd)

Source: [1] (P. L. Stanley et al., 2024); [2] (Browning et al., 2019); [3] (Barnes et al., 2008); [4] (F. Wang et al., 2021); [5] (Wilson et al., 2018); [6] (Hamilton et al., 2008); [7] (Pucheta et al., 2004)
* Read as grazing pattern → ecophysiological elements → soil C outcome.

1.6 What: AMP grazing mitigation via soil C change in practice (case studies)

While intensive rotational grazing is often applied non-adaptively in research settings (Teague & Barnes, 2017), a few studies have examined the impact of AMP grazing on soil C at commercial farm scales. Here, we review four such studies (**Table 1.3**), spanning diverse ecoregions in the U.S., including the Upper Midwest (Stanley et al., 2018), Southeastern (Machmuller et al., 2015; Rowntree et al., 2020), and Southern Great Plains (Wang et al., 2015). Reported soil C sequestration rates ranged from 2.29 to 8.0 Mg C ha^{-1} yr^{-1} across these studies.

To put these numbers into perspective, a synthesis by Conant et al. (2017) on improved grazing management studies globally (n=126 data points, predominantly from the U.S.) found an average soil C sequestration rate of 0.28 Mg C ha^{-1} yr^{-1} (Table 1.3). This disparity with the AMP farms may stem from various biotic (e.g., forage production, animal impact) and abiotic (e.g., climate and soil conditions) factors, though methodological differences and management practices are likely significant contributors. Conant et al. (2017) reported an average soil sampling depth of 19.5 cm, and as noted by the authors, only a few studies using rotational grazing were included in the synthesis, potentially diluting the benefits observed in studies with conventional rotational or more advanced AMP grazing practices. Interestingly, these studies had an average publication year of 1995 and an average study duration of 39 years, highlighting that even modest soil C sequestration rates can yield significant long-term benefits, and also the need for updated research to reflect current practices and methodological advancements.

In contrast, AMP grazing studies sampled deeper soil layers, down to 30 cm (Machmuller et al., 2015; Stanley et al., 2018), 60 cm (Wang et al., 2015), and 100 cm (Rowntree et al., 2020). Additionally, these case studies generally applied the four levers of a grazing pattern (Tables 1.2 and 1.3) in a way that may align more closely with "optimal" grazing events, potentially stimulating more conducive pathways for soil C sequestration (Stanley et al., 2024). Specifically, they reported grazing periods of 0.5 to 3 days, high stock densities, and ensured adequate forage recovery. Notably, the study durations were shorter than those in Conant et al. (2017), ranging from 4 to 20 years. As of 2024, there are no follow-ups, except for Stanley et al. (2018), which is discussed in Chapter 2 of this thesis, to evaluate potential saturation. However, Machmuller et al. (2015) observed an apparent plateau in their study in year 6 out of 7. While the reason for the relatively rapid apparent saturation of soil C after conversion is uncertain, the limited sampling depth to the top 30 cm could partially explain the observed saturation. Additionally, it is important to note that this study was the only one conducted on dairy grazing systems instead of beef and involved intense rotational grazing with high inputs (fertilizer, irrigation) and recovery periods sometimes as short as 15 days, rather than a typical AMP system in beef with low inputs and longer recovery periods.

Importantly, the methodologies of these case studies (Table 1.3) not only highlight the high uncertainties but also emphasize the complexity of evaluating soil C over time at the farm level. Only Stanley et al. (2018) used on-farm data with initial and repeated soil C measurements, whereas the other three studies established baselines based on nearby pastures under conventional grazing management. Furthermore, the sampling design across all four studies, despite generally using stratification, often relied on combining soil samples for analysis. In the absence of reported power analyses, these studies seem to have used few samples, considering the typically high heterogeneity of grazinglands. These combined factors add uncertainty to the reported soil C sequestration rates (Stanley et al., 2023) and underscore the need for improved soil sampling protocols in grazinglands, especially if one of the goals is to generate offsets for C markets or demonstrate reductions in beef production GHG emissions.

1.7 When: Potential readiness of AMP grazing for beef supply chain integration

There is a significant opportunity to deploy AMP grazing rapidly without altering production systems or expanding grazinglands. This is because beef cattle spend most of their lives grazing, with the cow-calf sector being the main land user. Additionally, current adoption rates of AMP grazing are far from saturation. While we were unable to find global surveys on diverse grazing types, as mentioned before, continuous grazing is recognized as the most common grazing type worldwide (McDonald et al., 2019; Wang et al., 2015). In the U.S., a survey revealed that 18% of producers (n=418) practiced AMP grazing, and only 5% were unaware of it (Clifford et al., 2020). However, it is unclear how representative these statistics are of the ~750,000 cow-calf and 15,000-20,000 stocker operations in the U.S. (MacDonald & McBride, 2009; Phillips et al., 2011). Overall, this highlights the ample availability of both land and non-AMP producers, underscoring the significant technical potential for wide-scale implementation of AMP grazing.

Increasing adoption rates could benefit from collaboration among stakeholders across the supply chain. A survey of non-adopters (n=353) of rotational grazing in the U.S. Great Plains found that perceived high initial investment costs were a significant barrier (Wang et al., 2020). However, the current push for climate action in the beef industry may help address these challenges. Many multinational corporations in the meat industry (e.g., JBS, Cargill, Tyson, and McDonald's) have set time-bound net-zero emission targets, including emissions from on-farm practices (commonly referred to as “Scope 3” emissions; Jia et al., 2023). These commitments can be achieved by reducing emissions and increasing removals (UNFCCC, 2015). As a result, these pledges could incentivize producers to adopt potentially low-GHG strategies like AMP grazing through corporate financial incentives or pressure.

Table 1.3. Case studies on AMP grazing and soil C sequestration in the U.S.

Item		Rowntree et al. (2020)	Stanley et al. (2018)	Machmuller et al. (2015)***	Wang et al. (2015) (based on Teague et al., 2011)	Conant et al. (2017)
Climate region in the US		Southeast (subtropical)	Upper Midwest (temperate)	Southeast (subtropical)	Southern Great Plains (subtropical)	Most data points from the US
Land use history		Degraded cropland	Overgrazed introduced pasture	Degraded cropland (cotton, peanuts, tillage for ~50 yrs.)	Native grassland never plowed	-
Stocking rate		0.81 cow/ha (assumed)	2.7 steers/ha	-	0.27 AU/ha, adjusted for drought	-
Stocking density in individual paddocks		(25-50 Mg ha^{-1}, 24h)	-	75-150 AU/ha (12h)	Higher than the continuous treatments in the study	-
Grazing levers	Frequency (i.e., time to recover)	-	-	15-45d	30-50d (fast growing season), 60-90d (slow growing season)	-
	Timing	Ensuring forage recovery	Ensuring forage recovery	15-45d , based on time of year, growth rate, forage quality, and dairy herd needs	Ensuring forage recovery	-
	Duration	1d	1d	0.5d	1-3d	-
	Intensity*	-	70%	-	Light-to-moderate defoliation	-
Soil type (s)		Fine sandy loam	Sandy loam, sandy, clay loam	Fine or fine-loamy	Clay loam	-
Soil depth (cm)		0-100	0-30	0-30 (no detected changes deeper)	0-60	0-19.5
No. of samples (T1 and T2**)		1 (T1) & 4 (T2)	3 (T1) & 9 (T2) (10 subsamples per sample)	18	10	-
Years since conversion		20 (chronosequence)	4	7 (chronosequence)	≥ 9 (chronosequence)	39
Initial SOC stock (Mg C ha-^{1})		~10	33.97	-	Based on nearby farms under continuous grazing	-

Table 1.3. (cont'd)

Final SOC stock (Mg C ha^{-1})	~50	48.33	38	129. 2	-
C seq rates (Mg C ha^{-1} y^{-1})	2.29	3.59	8.0	3.5	0.28

* forage utilization; ** T1 = initial time point or baseline, T2: final time point; ***

1.8 Challenges and future research directions to advance AMP knowledge

1.8.1 Distinguishing AMP from "improved" grazing when evaluating soil C sequestration potential

"Improved" grazing management has shown potential in increasing soil C stocks, but current estimates may be conservative. This is partly due to the underrepresentation of AMP-like grazing practices in meta-analyses and global assessments (often used in climate mitigation reports) and the limitations of soil sampling and modeling methods. One of the most commonly cited figures in the grazing and soil C literature is Conant et al. (2017), which, as previously mentioned, found an average soil C sequestration rate of 0.28 Mg C ha^{-1} $year^{-1}$ from synthesized improved grazing management interventions (n=126) adopted globally, but with little representation of rotational grazing studies, and with an average soil depth of only about 20 cm. Griscom et al. (2017) estimated that the global soil C sequestration potential through optimizing grazing intensity in grazinglands ranges from 148 to 699 Mt CO_2-e $year^{-1}$ by 2030, considering safeguards. This estimation primarily relies on Henderson et al. (2015), who used a process-based model (Century) focused on forage utilization to project soil C changes through grazing management. Thus, this study likely did not capture the effects of AMP grazing-like practices. In fact, Griscom et al. (2017) acknowledged their estimate is intentionally conservative due to limited global data on AMP practices.

In light of this, it is reasonable to hypothesize that greater soil C sequestration globally could be achieved by adopting potentially more suitable grazing strategies such as AMP grazing. However, to support these hypotheses scientifically and on a large scale, there is an urgent need to improve soil sampling methods (e.g., deeper sampling, larger sample sizes, sampling AMP farms) and refine process-based models to better estimate soil C.

1.8.2 Limited soil C inclusion and rare AMP grazing use in LCAs

LCA is a widely used method to assess the environmental impacts of beef (FAO, 2016). The core idea is to track a product throughout its supply chain within a defined boundary, recording inputs (e.g., resources) and outputs (e.g., GHG emissions) (Cederberg et al., 2013). LCAs can be full “cradle-to-grave”, covering all stages of a product's life, or partial “cradle-to-farm gate”, focusing on farm-related activities. To quantify beef production’s impact on global warming, LCAs compile all inputs and outputs within the boundary, usually for one calendar year. Emissions of GHGs like CH_4, N_2O, and CO_2 are then converted into CO_2-e using global warming potentials that reflect the warming effect of non-CO_2 gases relative to CO_2 over a fixed time period (Reisinger & Clark, 2018). The most common period used is 100 years (GWP100), where CH_4, N_2O, and CO_2 are multiplied by factors of 27, 265, and 1, respectively (IPCC, 2021). To ensure that emission reduction practices do not compromise productivity, the annual C footprint of beef is reported in kg CO_2-e per kg of product (Beauchemin et al., 2010), with CW being a common unit.

It is increasingly recognized that beef LCAs should account for both emissions and soil C changes to provide a more accurate picture of the net GHG footprint (Dondini et al., 2023). Traditional LCAs have typically focused on emissions, assuming soil C stocks are in continuous equilibrium due to a lack of on-farm data and reliable models (Gerber et al., 2013). This approach is somewhat reasonable for global or country-level LCAs that need to set conservative baselines. However, farm-level LCAs, which are more effective for developing supply-side mitigation strategies (Greenhut et al., 2013), have often also omitted the soil component.

Additionally, the grazing models used are often continuous or low rotational systems, not representing AMP.

LCA is arguably the best tool currently available to set baselines and identify mitigation strategies in beef production. However, more comprehensive LCAs are needed for mitigation practices focused on soil C sequestration. Future research on AMP grazing impact on climate should prioritize cradle-to-gate LCAs. Additionally, extensive on-farm data collection is key to more accurately capturing soil C and GHG dynamics under different grazing management practices. This effort will require collaboration with producers, who possess the most detailed knowledge of their management practices (Paustian, 2013).

1.8.3 Challenges in grassland soil C sequestration: context-dependence, saturation, reversibility

Despite being a cost-effective way to help mitigate climate change with numerous co-benefits (Minasny et al., 2023), capitalizing soil C sequestration outcomes at scale and over time can be challenging. The directions, magnitude, and timescale of grazing impacts on soil C sequestration are context-dependent, influenced by both biotic factors (e.g., vegetation properties, grazing pattern) and abiotic factors (e.g., soil texture, rainfall, time from conversion to new management) (Bai & Cotrufo, 2022; Byrnes et al., 2018). Furthermore, soil C sequestration potential diminishes over time, expected mainly due to a physical barrier since a significant fraction of soil C is associated with minerals and potential soil N limitations, both limiting further soil C sequestration (Georgiou et al., 2022; Smith, 2014). However, how long soil C can be sequestered in grasslands is more uncertain. It is often believed that soil C sequestration cannot continue indefinitely, though some argue that saturation does not occur (Smith et al., 2014). Additionally, soil C sequestered in grasslands (as several types of C sinks) carries an inherent risk of future non-permanence or reversibility (Smith et al., 2014), due to changes in management, climate change (particularly in arid and semi-arid regions; Bai & Cotrufo, 2022), or fires (Godde et al., 2020).

The interaction among these variables introduces uncertainties regarding the scalability and effectiveness of AMP grazing as a means for soil C sequestration and climate mitigation. To collectively address many of these uncertainties, future worldwide research efforts on AMP should conduct long-term and region-specific studies on AMP and soil C, employing robust sampling methods.

1.9 Conclusions

AMP grazing has emerged as a practical, cost-effective natural climate solution, offering potential to reduce the GHG contribution of current beef grazing systems. Applying the 5WH1 method reveals critical insights. AMP grazing, distinct from conventional grazing systems, necessitates clear differentiation in management to provide practical outcomes for farmers and researchers (who). Grasslands, vital for beef production and diverse ecosystem services, have often suffered from degradation and soil C loss due to mismanagement. This, coupled with beef production's significant GHG emissions, limited mitigation strategies, and the global prevalence of grazing across its supply chain (where), makes optimizing grazing management crucial (why). AMP grazing potentially offers two mechanisms that provide climate benefits: enhancing animal performance, which might be particularly impactful in developing countries, and promoting soil C sequestration (how). Several case studies indicate potential soil C gains surpassing conventional “improved” grazing methods (what). Additionally, the relatively low-cost nature of AMP might allow for immediate deployment without major producer investments or altering production systems (when).

