Sustainability of Regional Beef Production Systems

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Abstract

Background: The large-scale ecological burdens of beef production are coupled with increasing global demand for beef. As the top producer and a leading global consumer of beef, the U.S. should lead the development of innovative strategies to reduce the pollution and resource use of this system. While much attention has been devoted to increasing production efficiency or reducing consumption, analyses of transformative approaches that view structural change as requisite for sustainability have been limited. Increasing reliance on local or regional foods is such an approach, and consumer demand for these foods is high. Accordingly, the objective of this research is to explore whether regional beef production systems provide opportunities to enhance sustainability and the food supply in the Northeastern United States.

Methods: This project used a mixed-methods approach that included life cycle assessment and geospatial analysis with primary and secondary data. Objective 1 was an attributional, environmental life cycle assessment (LCA) of regional grass-fed and dairy beef production systems. Life cycle inventories were initially developed with secondary data and expert opinion, and then further calibrated with producer interviews (n=12). Objective 2 used geospatial analysis to enhance a method that estimates the land use efficiency of livestock systems from a human food supply perspective, the land use ratio (LUR). Land use data was collected during farm interviews (Objective 1) and combined with spatial data on land cover and potential crop productivity for grass-fed and dairy beef case studies. Objective 3 used consequential life cycle assessment to assess whether substituting food waste for corn in regional dairy beef cattle rations could reduce environmental burdens and the LUR. The induced effects of shifting the application of food waste from anaerobic digestion to feed were included in the system boundary.

Results: Per kg beef produced, Northeast dairy beef had lower global warming potential, eutrophication potential, acidification potential, and agricultural land use than grass-fed with higher fossil depletion and similar water depletion. However, per ha agricultural land, eutrophication and acidification of grass-fed were lower than dairy beef. Both grass-fed and dairy beef had LUR greater than one, indicating that converting suitable feed land to food crops could produce more human digestible protein. The LUR of grass-fed beef was 3-6 times larger (less efficient) than dairy beef. Finally, substituting food waste for corn in dairy beef cattle rations, instead of using it as a feedstock for anaerobic digestion, reduced global warming potential, acidification potential, and feed-food competition as measured by the LUR.

Implications: Innovations in dairy, beef, and waste management systems can be strategies to move toward sustainability in the region. A pilot program to develop regional waste-fed dairy beef should be prioritized. Furthermore, states and municipalities should develop policies and support structures that encourage waste-to-feed. Finally, although accounting for ecosystem services provided by pasture-based farming systems was not possible in this project, they should not be ignored. Standardized methods are needed to account for ecosystem services in LCA for more holistic assessments of environmental and social sustainability in the future.

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Chapter 1: Introduction

Background and Significance

The ecological burdens of the global livestock sector are large, with the production of cattle as a major contributor (Herrero et al., 2015; Opio et al., 2013). Livestock are responsible for 8-18% of global anthropogenic greenhouse gas (GHG) emissions, 64-78% of which are attributed to cattle (Herrero et al., 2016; Herrero and Thornton, 2013). Over one quarter of all water used for agriculture is dedicated to feed and maintain livestock (Herrero et al., 2015). Within that figure, over half (58%) of freshwater withdrawn (i.e., blue water) for livestock is used for ruminant meat and dairy production (Herrero et al., 2015). While livestock in general is often cited as the primary user of global agricultural land (Foley et al., 2011), ruminants are the driving force behind this statistic. More than 70% of global agricultural land is used for ruminant meat and dairy production (Foley et al., 2011; Herrero et al., 2015). This figure includes permanent grassland and a substantial fraction of cropland; 44% of global cropland is used for livestock feeds ruminants (Herrero et al., 2015). Per unit output, beef emerges as the largest land user among ruminant and all livestock production systems (de Vries and de Boer, 2010; Peters et al., 2014).

Given the footprint of the livestock sector, substantial attention has been directed to strategies to improve sustainability. The dominant paradigm in this space has been sustainable intensification: reducing impacts per unit output (Garnett, 2014; Herrero et al., 2015). This approach can be categorized as efficiency-oriented (Garnett, 2014). Research and proposals that support increasing efficiency, enhancing productivity, or reducing waste have proliferated in recent years (Capper and Bauman, 2013; Havlík et al., 2014). One common strategy to improve efficiency of cattle production is grain feeding, shifting from low-input extensive systems to mixed or zero-grazing systems (Havlík et al., 2014). Indeed, the U.S. beef industry has decreased its carbon footprint per animal since the 1970s through the use of high-grain finishing rations, in addition to pharmaceuticals (e.g., antimicrobials and growth hormone implants) and improved genetics (Capper and Bauman, 2013). However, feeding cereals and oilseeds to livestock instead of humans is an inefficient use of calories (West et al., 2014), particularly for cattle whose reproductive rates are low and maintenance requirements high (Herrero et al., 2015).