Yet, substantial research and practical challenges remain to fully harness AMP grazing's potential as a natural climate solution for beef production. These include but are not limited to: 1) expanding on-farm grazing studies globally with clear descriptions of the application of grazing levers, utilizing frameworks like the one provided by Stanley et al. (2024) to standardize terminology and inform pathways of grazing-soil C sequestration; 2) conducting long-term studies to assess durability amidst potential limitations such as saturation, reversibility, and context-specific factors; 3) advancing soil C-grazing modeling techniques to capture grazing pattern differences; and 4) integrating soil C into GHG LCAs across the beef supply chain for comprehensive emission mitigation evaluation. Importantly, research alone is unlikely to drive the necessary changes in the landscape at the scale, speed, and direction required. Pragmatism is essential in this context. Producers are ultimately the ones who can implement these practices and potentially reduce emissions. Thus, a pragmatic approach that balances the uncertainties of biological systems' research with the urgent need to reduce emissions using the best tools available–while more research and tools are on their way–and engaging and collaborating with producers may help us capitalize on potential reductions in beef GHG emissions, ultimately leading to more sustainable beef production systems.

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CHAPTER 2

ADVANCING BEEF SUSTAINABILITY: LONG-TERM IMPACTS OF ADAPTIVE MULTIPADDOCK GRAZING AND SOIL CARBON SEQUESTRATION IN THE MIDWESTERN USA

2.1 Introduction

Beef production has significantly influenced both terrestrial and atmospheric carbon (**C**) stocks over time. Overgrazing in beef grazing systems has contributed substantially to the historical reduction of soil C stocks in grasslands (Sanderman et al., 2017). Recent estimates indicate that livestock grazing has decreased soil C stocks by 17 ± 4% at a depth of 1 meter, resulting in a loss of 46 ± 13 petagrams (**Pg**) of C across ~24 million km^2 of grazinglands (Ren et al., 2024). Moreover, beef production is the single largest contributor to aggregated greenhouse gas (**GHG**) emissions in agriculture, accounting for about 5.9% of all human-caused GHG emissions (Gerber et al., 2013). Numerous studies have shown that beef, as currently produced, has a higher GHG footprint than other animal and plant-based proteins (Poore & Nemecek, 2018; Semba et al., 2021; Xu et al., 2021), leading to calls for reduced beef consumption as a demand-side climate mitigation strategy. This is particularly emphasized in the United States (**U.S.**), a high-income country with high per-capita and aggregated beef consumption. All this together, and as climate change becomes a more pressing issue, the future role of beef production in the U.S. agri-food systems is increasingly questioned.

At the same time, beef production offers a variety of benefits, including nutritional, social, economic, and ecological advantages (FAO, 2023). Therefore, in regions where beef production remains and will continue to remain essential for socio-economic and ecological reasons (Epstein et al., 2019; Leroy et al., 2022), there is a need for practical, effective, and low-cost strategies to reduce emissions of beef, while meeting growing consumer expectations for sustainability.

Most GHG emissions, resource use, and soil C impacts in beef production come from grazing systems, especially during the cow-calf stage, which represents most of the beef lifecycle (Rotz et al., 2019; Stackhouse-Lawson et al., 2012). However, existing and developing GHG mitigation solutions often focus on efficiency improvements in the finishing phase, which has a minor impact on the overall C balance of beef production. Additionally, in the U.S., efficiency-based solutions overall are considered to have limited scope (Cusack et al., 2021) since the American beef sector has historically been technology-driven (Drouillard, 2018), and about 95-97% of domestic production already is finished in feedlots, the industrial stage of the supply chain (MacDonald & McBride, 2009).

It is often suggested that truly sustainable agroecosystems should be not only GHG-efficient, but also long-term resilient (FABLE et al., 2020), primarily sustained by natural processes (i.e., through multifunctional landscapes that sustain services of support and regulation) rather than relying heavily on fossil fuel inputs (Jhariya et al., 2021). This aligns with the principle of "ecological intensification" rather than "sustainable intensification", which can justify any form of intensification (Tittonell, 2014). Therefore, to achieve significant GHG reduction and maintain social license, the beef industry will likely need to develop scalable GHG mitigation strategies across grazing systems while ensuring they maintain or improve other ecosystem services.

Adaptive multipaddock (**AMP**) (Stanley et al., 2018; Teague et al., 2011) grazing, a low-cost, low-input, but intensive form of rotational grazing with long and non-prescriptive recovery periods, has been proposed as an approach to help mitigate the environmental impacts of beef production, while providing economic and social benefits (Gosnell et al., 2020). This is mainly due to its ability to improve forage productivity, biodiversity, and water infiltration, as well as the C balance of beef production. It has been shown to increase soil C sequestration (Mosier et al., 2021) and, to a lesser extent, decrease GHG emissions (DeRamus et al., 2003).

Life cycle assessment (**LCA**), a widely used tool to quantify the GHG emissions associated with beef production, typically considers various emissions sources in the production process, including enteric fermentation, manure management, feed production, and on-farm energy use. However, a significant limitation of many studies using LCAs is that they often exclude soil C sequestration, which can play a crucial role in offsetting GHG emissions (Cusack et al., 2021). Additionally, they tend to model conventional grazing strategies (e.g., continuous grazing and/or high input use), with little representation of approaches like AMP grazing.

Initial evidence from beef LCAs incorporating both AMP grazing and soil C indicates significant reductions in the C footprint, ranging from a net carbon sink to ~80% reduction (Rowntree et al., 2020; Stanley et al., 2018). However, the long-term impacts of AMP on soil C and overall effects across all stages of beef production are not well understood, limiting its adoption and widespread implementation.

The purpose of this study is to identify effective grazing management strategies for mitigating climate impacts in beef production systems in the Upper Midwest. We aim to answer the following research questions: (1) How does AMP grazing influence long-term soil C sequestration? (2) What is its effect on the C footprint of beef production when integrated into an LCA model? (3) How do different finishing scenarios impact the overall C footprint of the beef system? To address these questions, we evaluate the long-term (>10 years) impact of AMP grazing on soil C and integrate these findings into an LCA model using on-farm data. Given that the cow-calf and stocker phases are typically pasture-based while most cattle are grain-finished, we model these stages under AMP grazing and examine two finishing scenarios: AMP grazing and feedlot (**FL**) finishing. Since only the finishing phase varies, the resulting cow-calf to finish systems are AMP-only and AMP+FL. Our hypothesis is that AMP grazing can reduce the C footprint of beef production systems through soil C sequestration.

2.2 Methods

We combined all data using a deterministic model developed in MS Excel to estimate annual average cradle-to-farm gate net GHG emissions for two beef production systems: AMP-only and AMP+FL (**Figure A.1; Figure A.2**). Both systems were modeled to have the same cow-calf and stocker phases managed under AMP grazing. The only difference in inputs and performance between them is in the finishing phase, where stocker steers in the AMP-only system are grass-finished, while those in the AMP+FL system are transitioned to a feedlot. While relevant unit processes were primarily estimated based on on-farm data, we supplemented gaps with peer-reviewed literature as needed.

2.2.1 Description of site and beef production systems

2.2.1.1 AMP stages description (cow-calf, stocker, and grass finishing)

Data for the three grassfed stages were sourced from farm records and research projects at the Michigan State University (**MSU**) Lake City AgBioResearch Center (**LCRC**) in northwest Michigan (Lake City; 44°18′N latitude, 85°11′W longitude, 377 m elevation; **Figure A.4**), covering an 11-year period from April 2012 to April 2023 under AMP grazing management. This long-term dataset ensures comprehensive accounting for inter-annual variability in inputs, production, and emissions, exceeding the 3-year minimum recommended by the Livestock Environmental Assessment and Performance Partnership (FAO, 2016). LCRC’s land was historically forested but has been primarily used for agriculture for several decades. From 2012 to 2023, the onsite weather station recorded an annual average rainfall of 35.05 inches and a mean temperature of 6.6 ºC (**Tables A.1 and A.2**) (US Department of Commerce, n.d.).

From the 1980s until 2010, LCRC operated as an input-intensive beef operation, characterized by continuous grazing, extensive haying, and regular application of nitrogen (**N**) fertilizer (Chiavegato et al., 2015a). In 2010-2011, LCRC transitioned to a low-input AMP-managed operation, emphasizing short-duration grazing with long and adaptive recovery periods to prioritize forage recovery, prevent overgrazing, and maintain high livestock densities for homogeneous grazing and trampling, while targeting an average forage utilization of around 50%. Moreover, the transition involved replacing the traditional Continental beef herd with Red Angus cattle of moderate frame, better aligned with the grazing philosophy.

Initially, the cowherd was managed at stock densities of around 150,000 kg/ha, with 2-3 moves per day until 2016. Subsequently, larger paddocks with longer grazing durations (one move/day) were used, aiming for densities near 80,000 kg/ha (Thompson et al., 2020). The average stocking rate for the period 2012-2023 was 1.4 animal units/ha. The average grazing season ran from May 13 to November 11, during which cattle grazed on high-quality forages with an average total digestible nutrients (**TDN**) of 60% and crude protein (**CP**) content ranging from 11% to 14%. These forages consisted of cool-season grasses and legumes, including orchardgrass (Dactylis glomerata L.), smooth bromegrass (Bromus inermis L.), Kentucky bluegrass (Poa pratensis), timothy grass (Phleum pratense), alfalfa (Medicago sativa L.), red clover (Trifolium pratense L.), white clover (Trifolium repens L.), and birdsfoot trefoil (Lotus corniculatus L.) (Chiavegato et al. 2015b; Thompson et al., 2020). In the non-grazing season, cattle were fed high-quality grass and alfalfa hay from varying cuttings (TDN 58-62%, CP 10-17%), adjusted to the animal category. Hay was fed by unrolling bales in various paddocks, with winter-feeding locations rotated throughout the farm to target specific areas for desired animal impact. It is likely that every paddock was used for feeding hay at least once during the eleven-year period (Ty Hughston, manager, LCRC, pers. comm.).

After weaning, all calves grazed on stockpiled pasture for ~35 days and were then supplemented with hay. A portion of these calves were sold after a ~10-day hay-feeding period, while the rest continued in the system. Steers entered the grass-finishing phase at ~12.5 months old, coinciding with the availability of spring grass. Simultaneously, the replacements continued to grow on the pasture.

Given the relatively consistent management and cattle numbers at LCRC from 2012 onwards, we used the annual averages of animal numbers and categories from 2012 to 2023 (**Table A.3**) to calculate annual beef production emissions under AMP management for each stage of the lifecycle. The mature cows (> 3 years old) had an average weight of 562 kg (σ = 61 kg). The cow-calf segment included a 365-day calving interval, a 179-day lactation period, and a 91% calf weaning rate. Calves were born on average on April 21 and weaned on October 17, averaging 179 days old and 222 kg (σ = 29 kg) at weaning. We considered as part of the stocker and finishing phases only animals initially intended for beef, considering grassfed finished heifers part of the cow-calf phase, a common outcome from open heifers. Of the 127 weaned calves, 113 stayed on the farm during winter, except for 14 sold 45 days post-weaning. At 12 months, 4 more stockers were sold, and 6 non-castrated males were castrated, joining the steer finishing group. After the first breeding season, 35 heifers and 7 non-castrated males were retained. Replacement heifers were included in the cowherd at an annual rate of 25%, with a 24-month age at first calving. The remaining heifers, at 19-20 months of age, were either sold as bred heifers or slaughtered if open. The cow-calf phase also yielded 17 bred cows, 13 cull open cows, and 3 culled bulls post-breeding season. The grass-finishing phase produced 47 steers.

Cattle mortality rates averaged 8.6% for sucker calves (n=12) and 0.5-1% for post-weaning categories. The Appendix (**Figure A. 2**) provides a simplified process flow for the LCRC breeding and production cattle.

AMP-finished steers averaged 517 kg (σ = 45 kg) of body weight at slaughter, with an average age of 603 days (~20 months). Total beef output also included beef from the cow-calf and stocker phases, including cull breed cows, cull bulls, finished heifers, stockers, and bred heifers and cows. Bred females and stocker cattle were sold to other farms instead of being slaughtered; we assumed they maintained their weight gained at LCRC and were eventually culled, except for two hypothetical mortalities at other farms. This assumption is significant given the substantial amount of beef leaving the farm as live animals. Dressing percentages for AMP stages were 54% for AMP steers (n=250), 55% for AMP-finished heifers and first-calf heifers culled (n=23), based on farm records. Cattle older than 24 months were assumed to yield a 50% carcass yield, following Rotz et al. (2019). To provide a conservative estimate, we applied a 50% dressing percentage to all categories sold to other farms, assuming eventual slaughter (see **Table A.4** with detailed slaughter data).

2.2.1.2 Feedlot stage

Data for the feedlot phase were sourced from Stanley et al. (2018), who used two years of on-farm data from a 90-day feedlot trial at the MSU Beef Center (East Lansing, Michigan). This trial involved steers from the LCRC (n=16), and the feedlot sourced its feed from local croplands. Recognizing this period was adjusted for logistical reasons rather than representing typical Upper Midwest timelines, Stanley et al. (2018) simulated an extension to 171 days on feed. We adapted their simulation values to estimate GHG emissions for a feedlot scenario equivalent to the number of steers in our AMP-finishing phase (n=47). For FL-finished steers, a

61.3% dressing percentage was applied to derive carcass yield, based on data from Angus-cross steers in Cooprider et al. (2011).

2.2.2 System boundaries and general modeling procedure

The system boundary for this LCA encompasses a cradle-to-farm gate assessment, covering all major GHG emissions and sinks from animal feed production to transport to the slaughterhouse. The supply chain starts with the resource extraction for agricultural inputs and ends at the slaughterhouse (**Figure A.3**). We completed the life cycle inventory by gathering data on inputs and outputs of the production systems.

Farm-stage net GHG emissions were categorized into several main sources and one sink: enteric methane (**CH_4**), manure CH_4, manure nitrous oxide (**N_2O**), feed production, on-farm energy and transportation emissions, and soil C sequestration. All emissions and sinks were directly attributed to beef production activities, so no allocation between different products or co-products was needed.

Beef cattle direct emissions, including enteric CH_4 and CH_4 and N_2O from manure, were estimated using the Intergovernmental Panel on Climate Change (**IPCC**) Tier 2 methodology (IPCC, 2006, 2019). To reduce uncertainty, we factored in annual variations in cattle management across different beef subcategories. Beef cattle were divided into fifty-nine and fifty-seven distinct subcategories for AMP-only and AMP+FL systems, respectively, based on factors such as sex, age, body weight, weight gain, physiological state, feeding conditions, weather, and diet type.

We calculated all emissions as 100-year Global Warming Potential (**GWP100**) CO_2-equivalent (**CO_2-e**) using the latest IPCC characterization factors (CO_2=1, CH_4=27, N_2O=273) (IPCC, 2021). We reported results using GWP100 to assess whether AMP management could reduce the farm-level GHG footprint of beef, providing an initial comparison with other studies due to its prevalent use in beef LCAs. For consistency with recent beef LCAs, we used one kilogram of carcass weight (**CW**) of a beef animal as the functional unit. Net GHG emissions were calculated by comparing annual resource inputs and waste output of each beef production system, expressed per kg of CW produced in 365 days, and reported as kg CO_2-e/kg CW.

2.2.3 Enteric CH_4 emissions

To estimate enteric CH_4 emissions for individual cattle categories within each production system, we used IPCC (2019) Tier 2 methodology. This method bases emissions estimates on the gross energy intake (**GEI**) of each animal, adjusted by a CH_4 conversion factor (**Ym**). Daily net energy requirements for cattle at each category were calculated based on energy used for maintenance, activity, growth, pregnancy, and lactation as applicable. The GEI needed to meet these energy requirements was then determined by considering the diet's energy density. To align with the IPCC recommendation to use national Ym values when available, we calculated enteric CH_4 emissions by applying a Ym of 6.5% for all pasture- and hay-based cattle and 3.9% for FL steers, using Midwest Ym values developed by the EPA (EPA, 2023).

To ensure the GEI estimates were biologically realistic, we divided GEI by feed energy intensity to obtain dry matter intake (**DMI**). For some categories, the DMI values were below the biologically expected minimum of 2% and lower than some LCRC measurements taken during both grazing and non-grazing seasons (LCRC Beef Report, 2014; Chiavegato et al., 2015a; Thompson et al., 2021). Thus, we recalculated DMI for all categories using the IPCC's Tier 2 simplified approach (IPCC, 2019), which predicts DMI based on body weight, animal physiological status, and diet quality. When the DMI of the simplified approach exceeded the full IPCC Tier 2 method by 10% or more (20 categories in the AMP system, 21 in AMP+FL), we used the simplified approach's DMI values to recalculate GEI, ensuring more conservative emission estimates.

The IPCC guidelines do not provide a method for modeling enteric CH_4 emissions from unweaned calves, assigning no CH_4 yield to them. Consequently, we excluded emissions from calves under four months. However, we conservatively assumed a diet transition to forage for calves aged 4-6 months, accounting for enteric CH_4 emissions based on a gradual decrease in milk intake to 75% at 4-5 months and 50% at 5-6 months, in line with the EPA's approach (EPA, 2023).

2.2.4 Manure CH_4 and N_2O emissions

Manure CH_4 and N_2O emissions were calculated using IPCC (2019) Tier 2 methods. In the three cow-calf to finish AMP phases, all manure was deposited directly by cattle on pasture, as housing was not used. Conversely, the feedlot phase involved a two-step manure management process: initial storage under slotted-floor confinement pens for approximately one year, followed by land application, which partially offset synthetic nitrogen (**N**) fertilizer in the model.

Manure CH_4 emissions depend on the quantity of manure produced by the animal and the extent to which it decomposes anaerobically (Beauchemin & McGeough, 2013). We calculated volatile solids (**VS**) production in the manure based on animal's GEI and diet digestibility. Emissions were then determined by multiplying VS production by the maximum manure CH_4 producing capacity (0.19 for all phases) and a CH_4 conversion factor specific to the manure management used (0.47 for AMP categories and 31.00 for FL-steers) (IPCC, 2019).

Direct and indirect manure N_2O emissions are estimated here, while emissions from synthetic N fertilizer applications are included in feed emissions. Soil N dynamics from pasture or cropland themselves were not modeled, based on the assumption that these emissions do not change significantly in response to beef production, following the approach of Stanley et al. (2018). We determined N excretion rates by animal category for each manure management system (**MMS**) based on animal weight gain, diet net energy for growth, and crude protein. The total N excreted from each animal via manure was multiplied by IPCC emission factors (**EFs**) specific to each MMS to estimate direct N_2O emissions.

Additionally, we accounted for N losses from manure due to volatilization, leaching, and runoff, which generate indirect N_2O emissions off-farm. The volatilized N fraction was estimated using a default IPCC EF for wet climates (0.014 kg N_2O-N; IPCC, 2019). Leaching and runoff fractions were estimated using a regression equation by Rochette et al. (2008), which establishes a relationship between precipitation (P) and potential evapotranspiration (PET). P and PET data were sourced from the MSU Enviro-weather database and averaged for the AMP stages (2012-2022) and the FL stage (2015-2016) during the growing seasons from May 15 to October 15. The fraction of N leached was determined to be 0.23 kg N leached per kg N excreted for AMP stages and 0.28 kg N leached per kg N excreted for the FL stage, similar to the IPCC default of 0.24 for wet climates (IPCC, 2019). Total manure N_2O emissions were calculated by summing emissions across all MMS for each respective system, including contributions from storage, land application, and grazing as appropriate.

2.2.5 Feed production emissions

Feed-related emissions for both systems included various on-farm and off-farm processes, such as hay production, feedlot feed production, lime application, fertilizer production and application, and supplemental minerals production.

The LCRC pastures used during the AMP stages, established over forty years ago, consistently support cool-season grasses and legumes. From 2012 to 2023, AMP-managed pastures were not irrigated, fertilized with synthetic fertilizers, or treated for pests, resulting in no GHG emissions from these processes. It was assumed that all hay produced for the AMP stages was sourced 25% from LCRC and 75% from off-farm. Additionally, total hay used was calculated assuming 15% feeding wastage (Ty Hughston, manager, LCRC, pers. comm.) and using predicted DMI. This total was cross-checked with farm hay inventories for three non-grazing seasons, confirming that our calculations, although slightly higher, provided a more conservative estimate over the 11-year period. We then multiplied the total annual hay used by an EF of 0.2 kg CO_2-e/kg DM for high-quality hay production (Stackhouse-Lawson et al., 2012).

GHG emissions from FL feed production were calculated using the average proportions of diet ingredients fed at the MSU Beef Center over two years (2015-2016) and the total feed required per animal for a 171-day simulation (Stanley et al., 2018), with predicted DMI calculated according to IPCC Tier 2 equations for enteric CH_4 (IPCC, 2019). Feed crops included high-moisture corn, dried distillers grains with solubles (**DDGS**), corn silage, corn grain, and alfalfa hay. We used Michigan-specific crop yields (USDA, 2016) to estimate the total land requirement per FL-finished steer.

For each ingredient, we multiplied the corresponding area by the GHG emissions per hectare from cultivation and manufacturing processes (i.e., insecticide, herbicide, lime, potassium oxide, phosphorus pentoxide, synthetic N, on-farm fuel use, grain drying, and transportation of feed ingredients to the feed mill) from Stanley et al. (2018) because the same production system (FL) is being modeled in this study. The authors sourced their emissions from the Farm Energy Analysis Tool (Camargo et al., 2013), and they calculated a 31.4% reduction in

N-fertilizer application due to FL manure spread on cropland, applied to all diet ingredients except alfalfa and DDGS. Additionally, they estimated GHG emissions for DDGS based on Kim and Dale (2008).

CO_2 emissions from lime application at LCRC for the AMP stages were calculated using on-farm records for application rates and U.S. Environmental Protection Agency (**EPA**) emission factors for limestone and dolomite (EPA, 2023). As lime was already included among the inputs for each crop ingredient, no additional lime was considered in the FL model.

GHG emissions from production of mineral supplements were estimated using daily mineral consumption levels for each cattle category from Lupo et al. (2013), multiplied by the EF derived from ecoinvent database (Ecoinvent Version 3.9, n.d.).

Indirect CO_2 emissions from land use conversion for the beef AMP stages were not accounted for because LCRC has supported perennial pasture-based agriculture for several decades. It was also assumed that existing hayland and cropland were used for cultivating off-farm hay and crops in the AMP-only and AMP+FL systems. GHG emissions related to machinery, infrastructure manufacturing, and seed production were excluded due to their assumed minor contribution to the overall GHG footprint (FAO, 2016; Lupo et al., 2013; Stackhouse-Lawson et al., 2012). Additionally, GHG emissions from feed crop irrigation were excluded, as it was assumed that producing FL-feed and off-farm hay for AMP stages did not involve irrigation, given that only about 10% of FL cropland in the Midwest is irrigated (Asem-Hiablie et al., 2016) and about 7% of Michigan pastures are irrigated (USDA NASS, 2024).

2.2.6 On-farm emissions from energy use and transportation

On-farm energy-related GHG emissions were estimated by summing the total usage of diesel, gasoline, and electricity, applying the respective EFs. Due to the absence of specific on-farm energy consumption data for cattle, fuel use per head was sourced from Ryan & Tiffany, (1998), as applied in previous studies in the Upper Midwest (Pelletier et al., 2010; Stanley et al., 2018). For electricity consumption, we assumed 30 kWh per animal based on values reported for Midwest beef operations, where higher values are largely attributed to irrigation needs (Asem-Hiablie et al., 2016), which do not apply to our systems. Michigan-specific EFs (EPA, 2022) were used to calculate emissions from fuel combustion and electricity production.

Transportation emissions from fuel combustion were estimated for transporting hay, feedlot feeds from the feed mill, minerals, lime, and animals to LCRC, the feedlot, or the slaughterhouse as appropriate. Estimated transport distances ranged from 30 to 80 km. Calculations assumed a standard heavy-duty diesel truck's load capacity of up to 26,000 kg, with a fuel efficiency of 5.86 mpg (Geotab, n.d.). Each truckload for cattle transport was assumed to carry up to 50 animals (Stackhouse-Lawson et al., 2012), and an empty return trip was included in all calculations (FAO, 2016).

2.2.7 Soil C sample collection

To assess 11 years of soil C sequestration from 2012 to 2023 under AMP management, soil samples were collected at LCRC in 2023.

2.2.7.1 Establishment of the 2012 baseline and subsequent 2016 sampling of AMP pastures

Baseline sampling (Time 1) took place in 2012 (Chiavegato et al., 2015b), followed by a second sampling (Time 2) in 2016 (Stanley et al., 2018), allowing for the assessment of 4-year soil C sequestration. The initial 2012 sampling occurred at three sites with sandy loam (**SL**) soil, which represents about 70% of the farm's soil types (Chiavegato et al., 2015b). In 2016, sampling expanded to nine sites total, including sandy (**S**), clay loam (**CL**), and SL types (**Figure A.4**). The original SL sites were re-sampled, and three new sites each for S and CL were added. However, the exact locations of the 2012 samples were somewhat uncertain; two of the 2016 SL sites were within 50 meters, and one was over 100 meters away from the 2012 locations, potentially introducing landscape variability into the results (Stanley et al., 2018). In both years, each site was sampled to a depth of 30 cm. In 2012, sampling at each site resulted in one composite sample comprising ten sub-samples. In 2016, sampling at each site included eight composite samples, each also consisting of ten sub-samples.

2.2.7.2 Sampling strategy for 11-year soil C assessment in AMP pastures

In the summer of 2023, we remeasured soil C stocks within 45 days of the 2012 and 2016 campaigns to assess changes over the full 11-year period. To determine the minimum sample size necessary to detect plausible changes in soil C, we conducted a power analysis using spatial heterogeneity data from the 2016 sampling, via the web app developed by Stanley et al. (2023) (available at https://scf.berkeley.edu/shiny/bosf/soil-carbon-statistics/). Given the lack of baseline soil C data on the LCRC pastures prior to cultivation and the 4-year soil C change observed post-AMP grazing implementation from 2012 to 2016, we assumed a 30% relative change compared to 2016, which indicated a need for 103 samples. To ensure consistent sampling across each site, we ultimately collected 108 soil cores, averaging 12 samples per site.

We revisited the nine georeferenced sites sampled in 2016. Although coordinates for the 2016 soil cores were not available, we located two wooden posts at each site that marked the north-south or east-west limits of where the 2016 samples were taken. To ensure we covered the 2016 sample area, we conservatively established a 25-meter buffer to the sides of the wooden posts, forming a rectangular sampling zone of 30 by 50 meters. Using QGIS, and open-source GIS software (QGIS.org, 2023), we randomly selected and marked 12 sampling spots within each of the 9 georeferenced areas. We collected the 108 soil cores and buried marker balls at each for future assessments. Following FAO (2019) guidelines, we excluded locations with animal excreta, pathways, driveways, or near watering points, using backup sampling locations as needed.

Soil samples were collected down to a depth of 30 cm, consistent with the 2012 and 2016 campaigns, and segmented at intervals of 0-10 and 10-30 cm. However, in 2023, we also sampled deeper down to 100 cm, with intervals at 30-50 and 50-100 cm, in response to expected changes in the lower soil profile and to establish a baseline for future deep soil C monitoring. For the evaluation of soil C sequestration and the LCA model, only data from the 0-30 cm depth were considered. Each depth interval was bagged on-site, resulting in 432 samples total. Samples were kept in cold storage and transported to Cquester Analytics LLC for analysis (Cquester Analytics, n.d.). There, they were processed and analyzed for soil organic carbon (**SOC**) concentration, bulk density (**BD**), and texture. Consistent with recommendations (FAO, 2018), the same core used for SOC analysis was used to estimate BD. From these measurements, we subsequently calculated soil C stocks for each sample.

2.2.8 Data analysis

2.2.8.1 Uncertainty analysis of GHG emissions

We conducted an uncertainty analysis based on procedures previously used (Rotz et al., 2019; Stackhouse-Lawson et al., 2012) to quantify the inherent uncertainties in the farm-level GHG emissions estimates of AMP-only and AMP+FL systems. We determined the uncertainties of each system's GHG footprint from the uncertainties of each major contributing emission source. Given the nature of biological systems, proper statistical quantification of the overall uncertainty is limited without comprehensive measured data for all emission sources. Thus, the uncertainty percentage for each source was set following IPCC (2006) based on expert opinion for their Tier 2 methodologies. We assigned uncertainties of ±20% for enteric and manure-derived CH_4, ±50% for N_2O, and ±20% for non-animal direct GHGs (feed production, on-farm energy use, and transportation), as used in other studies (McAuliffe et al., 2018; Rotz et al., 2019). The overall uncertainty was calculated as the square root of the sum of squares of each emission source multiplied by its estimated uncertainty. Uncertainties for AMP-only and AMP+FL are expressed as plus or minus the calculated overall uncertainty, assumed to represent a 95% confidence interval (IPCC, 2006, 2019).