While the current situation is problem enough, global consumption of beef to 2050 is projected to grow 1.2% per year (Alexandatos and Bruinsma, 2012). Curtailing consumption of livestock products, particularly in developed countries, has emerged as another broad approach to move toward sustainability (Garnett, 2014). Although reducing consumption of livestock products in developed countries is necessary, it is an insufficient solution to reduce the sector's footprint to sustainable levels (Garnett, 2011, 2009). Innovative strategies are needed to improve the sustainability of beef production and consumption systems.

The U.S. beef system

As the top global producer and a leading source of demand for beef, the United States is a dominant player in this market (OECD, 2016; USDA-ERS, 2015a). As such, beef cattle account for over half (55%) of the GHG emissions of the U.S. livestock sector. This burden is mostly produced to meet U.S consumer demand; over 90% of U.S. beef is consumed domestically (USDA-ERS, 2015b). At 24.7 kg yr^{-1}, per capita beef consumption in the U.S. is the fourth highest globally (OECD, 2016). Although this amounts to only 4% of the food supply by mass, beef disproportionately influences the carbon footprint of the average U.S. diet (Heller and Keoleian, 2015). Due to its emissions intensity, beef is responsible for over one-third of the GHG emissions of the U.S. food supply (Heller and Keoleian, 2015).

In response to growing awareness of this burden, the U.S. beef industry launched a multi-phase life cycle assessment to establish a benchmark, document progress over a six-year period, and identify areas for continuous improvement (NCBA, 2014). The fundamental issue with this and similar country-level and global initiatives is that the definition of sustainability (either explicit or implicit) is to sustain the beef industry (NCBA, 2014; "What Is Sustainable Beef?," 2016). This could be described as *internal sustainability*, defined by industry stakeholders with the goal of perpetuating the industry. Environmental stewardship undoubtedly concerns many industry stakeholders. However, whether current levels of beef production and consumption are sustainable is unquestioned, and understandably so; livelihoods are at stake. The strategy to move the industry toward sustainability is to identify and target supply chain hotspots to reduce impacts (NCBA, 2014). While process improvement is necessary, marginal changes within existing supply chains that are already characterized by a high degree of ecological efficiency fail to address whether the beef system itself is sustainable.

Indeed, the structure of the beef system itself may be an underlying cause of environmental degradation and source of future vulnerability (Meadows and Wright, 2008). For example, industry concentration, consolidation and specialization may disrupt feedback loops that could enhance sustainability (Meadows and Wright, 2008; Sundkvist et al., 2005). Over 80% of beef packing in the U.S. is controlled by four firms, whose facilities and immediate supply chains are spatially concentrated in the Great Plains (Ward, 2010). On farms, concentrations of feedlot animals by county have increased over time since the early 1980s (Kellogg et al., 2000). Increasing specialization and concentration at the farm level calls into question the ability of the surrounding environment to assimilate excess nutrients from manure. Between 1950 and 2000, despite a steady decrease in N and P excretion per unit meat, increased production and low nutrient recovery in crop and livestock systems rapidly concentrated nutrients spatially in excess of soil budgets (Bouwman et al., 2013). Kellogg and colleagues (2000) estimated that 160 counties in the U.S. had excess manure phosphorus and 73 counties had excess manure nitrogen in 1997. Within these counties, 259 feedlot operations accounted for about 46 million pounds of excess manure phosphorus.

Economics have been and will continue to drive structural transitions in cattle and cropping systems. Over the past half century, increases in productivity and efficiency led to increased profitability of livestock farms and ranches, providing an incentive to increase inventories (Herrero and Thornton, 2013). At the same time, high risk and low reward financial environments have fueled consolidation in the sector and resultant corporate ownership of feedlots, a trend likely to continue (Galyean et al., 2011). A ten year Iowa State University study found negative returns in most years for finishing calves and yearlings (Galyean et al., 2011). This environment is coupled with declining beef consumption in the U.S. Thus, beef trade organizations are pursuing intense marketing strategies to increase domestic consumption (Schroeder et al., 2000) and trade agreements to increase exports. Given these trends and global meat demand dynamics, projections indicate U.S. cattle inventories will increase by 2050 (Bouwman et al., 2013). Absent increased environmental regulation and oversight of crop and livestock farms, it is reasonable to hypothesize that increasing beef production within the current system structure may further degrade local and regional ecosystems.