2.2.8.2 Sensitivity analysis of enteric CH_4 emissions

Given the sensitivity of enteric CH_4 emissions to diet composition and intake levels, and their role as the primary contributor to the overall C emissions of beef production, we performed a sensitivity analysis. This analysis compared the impact of the IPCC Tier 2-based method to results based on measured data at LCRC (Chiavegato et al., 2015a; Thompson et al., 2021) and the simplified IPCC Tier 2 approach (IPCC, 2019). For a detailed explanation of the methodology, refer to **Section A.1** in the Appendix.

2.2.8.3 Soil C analyses: equivalent soil mass, statistical methods, and integration into LCA

To account for variations in soil BD across different sampling years in AMP pastures, we used the equivalent soil mass (**ESM**) approach according to Fowler et al. (2023), applying their equations for split-depth sampling. We compiled a dataset consisting of 186 soil samples from the 0-30 cm layer, collected at three timesteps: 2012 (6 samples), 2016 (72 samples), 2023 (108 samples). Each sample was georeferenced, and we recorded the clay+silt content. Since soil texture data was unavailable for the 2016 samples, but we sampled the same nine georeferenced sites in 2023, we used the average clay+silt content from the 2023 sites for the 2016 samples, assuming consistency across sites. This allowed us to include clay+silt content as a covariate in our statistical model. The decision to use clay+silt content as a covariate was based on a linear regression analysis, which indicated a statistically significant correlation between soil C stocks and clay+silt content ($R^2 = 0.208$, $p < 0.001$) (**Figure A.5**).

All statistical analyses were performed using RStudio (RStudio Team, 2020). To account for differences in soil texture and repeated measures across soil sites when estimating soil C stocks, we employed a linear mixed model using the lme4 package. We calculated the 11-year annual soil C sequestration rate by subtracting the 2023 soil C stock from that of 2012 and dividing by 11 years. Additionally, we assessed segmented soil C sequestration rates over 4 years (2012-2016) and 7 years (2016-2023).

To integrate the 11-year annual soil C sequestration rate into the GHG balance, we extrapolated soil C changes to AMP-managed pastures at LCRC for AMP-only and AMP+FL systems. Of the total 231 ha of pastures, 37 ha were excluded from calculations due to the planting of alfalfa and cover crop research experiments over 1-2 years, which differed from the farm’s typical landscape. Thus, the soil C sequestration rate was applied to the remaining 194 ha for AMP-only. In AMP+FL, where stocker steers are assumed to be finished in feedlots instead of LCRC, we subtracted the land typically used for AMP finishing by calculating the farm's overall stocking rate and considering the average steer weight and headcount. This left 154 ha where soil C sequestration rate was applied for the AMP+FL system at LCRC, accounting only for land use of cow-calf and stocker stages. The C sequestered for each system was then converted to kilograms of CO_2-e units and subtracted from the overall emissions of the evaluated beef systems. We assumed a steady state for soil C on land used to grow hay (for AMP and FL) and crops (for FL). Stanley et al. (2018) found that the effect of soil erosion, in the near term, on net FL emissions was practically negligible. Additionally, since most Midwest farms do not apply regenerative cropping practices such as no-till combined with cover crops (Asem-Hiablie et al., 2016), we conservatively considered soil C to be in equilibrium.

Lastly, to explore the minimum soil C sequestration rate needed to completely offset cow-calf to finish emissions in both systems, we calculated hypothetical sequestration rates based on our LCA emissions results and stocking rates. By assuming constant emissions and stocking rates, we projected when both systems would transition from being a net sink to a net source of emissions, using these hypothetical sequestration rates and incorporating our soil C measurements from 2012-2023.

2.3 Results

The following sections detail the results of our study, focusing on total CW produced, land use, GHG emissions, and soil C sequestration across different phases of each beef production system. To facilitate a clearer understanding of the numerical data presented, we begin with a visual representation that summarizes these key metrics, excluding soil C sequestration (**Figure 2.1**).

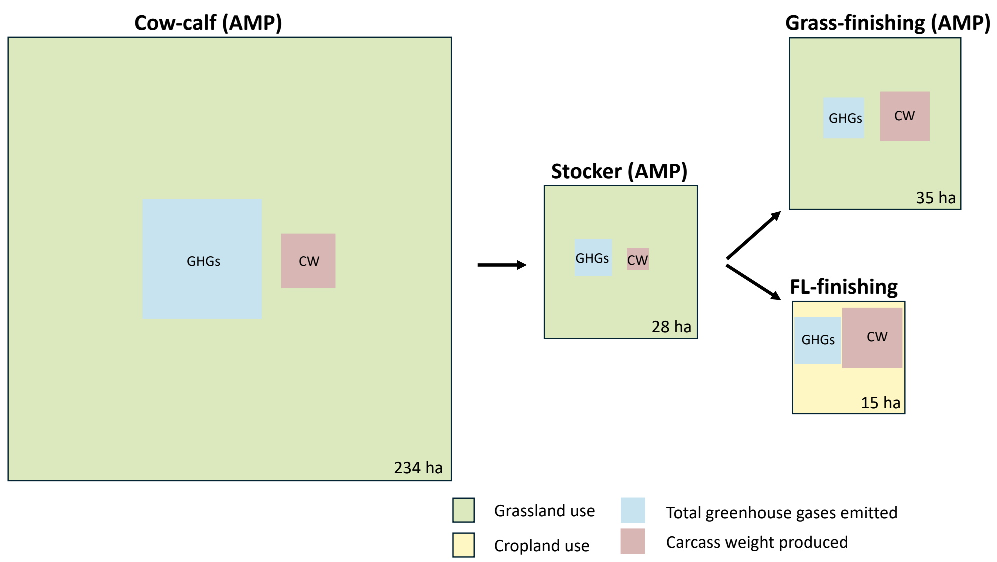

Figure 2.1. Representation of land use, GHG emissions, and CW results by the production phase for AMP-only and AMP+FL.

Figure 2.1. (cont'd)
The sizes of the squares representing GHG emissions, CW, and land use were calculated based on the largest value within each category, ensuring a proportional representation of each phase.

2.3.1 Animal production and land use

Overall yearly beef production was 17% greater in AMP+FL compared to AMP-only, totaling 36,072 kg CW yr^{-1} versus 30,381 kg CW yr^{-1}, respectively (**Table 2.1**). Both systems were modeled to have identical cow-calf and stocker performance, but steer performance in the finishing phases varied. The finishing phase contributed 51% of total meat production in AMP+FL, while it contributed 42% in AMP-only. The cow-calf stage accounted for 43% of total meat in AMP+FL and 51% in AMP-only. The stocker phase contributions were similar in both systems (Table 2.1). Overall, as expected, FL-finished steers outperformed AMP-finished in several productivity metrics, detailed in **Table 2.2**.

AMP+FL demonstrated greater productivity on a CW basis per ha, producing 129 kg CW ha^{-1}, while AMP-only produced 102 kg CW ha^{-1} (Table 2.1). Thus, to produce the same amount of CW per year, AMP-only would require about 20% more land than AMP+FL. Specifically, to produce 36,072 kg of CW annually, AMP+FL required 280 ha (94.5% pastureland for cow-calf and stocker phases, and 5.5% cropland for feedlot), while AMP-only would require 354 ha (100% pastureland).

Total feed biomass to produce 1 kg CW was 39 kg DM for AMP-only and 32 kg DM for AMP+FL. AMP-only feed consisted entirely of grass. AMP+FL comprised 93% grass, 4% grain, 2% byproducts, and 1% corn silage, indicating that about 4-7% of feed came from potentially human-edible plants (Table 2.1).

2.3.2 GHG emissions

The overall estimated GHG footprints were 29.80 ± 4.54 kg CO_2-e/kg CW for AMP-only and 25.85 ± 3.69 kg CO_2-e/kg CW for AMP+FL, with AMP-only showing ~13% greater emissions compared to AMP+FL (**Tables 2.3, Figure 2.2**).

Table 2.1. Animal productivity and land use characteristics of AMP-only and AMP+FL.

Characteristic	AMP-only	AMP+FL
Total CW produced yr^{-1} (kg)	30,381	36,072
Distribution of CW output		
Cow calf (%)	51	43
Stocker (%)	7	6
Finishing (%)	42	51
Stocking rate on grazingland (AU/ha)	1.4	1.4
Total land use (ha)	298	280
CW produced per ha (kg)	102	129
Feed biomass use (kg DM kg CW^{-1})	39	32
Feed biomass composition		
Grass[1] (%)	100	93
Grain[2] (%)	0	4
Byproducts[3] (%)	0	2
Corn silage[4] (%)	0	1

[1] Refers to grazed pasture and hay.
[2] Refers to corn grain and high moisture corn.
[3] Refers to dried distillers grains with solubles (DDGs).

Table 2.2. Animal performance characteristics of AMP-finished and FL-finished steers.

Characteristic	AMP-finished steers	FL-finished steers
Time in the finishing phase (d)	216	171
Initial-Final live weight (kg)	346(σ = 35 kg)-517(σ =45 kg)	346(σ=35 kg)-654(Stanley et al., 2018)
Dressing percentage	54%(σ = 2%)	61%(Cooprider et al., 2011)

Table 2.2. (cont'd)

Carcass weight (kg)	277	401
Average daily gain (kg/d)	0.79	1.80
Dry matter intake (kg/d)	11.5	10.7
Feed conversion ratio	14.6	5.9
Land use at finishing (ha) per steer	0.75	0.33

Relative contributions by phase of the production cycle were similar for both systems. The cow-calf phase contributed the most to GHG emissions, accounting for 83% in AMP-only and 80% in AMP+FL. The stocker phase contributed similarly to both systems, while AMP-finished steers contributed 9% of emissions and FL-finished steers 12% to their respective systems (**Table 2.4**). Emissions from the cow-calf phase included those from animals generally residing 365 days in the system, whereas stocker and finishing stages were much shorter (171 to 216 days, depending on the phase). Categorizing total GHG emissions by the source indicated that regardless of the system, enteric CH_4 was the largest contributor (72% for AMP-only; 66% for AMP+FL), and manure N_2O and feed production combined contributed to most of the rest of the emissions in both systems (about 25% for AMP-only, and 30% for AMP+FL) (Table 2.4; **Figure 2.3**).

As recommended for a more comprehensive assessment of beef management's impact on GHG emissions (Cusack et al., 2021), we calculated the estimated GHG intensity on a land area basis and found it to be about 9% lower for AMP-only compared to AMP+FL (Table 2.3).

Additionally, our sensitivity analysis of enteric CH_4 emissions for AMP-only indicated that using on-farm measured data or simplified IPCC Tier 2 emissions instead of the IPCC Tier 2 method employed in this study would result in a decrease of enteric CH_4 emissions by 3% or an increase by 10%, respectively (**Table A.5**).

Table 2.3. Breakdown of total GHG emissions and GHG intensity per unit of beef and per unit of land area for AMP-only and AMP+FL systems.

Item	AMP-only	AMP+FL
Total emissions (kg CO_2-e)	905,239	932,456
Enteric CH4	650,791	614,581
Manure CH4	7,348	19,041
Manure N2O	113,821	113,103
Feed emissions	111,578	161,679
On-farm energy and transportation	21,702	24,051
GHG intensity on a CW basis (kg CO_2-e/kg CW)	**29.80 ± 4.54**	**25.85 ± 3.69**
GHG intensity on a land area basis (kg CO_2-e/ha)	**3,036**	**3,327**

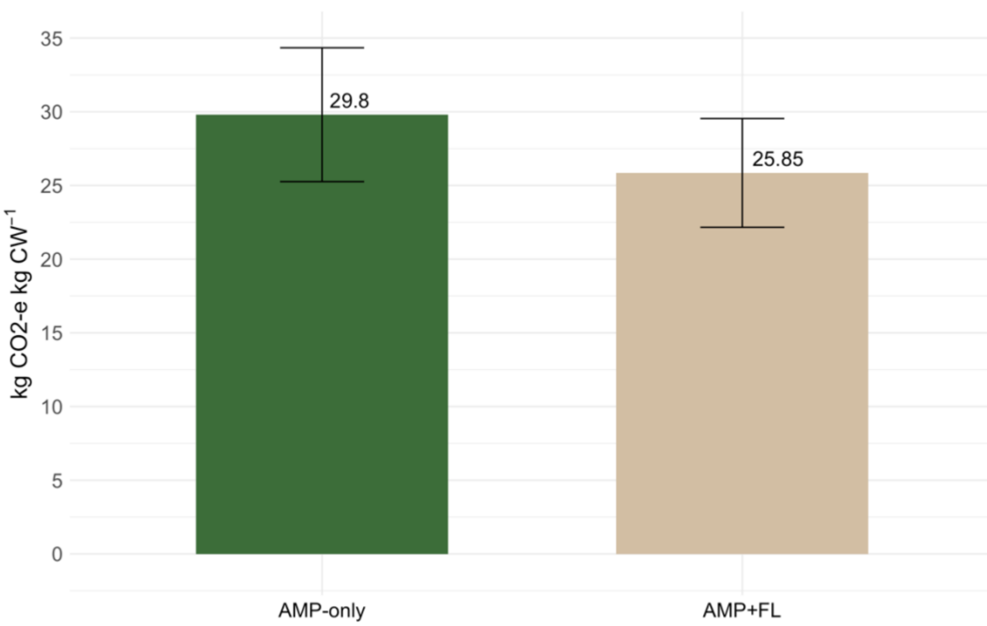

Figure 2.2. Total GHG emissions (kg CO_2-e/kg CW) for AMP-only and AMP+FL systems. Error bars represent an assumed 95% confidence interval based on AMP-only and AMP+FL whole uncertainties in relation to the point estimate (IPCC, 2006, 2019).