Sustainability measurement and transformative approaches

Life cycle assessment (LCA) is a generally recognized method to assess the environmental and social impacts of products, systems, or choices. There are two general LCA methods: attributional and consequential. Attributional LCA estimates a life cycle inventory by "attributing" to specific products a portion of the resources into, and pollution from, multi-product systems. Consequential LCA seeks to model the consequences or system-wide changes induced by a decision or direct change. Due to its structure, LCA is a tool particularly amenable to the efficiency perspective, and has strengthened that perspective's dominance in conversations regarding food system sustainability (Garnett, 2014). LCA has had much more limited application measuring the impact of transformative approaches, which are underpinned by the notion that structural change is necessary for environmental sustainability (Garnett, 2014).

Holistic analyses of beef production systems that use life cycle thinking and assessment can help shape the path forward. Life cycle assessments of beef systems are often done at the regional scale to account for varied climate, terrain, and management practices across the country (Lupo et al., 2013). Applying LCA at the regional scale, however, may also be used as a lens to inform the development of food systems that are more geographically proximal to consumers (i.e., local or regional). Developing these systems may improve resilience and sustainability by closing feedback loops (Sundkvist et al., 2005). This is particularly applicable to eutrophying and acidifying emissions from agriculture, the effects of which are experienced locally and regionally. LCA can be used to assess eutrophication and acidification potential of beef systems per unit land to facilitate decision making regarding risks to regional environmental quality.

U.S. consumer demand for geographically identified foods is high, particularly in the Northeast (Low and Vogel, 2011; Martinez et al., 2010). Domestic consumers value regional or source-identified beef and are willing to pay premium prices for it (Abidoye et al., 2011; Umberger et al., 2009). In the Northeast, several sub-regional initiatives to increase production and improve market access also suggest strong interest in regional beef and dairy systems (Donahue et al., 2014; Vermont Sustainable Jobs Fund, 2013). Given this outlook, combining knowledge of the landscapes, industries, and policies in the Northeast with a life cycle perspective provides an opportunity to explore creative sustainability strategies.

Multifunctional systems that produce both milk and meat demonstrate substantial potential to reduce environmental burdens of beef from a life cycle perspective (de Vries et al., 2015). The Northeast region is a dairy production center, which may provide ecological leverage for more sustainable beef production. The Northeast is 76% regionally self-reliant (RSR) in dairy products, despite its high population density and declining agricultural land base (Griffin et al., 2014). LCAs of dairy beef production have been conducted in other U.S. regions with varying coverage in impact categories (Rotz et al., 2015; Stackhouse-Lawson et al., 2012).

While not a major beef producing region [RSR of beef =16%; (Griffin et al., 2014)], grass-fed beef is growing in popularity in the Northeast. LCA research on the environmental benefits of grass-fed beef, however, is mixed. When the potential soil carbon sequestration of well-managed pasture is considered, some grass-finishing operations may outperform intensive systems and even reduce carbon emissions (Allard et al., 2007; Liebig et al., 2010; Lupo et al., 2013; Pelletier et al., 2010). While comparisons of eutrophication potential between conventional and grass-fed beef systems are mixed (Lupo et al., 2013; Pelletier et al., 2010), Lupo et al. (2013) found that both eutrophication and acidification potential were lower in a grass-finished system than in both cow-calf/feedlot and cow-calf/backgrounder/feedlot systems in the Northern Great Plains. However, the grass-fed systems modeled previously were finishing strategies for calves from conventional cow-calf herds, rather than as distinct production systems (Lupo et al., 2013; Pelletier et al., 2010). The impact of whole herds managed with management intensive grazing (MiG), as is common in the Northeast (Steinberg and Comerford, 2009), maintained entirely on forage year-round, and bred with grass-fed genetics is currently unknown.

Grass-fed beef is also highly if not entirely reliant on forages, converting human inedible energy and protein to edible products. For a grass-fed system in the Upper Midwest, Pelletier et al. (2010) found a 69.1% human-edible energy return on investment, over an order of magnitude greater than a feedlot-based system. While this provides a more nuanced perspective of feed efficiency, it fails to address whether the underlying forage base could be used instead to produce human food directly. Given the large fraction of agricultural land used by ruminants, this tradeoff ought to be addressed.