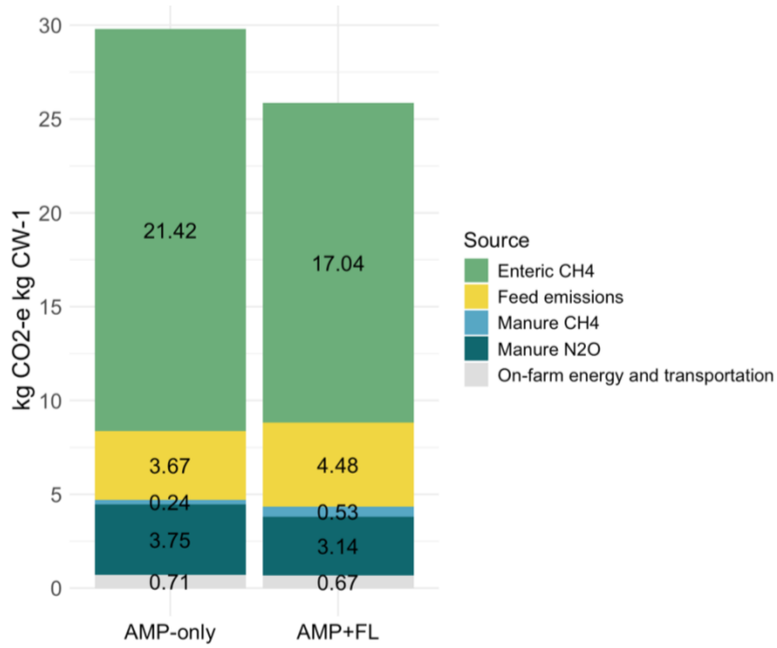

Figure 2.3. Breakdown of GHG emissions (kg CO_2-e/kg CW) by source for AMP-only and AMP+FL systems.

Table 2.4. Distribution of GHG emissions by source and beef production stage.

Item	AMP-only	AMP+FL
GHG intensity (kg CO_2-e/kg CW)	29.80	25.85
By source		
Enteric CH4	21.42 (72%)	17.04 (66%)
Manure CH4	0.24 (1%)	0.53 (2%)
Manure N2O	3.75 (13%)	3.14 (12%)
Feed emissions	3.67 (12%)	4.48 (17%)
On-farm energy and transportation	0.71 (2%)	0.67 (3%)
By production stage		
Cow-calf	24.69 (83%)	20.80 (80%)
Stocker	2.30 (8%)	1.93 (8%)
Finishing	2.81 (9%)	3.12 (12%)

2.3.3 Soil C sequestration

After correcting soil C stocks by equivalent soil mass and accounting for clay+silt as a covariate, the 2023 soil C stock in the top 30 cm was 18.72 Mg C ha^{-1} greater than the 2012 baseline (37.6 Mg C ha^{-1}). These data indicate that, on average, LCRC's pastures under 11-year AMP management sequestered soil C at 1.70 Mg ha^{-1} yr^{-1}. This rate, used to calculate the 11-year net GHG emissions, was not linear. During the first 4 yrs (2012-2016), soil C increased at a rate of 3.73 Mg C ha^{-1} yr^{-1}, while during the subsequent 7 yrs (2016-2023), it continued to sequester but at a lower rate of 0.54 Mg C ha^{-1} yr^{-1} (**Figure 2.4**).

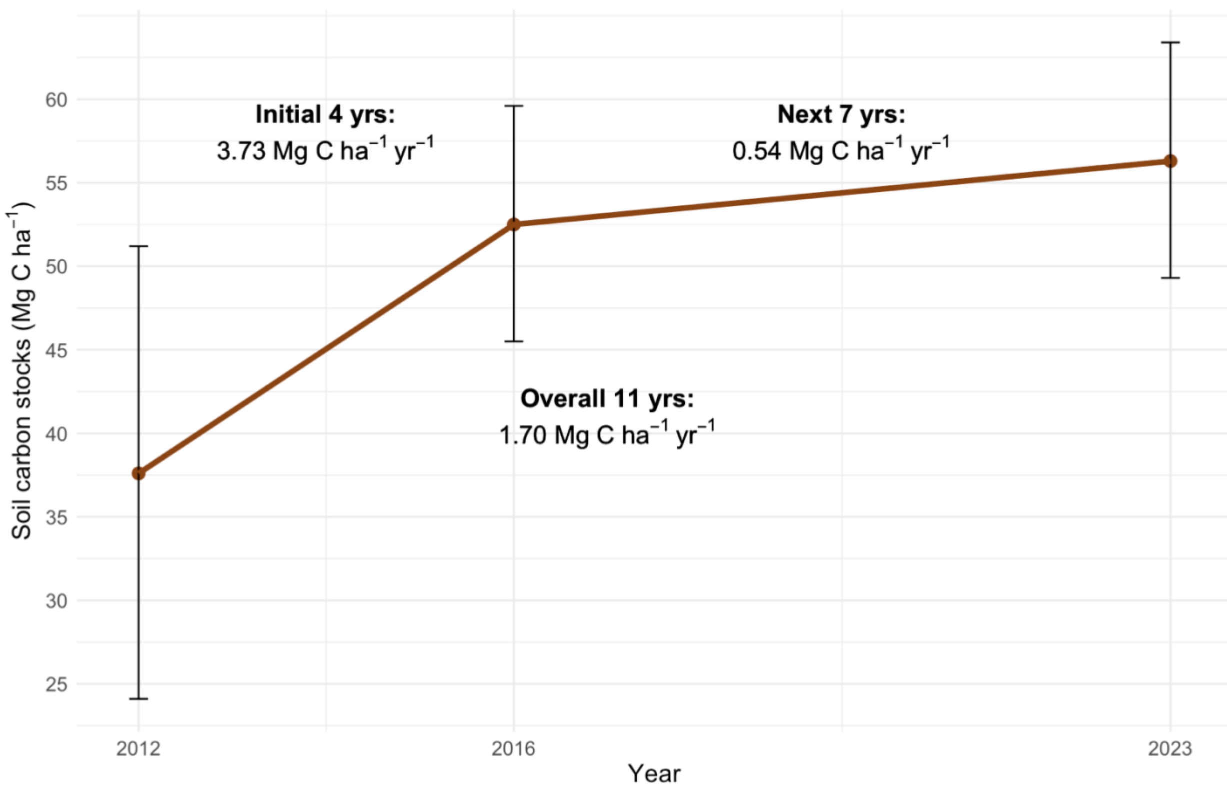

Figure 2.4. **Soil C stock (Mg C ha^{-1}) in the top 30 cm monitored over 11 years in adaptive multipaddock (AMP) pastures at Lake City AgBioResearch Center.**
Error bars represent 95% confidence intervals.

2.3.4 Net GHG footprint

We integrated 11-year measured soil C sequestration into the GHG balance for all AMP stages in both AMP-only and AMP+FL systems, assuming C sequestration across all AMP-managed land (except those planted in the 11-year period). Incorporating soil C sequestration as a GHG sink reduced emissions by 134% (from 29.80 to -10.04 kg CO_2-e/kg CW) and 109% (from 25.85 to -2.41 kg CO_2-e/kg CW) for AMP and AMP+FL, respectively, making both systems net sinks (**Figure 2.5**).

To assess emission variations based on observed changes in soil C sequestration rates over time, we tested net GHG emissions for both systems using the initial 4-year (3.73 Mg C ha^{-1} yr^{-1}) and subsequent 7-year (0.54 Mg C ha^{-1} yr^{-1}) sequestration rates. Under the initial higher sequestration rate, both systems were estimated to be significant net sinks. However, under the lower sequestration rate observed in the following 7 years, both systems became net sources of emissions with similar GHG intensity (**Figure 2.6**).

The minimum soil C sequestration rates in AMP stages needed to offset 100% of the GHG emissions of both systems were estimated to be 1.25 Mg C ha^{-1} yr^{-1} for AMP-only and 1.52 Mg C ha^{-1} yr^{-1} for AMP+FL (about 26% and 10% lower, respectively, than the 11-year observed rate of 1.7 Mg C ha^{-1} yr^{-1}). Assuming constant emissions, stocking rates, and a continued sequestration rate of 0.54 Mg C ha^{-1} yr^{-1} from 2023 onwards, AMP-only would become a net GHG source (at 0.51 kg CO_2-e/kg CW) in 2030 or year 18 after conversion and AMP+FL (at 0.54 kg CO_2-e/kg CW) in 2025 or year 13 after conversion (**Figure 2.7**). With a 50% reduction in the sequestration rate (0.27 Mg C ha^{-1} yr^{-1}), AMP-only and AMP+FL would become net sources in 16 and 13 years post-conversion, respectively (data not shown).

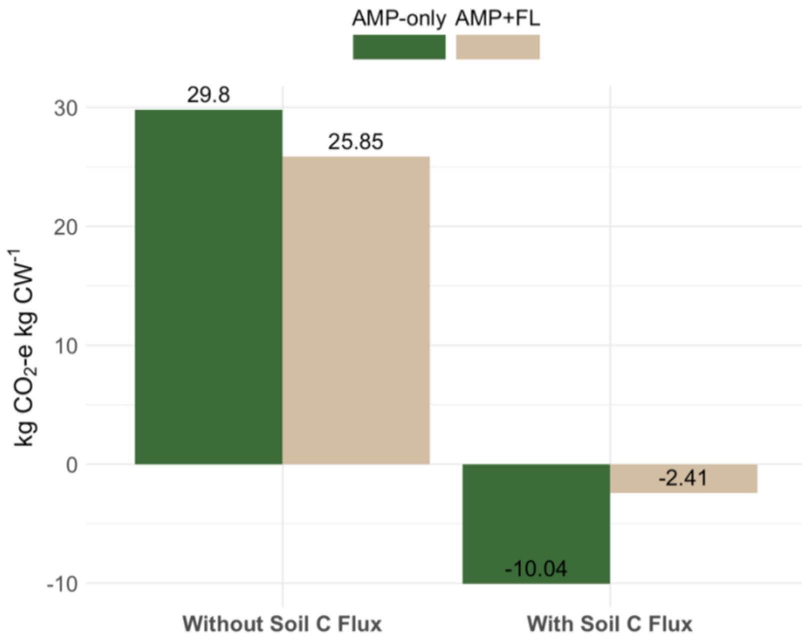

Figure 2.5. Estimated annual GHG emissions (kg CO_2-e/kg CW) for each production system – AMP-only and AMP+FL – before (left) and after (right) incorporating soil C sequestration.

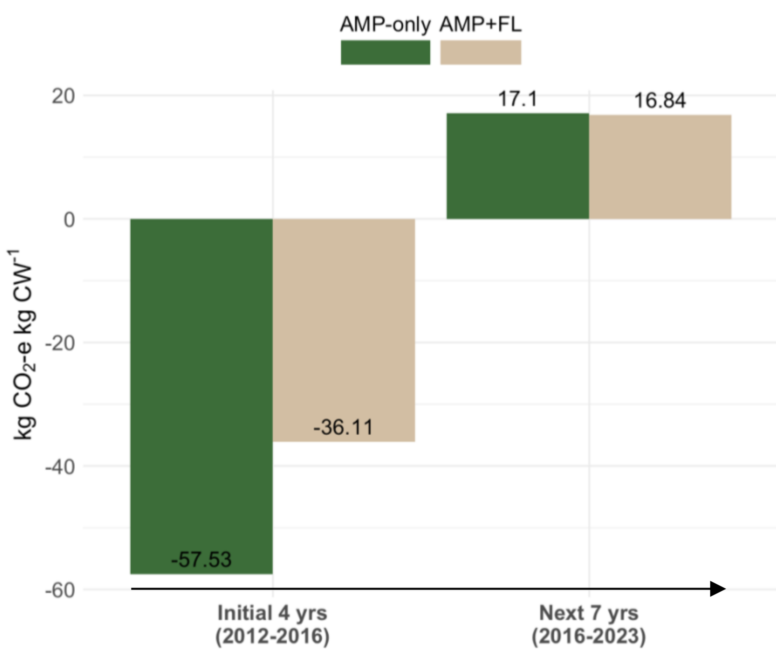

Figure 2.6. Estimated annual GHG emissions (kg CO_2-e/kg CW) for each production system – AMP-only and AMP+FL – when incorporating soil C sequestration rates from the initial 4-yr period and the subsequent 7-yr period of AMP implementation.

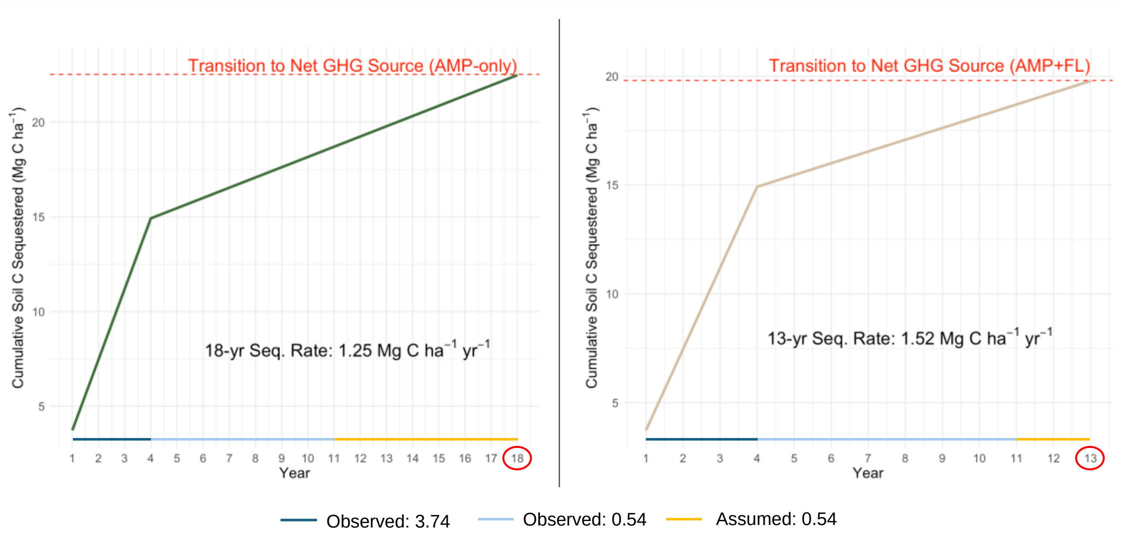

Figure 2.7. Projection of point in time when AMP-only and AMP+FL would become net GHG sources assuming constant GHG emissions, stocking rates, and soil C sequestration rate.

We assumed soil C sequestration of 0.54 Mg C ha^{-1} yr^{-1}, the same rate observed during the 4-11 year period.

2.4 Discussion

Evaluating the net GHG impacts of the modeled beef production systems requires discussing interconnected elements, including beef yield, land use, whole-farm emissions, and soil C sequestration.

2.4.1 Animal production and land use

Our results showed that AMP+FL produced about 21% more beef per hectare than AMP-only (Table 2.1), aligning with other studies that highlight the efficiency benefits of grain-finishing cattle over grass-finishing (Klopatek et al., 2021; Pelletier et al., 2010). Days to finish and finishing weights for our FL steers were similar to Midwest averages (171 days and 654 kg vs. 175 days and 617 kg, respectively; Asem-Hiablie et al., 2016). Additionally, our 11-year average for AMP steers showed a slight reduction in finishing weight and an increase in days to finishing (-11 kg and +15 days, respectively) compared to the initial 5-year average reported by Stanley et al. (2018) at LCRC. However, our 11-year average finishing weight is slightly higher

(+12 kg) and finishing time is faster (about 30 days less) than the pasture-finished steers reported by Pelletier et al. (2010) for the Upper Midwest.

Our results align with other studies showing that more land is needed to produce the same amount of beef on grass compared to feedlot subsystems (Table 2.1). However, our findings emphasize the need for a whole-system perspective and consideration of the current production context when evaluating beef production sustainability. Similar to GHG emissions, most land use in our modeled beef systems came from the cow-calf stage, with smaller contributions from pasture for stockers or grass-finishers and cropland for feedlots (Figure 2.1). In the Upper Midwest and the U.S. overall, most beef production involves pasture-based cow-calf and stocker operations combined with grain-based feedlots (Drouillard, 2018; Pelletier et al., 2010). While feedlots currently allow for finishing beef cattle with less land and at heavier weights, small increases in stocking rates and carrying capacity (expressed in AU/ha) within the cow-calf stage may further enhance land use efficiency in overall cow-calf to finish systems. Our 11-year average annual stocking rate for AMP subsystems of 1.4 AU/ha (Table 2.1) is comparable to or slightly higher than those reported for cow-calf operations in the Midwest (1.14 ha/cow-calf pair; Asem-Hiablie et al., 2016) and Michigan (1.01-1.83 ha/cow-calf pair; Lindquist, 2014). The cropland needed to finish a FL steer (Table 2.2) was about 25% higher than that reported for feedlots in the Midwest (Asem-Hiablie et al., 2016).

Taken together, our animal performance and land use results suggest that our low-input grassfed-based systems implemented through AMP-only and AMP+FL can maintain or slightly increase beef output per hectare compared to current cow-calf to grain or grass finish systems in the Midwest. This is crucial to ensure that potential mitigation practices do not reduce productivity.

2.4.2 GHG emissions

In this study, we estimated the average 11-year GHG emissions of LCRC using AMP grazing to evaluate the long-term potential for emissions reduction and fossil fuel limitation by improving grazing management. This approach extends our understanding beyond previous LCAs, which assessed AMP grazing over shorter temporal boundaries (Rowntree et al., 2020; Stanley et al., 2018).

Overall, our GHG emissions (per kg CW) fell within the range reported for the Midwest (Rotz et al., 2019) and Upper Midwest (Pelletier et al., 2010). Rotz et al. (2019) established U.S. regional baselines of business-as-usual GHG emissions in beef production, providing benchmarks for evaluating GHG mitigation strategies. For specialized beef production systems (i.e., excluding dairy beef) in the Midwest, the reported mean was 25.5 kg CO_2-e/kg CW, with individual systems ranging from 20.6 to 30.9 kg CO_2-e/kg CW.

Our AMP+FL emissions (25.85 kg CO_2-e/kg CW) align closely with the regional mean. This nearly exact agreement may be partly fortuitous due to differences in methodologies. Rotz et al. (2019) used a process-based model (Integrated Farm System Model) to estimate emissions, which falls within the Tier 3 approach category according to IPCC (2019). Conversely, we used the IPCC Tier 2 approach and made conservative adjustments to DMI. We hypothesize that our approach likely overestimates enteric CH_4 emissions (and therefore overall emissions), as on-site measurements on lactating cows and finishers at LCRC during the grazing season (Chiavegato et al., 2015a; Thompson et al., 2021) showed enteric CH_4 emissions approximately 17% lower for lactating cows and 24% lower for finishers compared to our IPCC-based model estimates (Table A.5). However, this reduction was not fully captured in our sensitivity analysis, where using these measurements and extrapolating to all other animal categories in AMP-only resulted in only 3% lower emissions than our model (Table A.5), likely diluted due to the extrapolation method based on metabolic body weight. This underscores the challenges of accurately accounting for cow-calf to finish enteric CH_4 emissions when multiple beef categories require precise measurement or modeling.

AMP-only emissions (Figure 2.2) fall within the upper range for Midwest operations reported by Rotz et al. (2019). However, the authors modeled feedlot as the only finishing strategy. It is therefore reasonable to hypothesize that other grass-finished cow-calf to finish operations may yield comparable or higher GHG emissions than those modeled in AMP-only. For instance, Pelletier et al. (2010) modeled a grass-finished cow-calf to finish operation in the Upper Midwest and found a GHG intensity about 15% greater that our AMP-only system (34.9 kg CO_2-e/kg CW, using a 0.55 carcass yield to convert from live weight emissions). This small but meaningful difference may be attributed to the fertilizer modeled in their managed pastures, along with lighter finishers and longer time to finishing (Pelletier et al., 2010).

Additionally, the GHG intensities of both our modeled systems fall within the range reported in kg CO_2-e/kg CW for the Northern Great Plains (23-31.6; Lupo et al., 2013), California (21.3-22.6; Stackhouse-Lawson et al., 2012), and the Southeastern (33.55; Rowntree et al., 2020), within the U.S. However, GHG intensity comparisons across different regions should be made more cautiously due to the inevitable variation in agricultural inputs by region (Stackhouse-Lawson, 2012).

Our emission breakdown by production stage (Table 2.4) underscores the pivotal role of the cow-calf phase in beef sustainability, accounting for more than 80% of total emissions in both systems. These findings are consistent with previous North American studies, which also identified the cow-calf phase as a major contributor to the total GHG footprint (68-80%), while stocker/backgrounding and finishing phases contribute less significantly (8-22% and 8-27%, respectively) (Alemu et al., 2017; Beauchemin et al., 2010; Stackhouse-Lawson et al., 2012).

Our emission breakdown by sources (Table 2.4) emphasizes the dominant role of direct animal emissions (i.e., enteric CH_4 and manure emissions) in both systems, representing about 80% of total emissions. Previous studies have reported similar distributions, with enteric CH_4 contributing 63-81% (Alemu et al., 2017; Beauchemin et al., 2010; Rowntree et al., 2020; Wang et al., 2015), manure N_2O contributing 14-16% (Alemu et al., 2017; Wang et al., 2015), and manure CH_4 contributing 2-10% (Alemu et al., 2017; Wang et al., 2015). Although feed production emissions can be comparable to or significantly higher than manure N_2O emissions (as observed in our study, Table 2.4, and in Pelletier et al. (2010), respectively), the remaining emissions typically constitute a minor fraction.

While we observed some differences in the emission profiles of our modeled systems due to distinct manure management and feed production among finishing strategies (Table 2.4), the cow-calf to finish perspective underscores the minor role these differences play in the total footprint (Figure 2.1). Direct emissions, particularly enteric CH_4 from the cow-calf phase, are the main source of emissions. In our study, the cowherd accounted for 83-88% of enteric CH_4 emissions, aligning closely with the 84% reported by Beauchemin et al. (2010).

In this study, we aimed to be conservative when calculating enteric CH_4 emissions as the main goal was to evaluate the impact of soil C sequestration on the net C footprint. Future studies evaluating AMP-managed cattle and enteric CH_4 should consider the IPCC Tier 2 modeling limitations and cattle grazing behavior under AMP grazing. For instance, we selected our Ym for grassfed cattle based on on-farm pasture and hay digestibility samples. However, as the lower and less-digestible parts of the plant are typically not eaten under AMP grazing at

LCRC due to the goal of about 50% forage utilization, but they are typically included in the forage samples for analysis, it is possible that we were unable to capture the actual forage quality consumed by the animals. These nuances may be important for achieving meaningful GHG efficiency improvements in grazing settings.

2.4.3 Soil sampling considerations and soil C sequestration

The study's main objectives were to evaluate AMP grazing's long-term potential to preserve and increase soil C stocks in historically continuously grazed pastures, and to integrate these changes into a GHG model to quantify potential GHG mitigation in cow-calf to finish beef systems. To date, addressing these interconnected objectives has been challenging and only partially achieved due to several factors: few studies on grazing and soil C have truly applied AMP grazing (Teague et al., 2013), most beef LCAs have assumed soil C equilibrium (Cusack et al., 2021), and the ones that have included soil C typically have relied on literature values (Lupo et al., 2013; Pelletier et al., 2010), have used space-for-time substitution (i.e., chronosequence approach) instead of direct measurements over time (Machmuller et al., 2015; Rowntree et al., 2020; Wang et al., 2015), land use history has been degraded croplands (Machmuller et al., 2015; Rowntree et al., 2020), and have been short-term when including on-site sampling over time and land use history has been overgrazed grazingland (Stanley et al., 2018).

Our study builds on the initial findings reported by Stanley et al. (2018) in the AMP-finishing stage at LCRC. We extended the evaluation period and included all beef production stages. This provides long-term on-farm emissions data and soil C measurements over time, potentially offering a more accurate approximation of AMP grazing’s net GHG impacts over longer-term scales, despite some inherent limitations. More specifically, our baseline soil C measurements were particularly subject to spatial heterogeneity and low sampling density–

common challenges in soil C and grazing studies (Stanley et al., 2023)–increasing potential for variability compared to the second and third sampling periods, as illustrated by our wider confidence intervals (Figure 2.4). To mitigate these limitations, we included three additional samples for the 2012 baseline, taken 2.5 months earlier than those used by Stanley et al. (2018), which ultimately increased the baseline. Additionally, we corrected soil C stocks using the equivalent soil mass approach and accounted for combined clay+silt content as a covariate, enhancing our confidence in the results. Despite these adjustments, caution should be exercised when extrapolating the observed soil C sequestration for the 11-year period.

Overall, our soil data indicate that AMP grazing, applied during the cow-calf and stocker phases, with or without the finishing phase, can significantly increase soil C stocks in pastures with a history of continuous grazing and haying in the Upper Midwest over the long-term (Figure 2.4). We estimated an 11-year soil C sequestration rate of 1.7 Mg C ha^{-1} yr^{-1} in AMP-managed pastures, highlighting the potential for soil C accrual through improved management. Prior work has shown that AMP grazing can lead to greater soil C outcomes than continuous grazing strategies (Mosier et al., 2021, 2022; Wang et al., 2015). In contrast, continuous grazing typically reduces plant cover, diversity, and productivity, which diminishes root inputs and plant- and microbial-mediated soil C accumulation. This grazing strategy also exacerbates soil C losses through higher microbial turnover and erosion due to greater soil compaction and reduced ground cover (Bai & Cotrufo, 2022).

Comparing our 11-year soil C sequestration rate with other studies using AMP grazing, we find it lower than the rates reported in the Southeastern U.S. (2.29 Mg C ha^{-1} yr^{-1} over 20 years; Rowntree et al., 2020; and 8.0 Mg C ha^{-1} yr^{-1} over 7 years; Machmuller et al., 2015) and the Southern Great Plains (3.5 Mg C ha^{-1} yr^{-1} over 9 years; Wang et al., 2015). On the other hand, our estimate greatly exceeds the 39-year average rate of 0.28 Mg C ha^{-1} yr^{-1} reported for "improved grazing management" in a global synthesis, which included mostly U.S. studies but limited rotational grazing data (Conant et al., 2017). Overall, these studies indicate significant potential for soil C sequestration over several years. Importantly, there is great variation across these studies in soil sampling, depth, and other factors.

However, our study found that the sequestration rate dropped by 85% from year 4 to year 11 after implementing AMP practices (Figure 2.4). While this decline was somewhat expected, the magnitude of the reduction was surprising given that generally, the same management practices were applied over the 11-year period. However, since 2016, a shift from a more intensive to a less intensive AMP approach occurred. Specifically, cattle were moved from 2-3 times per day to once daily, reducing stocking densities by about 50% (Thompson et al., 2020). This reduction in stocking density may have contributed to the decreased sequestration rate, as higher densities lead to more uniform forage use, even distribution of urine and feces, and increased trampling of vegetation, which enhance nutrient cycling and overall ecosystem functioning (Thompson et al., 2023). In the few studies where longitudinal soil C data were available, generally, lower sequestration rates were observed over time (Machmuller et al., 2015; Rowntree et al., 2020). Specifically, Machmuller et al. (2015) reported an apparent plateau in sequestration by year six out of seven. Nevertheless, it is important to note that each managed nexus of grazingland and cattle will yield different results due to various biotic and abiotic factors such as land use history (e.g., conversion from overgrazed grazingland or degraded cropland), soil and climate conditions, and other elements often beyond the control of farmers. Therefore, taking into account farm-level differences is important.

Considering the soil C gains and land history of other studies using AMP grazing, along with the specific farm conditions of our study, we believe our observed 11-year sequestration is reasonable for several reasons. Although the initial average soil C stock in our study (Figure 2.4) was not as low as those reported in degraded croplands (Machmuller et al., 2015; Rowntree et al., 2020), +30 years of continuous haying and grazing may explain the degraded land and subsequent high soil C sequestration observed since starting AMP management in the perennial pastures. Stanley et al. (2024) introduced a framework to understand how managing specific grazing levers–frequency, timing, duration, intensity–could optimize soil C sequestration. They outlined three pathways: increasing soil C inputs, reducing soil C losses, and enhancing efficiency of below-ground transformations. Grazing management at LCRC has adapted the four grazing levers to prevent overgrazing. This overarching goal has led to increased fencing infrastructure to create smaller paddocks (over 25 paddocks per grazing group), grazed paddocks with low frequency (2-3 times per year), with planned and adjusted recovery periods (50-90 days) to promote growth of cool-season grasses and legumes, 1-2 moves per day, and generally aiming for 50% grazing intensity (i.e., biomass utilization) (Chiavegato et al., 2015a; LCRC Beef Report, 2014; Thompson et al., 2020). We hypothesize that consistently applying these practices with daily monitoring and necessary adjustments on degraded pastures has improved key ecophysiological factors affecting soil C sequestration pathways, such as increased ground and canopy cover, enhanced forage productivity, improved aboveground input quality through more evenly distributed manure, and greater functional group diversity (Stanley et al., 2024).

Although we assumed that croplands and pasture for hay are in soil C equilibrium, there is room for improvement in farm practices at this level. For instance, incorporating cover crops–which about 27% of feedlots with croplands have adopted in the U.S. (Bowman et al., 2024)–has been reported to increase soil C stocks by about 0.3 Mg C ha^{-1} yr^{-1} (Poeplau & Don, 2015). However, the benefits of feedlot cropland management on the beef's GHG footprint may be minor due to the small fraction of land use that feedlots represent from a cow-calf to finish perspective (Figure 2.1). Despite this, if large-scale feedlots managing extensive hectares were to enhance their cropland management practices, they could potentially make a significant impact on the overall C footprint of beef production. Input management (e.g., N fertilizer) in hay operations may also help reduce net GHG emissions. This is particularly important in regions like the Upper Midwest, where beef cattle diets rely heavily on hay for about half of the year, and this hay is typically produced using diverse range of fossil-derived inputs (Asem-Hiablie et al., 2016).

2.4.4 Net GHG emissions

The overarching goal of our study was to assess the long-term effectiveness of AMP grazing in mitigating GHG emissions in beef production, primarily through soil C sequestration. Our 11-year assessment demonstrated that incorporating soil C change into the LCA boundaries significantly reduced (>100%) the GHG footprint of both modeled systems. After accounting for soil C sequestration, both AMP-only and AMP+FL became net sinks, reducing their GHG footprint by about 133% and 109%, respectively (Figure 2.5). The larger reduction in AMP-only is attributed to its greater use of grazingland.

Compared to other literature integrating soil C into the GHG balance, our values tend to be at the upper end of mitigation magnitude. For instance, Pelletier et al. (2010) estimated that hypothetical soil C sequestration rates of 0.12 Mg C ha^{-1} yr^{-1} could offset 43% of GHG emissions in a grassfed Upper Midwest beef system. Similarly, Lupo et al. (2013) reported a 24% reduction in continuous grassfed beef production emissions when including a literature sequestration rate of 0.41 Mg C ha^{-1} yr^{-1}. Additionally, Rowntree et al. (2020) estimated an 85% reduction in the GHG footprint of a grassfed cow-calf to finish operation under AMP management when including a soil C sequestration of 2.29 Mg C ha^{-1} yr^{-1} into the GHG model. Despite different sequestration rates, varying stocking rates may partly explain the different impact of soil C on net emissions in these studies.

Determining how long our modeled beef systems can remain as a C sink requires longer-term assessments. However, our results showed that after year 4, both systems would become net sources if the initial sequestration rate were not factored into the GHG model. Additionally, our projection-under several but reasonable assumptions- revealed both systems would be net emitters by 2025 (AMP+FL) or 2030 (AMP-only). This suggests it is unlikely for these systems to remain sinks in the mid to long term with relatively high stocking rates. However, for several years, they will likely still present low GHG intensities.

Overall, our results highlight the significant impact of initial high soil C sequestration rates on the total long-term GHG footprint of beef production, underscoring the importance of establishing accurate soil C baselines. Additionally, our findings demonstrated that AMP grazing in cow-calf to finish systems can be an effective GHG mitigation tool, even when cattle are grain-finished. Longer-term monitoring should assess potential saturation in the top-30cm layer and include deeper soil sampling. Equally important, holistic and realistic GHG mitigation targets should be established to fully evaluate the effectiveness of AMP grazing as a mitigation strategy in beef production, moving beyond the simplistic dichotomy of sink or source.

2.5 Conclusions

This study set out to assess the long-term effects of AMP grazing on soil C sequestration in historically overgrazed and hayed pastures, and the overall net GHG emissions of beef production. To our knowledge, this study provides the most in-depth assessment of AMP grazing's impact on the GHG balance of beef production systems to date. We attribute this to our long-term (11 yrs) on-farm soil C measurements and detailed on-farm data of commercial-sized cow-calf to finish beef cattle AMP-managed over the same period, which were used to model net GHG emissions. Specifically, we calculated net GHG emissions for two systems in the Upper Midwest using the LCA methodology: AMP-only and AMP+FL, deriving several important insights.

Our results demonstrate that rotational-intensive (not input-intensive) grazing management, focused on preventing overgrazing and improving ecological function while maintaining or improving beef production compared to conventional grazing systems, can substantially reduce GHG emissions from cow-calf to grain- or grass-finished, ultimately becoming net sinks through prolonged soil C sequestration. Comparing AMP-only, which applies AMP across all beef production phases, to AMP+FL, which applies AMP during cow-calf and stocker phases but finishes cattle in a feedlot, AMP-only showed greater and potentially longer-lasting GHG mitigation via soil C sequestration. However, it would require ~20% more land to produce the same amount of CW as AMP+FL.

Our findings provide insights into the highly underrepresented AMP grazing practice in scientific literature and directly address the gap of lacking meaningful GHG mitigation strategies in beef production. By focusing on the pasture-based phases (i.e., cow-calf and stocker), which are responsible for most GHG emissions, we demonstrate that implementing AMP grazing can reduce emissions by sequestering soil C and lowering fossil fuel inputs without needing land expansion.

Continued research should focus on the impact of AMP across different ecoregions, longer-term effects, deeper soil depths, and developing an accurate representation of AMP practices in soil C and GHG models for large-scale monitoring. Such research could enhance our understanding of the full potential of AMP grazing as a climate mitigation strategy. We have shown that ecological intensification of beef grazing systems in the Upper Midwest can: 1) create low-input but productive operations, yielding high meat production from mostly non-human edible sources, 2) preserve and increase soil C stocks, and 3) completely offset (at least for about a decade) the GHG emissions of cow-calf to finish systems. Therefore, we suggest incentivizing AMP practices in current grazing operations alongside ongoing research rather than sequentially, given the relevant timescales required to observe soil C sequestration and ultimately influence the C balance of beef production.

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APPENDIX CHAPTER 2

Table A.1. Monthly precipitation (in) for 2022-2023 at the MSU Lake City AgBioResearch Center.

Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
2012	2.01	1.34	3.96	2.20	5.30	3.03	7.32	1.97	3.45	4.35	0.42	3.46
2013	3.70	1.94	1.00	5.09	3.02	1.87	2.03	4.15	1.66	3.09	5.80	2.01
2014	1.64	1.26	1.19	6.58	3.29	2.94	3.17	1.69	4.07	4.29	3.33	0.90
2015	1.00	0.46	0.55	2.58	4.57	2.91	2.25	4.10	4.14	2.78	3.81	3.86
2016	1.48	2.07	3.78	2.20	2.26	2.21	5.74	2.25	3.30	3.07	1.68	2.16
2017	2.83	1.86	2.92	5.50	2.78	4.96	2.43	2.31	1.66	7.62	2.42	1.52
2018	1.29	2.23	0.75	3.69	3.70	1.01	2.24	3.69	2.15	5.00	1.97	1.43
2019	2.37	2.71	1.59	3.32	4.00	5.57	1.74	2.14	5.29	5.49	2.50	3.61
2020	2.26	1.07	2.80	3.35	5.73	2.92	3.02	3.59	2.53	3.82	2.17	1.30
2021	1.10	0.80	2.36	1.82	4.21	2.81	5.43	6.32	2.81	4.38	1.51	3.77
2022	1.04	1.04	3.89	3.70	2.14	2.34	2.15	4.44	4.22	3.91	2.01	1.25
2023	2.21	2.21	2.36	3.46	1.07	0.99	3.36	3.83	0.99	2.34	1.83	1.84
11-year mean	***1.90***	***1.60***	***2.11***	***3.64***	***3.73***	***2.96***	***3.41***	***3.33***	***3.21***	***4.35***	***2.51***	***2.30***

Source: Lake City Exp Farm, NOAA Online Weather Data at https://www.weather.gov/wrh/Climate?wfo=apx

Table A.2. Monthly temperature (ºC) for 2022-2023 at the MSU Lake City AgBioResearch Center.

Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
2012	-5.0	-2.6	6.2	6.3	14.3	18.5	22.9	18.6	13.3	7.5	1.5	-1.4
2013	-6.1	-7.2	M	2.9	13.3	17.4	20.1	18.3	13.9	8.1	0.4	-8.0
2014	-11.5	-11.1	-8.4	4.4	11.8	18.1	17.4	18.5	14.2	7.9	-1.5	-2.7
2015	-9.9	-15.1	-3.3	5.3	13.6	16.1	19.0	18.9	17.0	8.0	4.3	1.1
2016	-6.0	-4.6	1.6	4.1	12.5	17.3	20.7	20.6	16.7	9.9	5.0	-4.3
2017	-4.8	-2.4	-1.9	7.9	11.1	17.8	19.6	17.7	15.9	10.8	0.7	-7.4
2018	-7.4	-5.2	-2	0.4	15.4	17.8	21.1	21.0	15.7	6.6	-1.5	-2.7
2019	-10.1	-7.7	-4.5	5.0	10.2	16.1	20.6	17.7	15.6	6.9	-1.6	-2.8
2020	-3.8	-5.6	0.2	3.6	10.8	17.6	21.7	19.7	12.9	5.4	3.8	-2.8
2021	-5.2	-9.1	1.2	6.3	11.4	19.1	19.6	20.4	14.6	12.2	1.2	-2.6
2022	-10.1	-7.7	-2.6	3.1	13.8	17.5	19.9	18.9	14.9	7.6	3.0	-3.6
2023	-2.5	-4.3	-1.8	6.8	11.8	17.8	19.8	17.8	15.7	9.6	1.6	1.2
11-year mean	***-7.0***	***-7.3***	***-2.2***	***4.5***	***12.6***	***17.6***	***20.2***	***19.1***	***15.0***	***8.3***	***1.4***	***-3.4***

Source: Lake City Exp Farm, NOAA Online Weather Data at https://www.weather.gov/wrh/Climate?wfo=apx

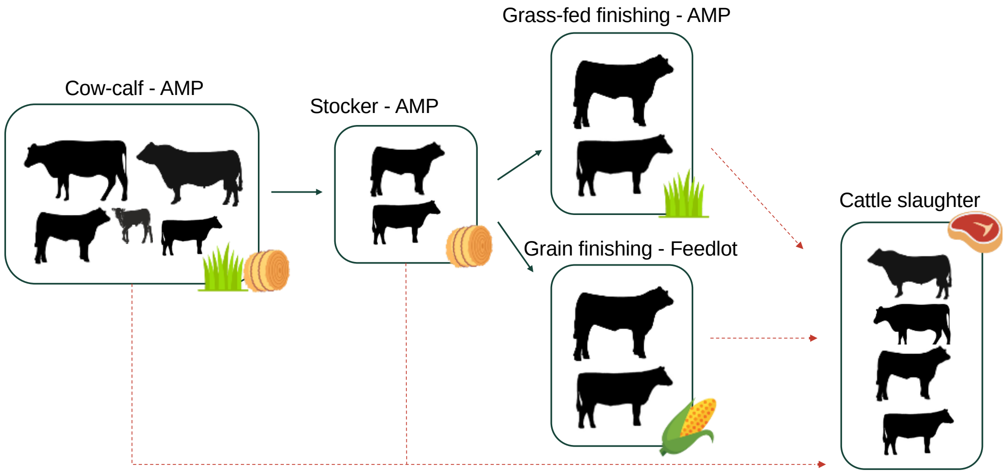

Figure A.1. Beef production systems with corresponding phases included in the LCA.

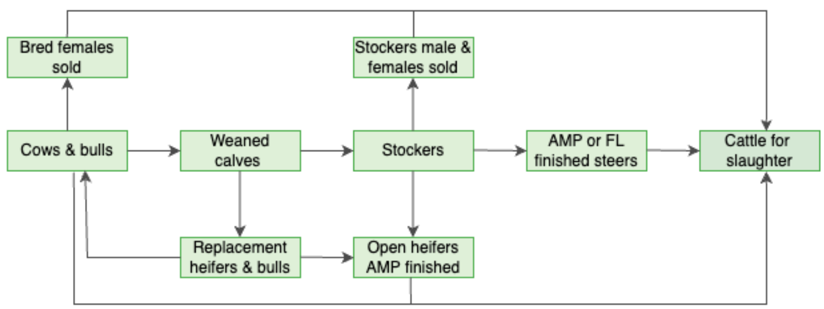

Figure A.2. Simplified process flow for the breeding herd in relation to the production cycle for AMP-only and AMP+FL systems.

Table A.3. Simplified classification of the annual average animal population at Lake City AgBioResearch Center (2012-2023) used as input in the LCA model.

Production phase	Animal Category	No. of animals
Cow-calf	Cows	139
	Calves	127
	Bulls	5
	Repl. heifers 6-12mo	57
	Repl. heifers 12- 19/24mo	55
	Repl. bulls 6-12mo	13
	Repl. bulls 12-24mo	7
Stocker	Post-weaned calves sold	14
	Stocker steers 6-12mo	43
Finishing	AMP steers (AMP-only)	47
	FL steers (AMP+FL)	47

Table A.4. Beef processing conversion used for AMP-only and AMP+FL cattle.

Animal category	No. of animals	LW at slaughter (kg)	Dressing %
Cull cows	17	539-562	50%
Bred cows/heifers sold[1]	20	437-562	50%
Cull bulls	3	678-726	50%
Post-weaned female and castrated male calves sold[1]	13	233-251	50%
Heifers and steers sold at ~12 mo. old	4	296-344	50%
Open heifers and unproductive 1st-calf heifers	17	457-465	55%
AMP-finished steers	46	517	54%
FL-finished steers	46	654	61.3%

[1] Note that bred females sold and post-weaned calves sold are included in the total annual beef output by assuming a dressing percentage of 50% based on Rotz et al. (2019).

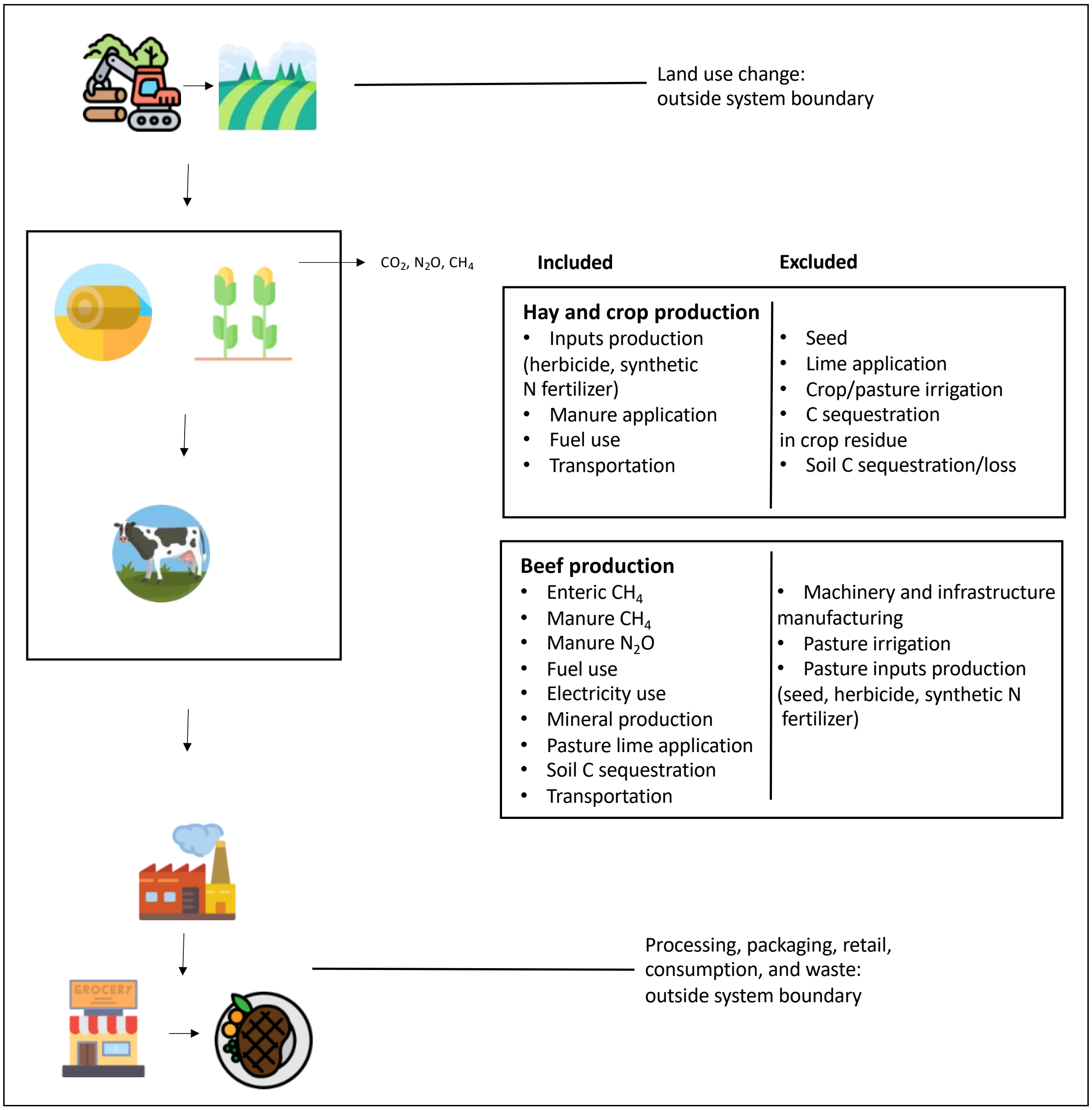

Figure A.3. System boundary used in our cradle-to-farm gate LCA.

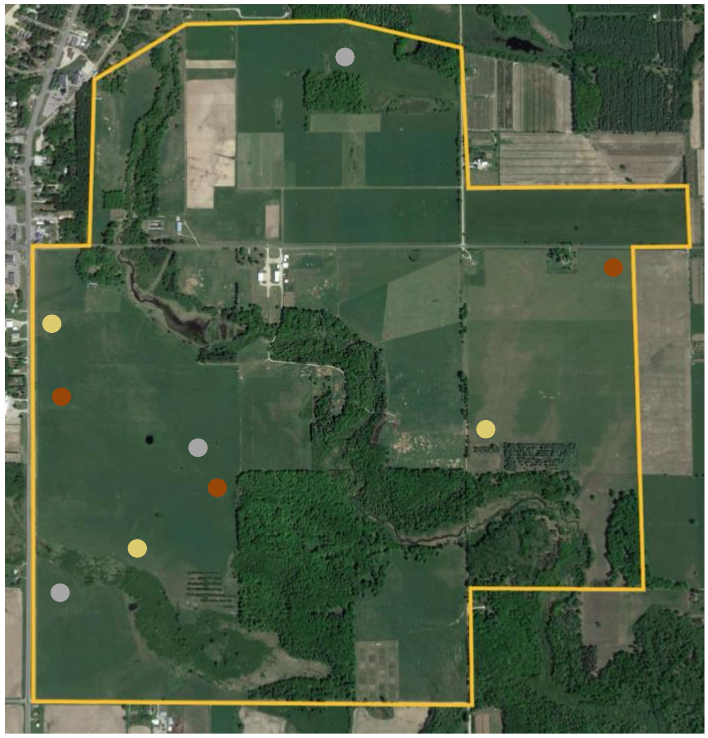

Figure A.4. Georeferenced sampling sites at Lake City AgBioResearch Center, categorized by soil type.
These sites correspond to the farm's three main soil types within AMP-managed pastures: sandy loam (red), clay loam (gray), and sand (yellow).

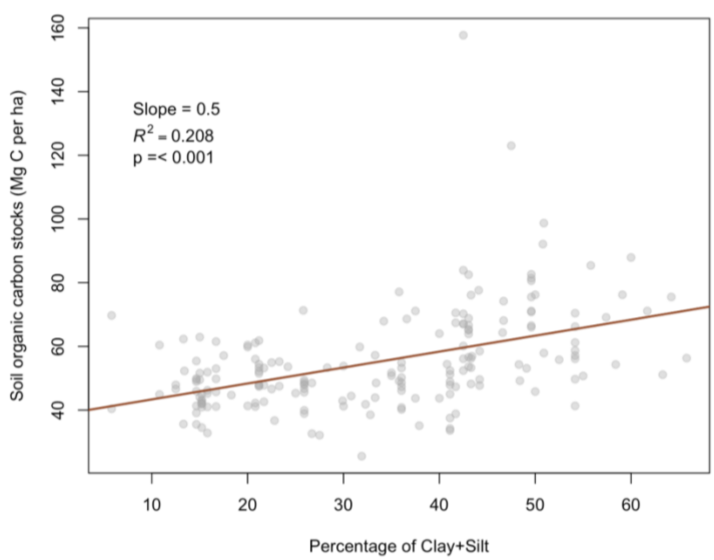

Figure A.5. Linear regression analysis of soil C stock and clay+silt content.
The relationship between soil C stock measurements taken at Lake City AgBioResearch Center and clay+silt content shows a weak correlation (R^2=0.208), but it is statistically significant (p=<0.001).

Section A.1: Enteric CH_4 sensitivity analysis

Table A.5. Enteric CH_4 sensitivity analysis. Comparison of IPCC Tier 2 estimates with measured data and Simplified IPCC Tier 2 approaches.

Beef system	Methodology	Kg CH_4 production system^{-1}	% difference with IPCC Tier 2	% difference with IPCC Tier 2 by animal category
AMP-only	IPCC Tier 2[1]	24,103	-	-
	On-site measurements[2]	23,364	-3%	~17% for lactating cows grazing; ~24% for finishers 12-18 mo. old
	IPCC Tier 2 Simplified[3]	26,773	+10%	-
AMP+FL	IPCC Tier 2[1]	22,762	-	-
	On-site measurements[2]	22,691	-0.3%	-
	IPCC Tier 2 Simplified	25,075	+9%	-

[1] Emission factors for each animal category were developed based on DMI calculated according to the animals' gross energy requirements (IPCC, 2019) and a Midwest Y_m of 6.5 for grassfed cattle and 3.9 for FL cattle, as developed by the EPA (2023).

[2] Measured enteric CH_4 data for grazing lactating cows were sourced from Chiavegato et al. (2015a) using the SF_6 tracer gas method, and data for yearling heifers and steers were obtained from Thompson et al. (2021) using the GreenFeed system at LCRC. Additionally, measured enteric CH_4 data for the FL stage were derived from Roque et al. (2021) using the GreenFeed system.

On-site measurements calculations

To calculate enteric CH_4 emissions for each system, we used data collected at LCRC to extrapolate emissions for our AMP categories, while emissions for the FL stage were derived from Roque et al. (2021). Data from LCRC included measurements from specific categories: grazing lactating cows assessed using SF6 tracer gas, as reported by Chiavegato et al. (2015a) in 2012 and 2013, and grazing yearling heifers and steers evaluated with the GreenFeed system, as detailed by Thompson et al. (2021) during 2018-2020. For the FL stage, we relied on data collected from FL steers using the GreenFeed system (Roque et al., 2021).

Since the measurements from these three specific categories do not cover all 59 or 57 categories within our AMP-only and AMP+FL models, respectively, we extrapolated emissions based on the average body weights (BW) and CH_4 emissions of the measured animals. The average BW of these animals was adjusted to metabolic body weight (MBW) by elevating it to the power of 0.75. To derive emissions for the unmeasured categories, we calculated the metabolic body weight ratio (MBWR) by dividing the MBW of each unmeasured category by that of the measured category. This ratio, which reflects differences in average weights among categories, ranged from 0.6 to 1.4. The resulting MBWR for each category within the AMP-only and AMP+FL systems was then multiplied by the measured daily enteric CH_4 emissions to estimate emissions for the unmeasured categories. For grassfed categories under 2 years old, we used extrapolations derived from Thompson et al. (2021), while for those over 2 years, we referenced Chiavegato et al. (2015a), as they generally had a MBWR closer to 1. The formulas used in this calculation are summarized in Table 1 below.

Table A.6. Calculations to extrapolate measured enteric CH_4 emissions to other beef categories.

Formula	Description
$MBW = BW^{0.75}$	Calculates metabolic body weight.
$MBWR = \frac{MBW\ unmeasured}{MBW\ measured}$	Calculates the metabolic body weight ratio (MBWR) for unmeasured categories.
$Emissions_{unmeasured} = Emissions_{measured}\ x\ MBWR$	Estimates enteric CH4 emissions for unmeasured categories using the MBWR.

IPCC Tier 2 Simplified

The simplified Tier 2 is an alternative method to estimate enteric CH4 emissions factors by predicting dry matter intake (**DMI**) based on animal physiological condition, live weight, and diet quality, and then calculating emissions by multiplying predicted DMI with IPCC's default CH4 yield values expressed in g CH_4/kg DMI.

CHAPTER 3

CONCLUSIONS, IMPLICATIONS AND FUTURE RESEARCH

This research provides new insights into the nexus of soil C, grazing management, and GHG emissions in beef cattle production at a farm level. The findings indicate that AMP grazing applied in pasture-based stages of the beef supply chain can sequester soil C and fully compensate the GHG emissions beef cattle generate for long periods of time (over 10 years) in cow-calf to grain or grass-finished beef systems in the Upper Midwest, suggesting that grazing management can be a potentially effective tool for GHG mitigation.

The common assumption that grazing practices to sequester soil C require more land needs questioning. The study's outcomes highlight the potential significant GHG benefits of adopting AMP grazing in current grazing-based stages–which contribute 70 to 80% of the U.S. beef industry's C footprint–without necessitating a transformation of the industry's structure.

Although our results show that the GHG mitigation is even more pronounced if the finishing phase is grass-finished and AMP-managed instead of being finished in a feedlot, changing the finishing strategy would require significantly more land, which is undesirable from a GHG perspective if it involves land clearing. Nonetheless, AMP-finishing subsystems may also serve as a supplementary soil C sequestration strategy if implemented in croplands aiming to enhance soil health and farm resilience by incorporating livestock grazing into their rotations. Combining several practices and technologies to reduce beef GHG emissions may be more beneficial than relying on a single solution in such a decentralized beef supply chain. Feedlots, for example, can provide additional GHG reductions through cropping annual management, beef productivity gains, and potential feed additives interventions that are more difficult to apply in grazing settings.

Despite being a case study in the Upper Midwest region, and therefore caution is needed when extrapolating conclusions, we were fortunate to have one of the few longer-term soil C data sets, including baseline soil C stocks and AMP grazing management, to address the gap in long-term GHG mitigation with AMP grazing. However, considerably more work is needed to determine the global GHG mitigation potential of AMP grazing in beef production. Different ecoregions will require AMP grazing and key grazing levers–frequency, timing, duration, and intensity–to be applied differently, likely resulting in different optimal stocking rates and achievable soil C sequestration rates based on management, as well as soil, climate, land use history and other considerations that affect soil C change. Prioritizing countries and regions where beef production and its environmental trade-offs are prevalent is key to determining effective mitigation strategies. Additionally, global (as opposed to per hectare) assessments of the soil C sequestration potential with AMP grazing will require significant developments in the modeling capacity of soil C in grazinglands.

Another area of research on AMP grazing and GHG mitigation is mitigation through productivity gains, such as increased weight daily gains in stocker and finishing cattle to reduce time to market, or improved reproductive performance of the cowherd. While significant productivity gains may be challenging in an already highly efficient beef industry like in the U.S., evaluating AMP grazing as a global GHG solution should consider potential efficiency improvements in developing countries, where consistency and quality of forage are often not guaranteed, thus potentially yielding substantial GHG reductions with improved grazing management.

Holistic GHG mitigation targets, which integrate environmental, economic, and social considerations, should be developed to assess the effectiveness of mitigation practices in beef production. While continuous improvement is key, policy and management interventions should balance productivity, sustainability, and societal values, customized to the specific needs of each region and nation. Otherwise, relying solely on dichotomies, such as being a net sink or emitter, can lead to unintended consequences.

While ongoing research advances and validates our findings, based on our data, we recommend that beef grazing managers in the Upper Midwest and similar ecoregions worldwide consider adopting AMP practices to reduce fossil-derived input use while maintaining productivity and potentially increasing soil C levels. Importantly, to catalyze the adoption of these emergent agricultural practices and translate scientific findings into practice, policy must play a significant role. Legislation should promote the adoption of these practices, which can preserve current grassland soil C stocks, produce food, support farmers' livelihoods, and improve farm resilience.

Overall, this study underscores the critical importance of evaluating the potential of AMP grazing to sequester soil C over time, suggesting it may be more beneficial in offsetting GHG emissions from beef cattle production than previously thought. Specifically, our findings indicate that AMP grazing offers a long-term, sustainable, and viable solution for pairing food production with significant soil C sequestration and GHG mitigation in current beef grazing systems in the Upper Midwest.