Van Zanten et al. (2015) developed a novel metric to quantify the opportunity cost of land used to feed livestock instead of growing human food crops directly. This land use ratio (LUR) leverages data collected for LCAs, providing an assessment for how far from an "ecological leftovers" approach to livestock rearing current systems may be. The ecological leftovers approach has as a premise that livestock should be reared only to the extent that they add value to inedible byproducts, residues, and marginal lands with low opportunity costs to meet other human needs (Garnett, 2014). For example, beef production systems that graze on marginal land may result in a net positive contribution to the food supply (de Vries et al., 2015; Eisler et al., 2014; van Zanten et al., 2015). However, only about half of global pasture land is marginal (van Zanten et al., 2016). Whether the production of beef results in feed-food competition or a net positive contribution to the food supply may depend largely on whether marginal land is used for forage production. The land use ratios of Northeast beef production systems are unknown.

Interest in local and regional meat production coupled with high population density may provide a significant opportunity for a sustainability and food security win-win in the Northeast: repurposing food waste as livestock feed. This strategy could increase beef production system's reliance on "ecological leftovers." Several New England states and New York City have recently adopted organic waste landfill bans (Edwards et al., 2015). At the same time, landfill bans and energy policy (e.g., The Clean Power Plan) encourage anaerobic digestion of organic wastes (Edwards et al., 2015), increasing competition for these materials. The U.S. EPA's Food Recovery Hierarchy prioritizes diverting food waste to animal feed over energy production, indicating its superior benefits to ecosystems and society (U.S. EPA, 2016). To determine the optimal use of food byproducts, their alternate use and displaced products should be considered in a consequential LCA framework (van Zanten et al., 2014).

Research Objectives

The large-scale ecological burdens of beef production are coupled with increased global demand for beef. Innovative strategies are needed to increase the sustainability of beef production and consumption. **To this end, this dissertation focuses on the following research question: can regional beef production systems provide opportunities to enhance sustainability and the food supply?** Working in the Northeast region of the U.S., I address this central research question by pursuing the following objectives:

- Objective 1 (Chapter 2): Compare the environmental performance of regional grass-fed and dairy beef production systems
    - *Sub-objective 1.1*: Develop a life cycle inventory for a management-intensive grazing (MiG) grass-fed beef production system
    - *Sub-objective 1.2*: Develop a life cycle inventory for a confinement dairy beef production system
    - *Sub-objective 1.3*: Identify production system hotspots to target for process improvement
    - *Sub-objective 1.4:* Compare eutrophication and acidification burdens on mass and area bases

- Objective 2 (Chapter 3): Calculate an enhanced land use ratio (LUR) for regional grass-fed and dairy beef production systems
    - *Sub-objective 2.1*: Incorporate geospatial data and analyses at multiple scales into the LUR calculation

  - *Sub-objective 2.2:* Assess the influence of limiting perennial land conversion on systems' LUR

- Objective 3 (Chapter 4): Assess the environmental and food supply impacts of feeding food waste to regional dairy beef cattle instead of using it as a feedstock for anaerobic digestion
  - *Sub-objective 3.1*: Construct a U.S. specific anaerobic digestion model for food waste
  - *Sub-objective 3.2:* Construct a dairy beef production system model that substitutes food waste for feed corn

- Objective 4 (Chapter 5): Translate research findings to inform the public and influence policy

Chapter 2: Life cycle environmental consequences of grass-fed and dairy beef production systems in the Northeastern U.S.

Written in the style of *The Journal of Cleaner Production*

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Abstract

Innovative strategies are needed to improve the sustainability of beef production and consumption systems. Increasing reliance on regional or local food systems may improve resilience, and consumer demand for such foods is high. In the Northeastern U.S., the dairy sector may provide beef at a low environmental cost relative to other systems due to multi-functionality (i.e., milk and meat outputs). Additionally, landscape and market factors indicate suitability and demand for regional grass-fed beef. We used ISO-compliant life cycle assessment (LCA) to quantify the environmental burdens of grass-fed beef with management-intensive grazing (GF) and confinement dairy beef (DB) production systems in the Northeastern U.S. The impact scope included global warming potential, eutrophication and acidification potential, fossil and water depletion, and agricultural land use. The foundation of the production system models was a herd-level, life cycle livestock feed requirements model by Peters et al. (2014), which we adapted and applied for the first time within LCA. Per kg carcass weight beef produced, DB had lower global warming potential, eutrophication potential, acidification potential, and agricultural land use than GF with higher fossil depletion and similar water depletion. Calculating eutrophication and acidification per ha agricultural land resulted in lower impacts for GF compared to DB. Maintaining the breeding herd accounted for over half of GF (60%) and DB (55%) impacts on average across categories. Sensitivity analyses indicated potential pasture carbon sequestration and lower enteric methane emissions under management-intensive grazing may substantially reduce the carbon footprint of GF (though not lower than DB), which should be explored with further research. Future research should also examine holistic strategies to reduce regional GF and DB system footprints, such as substituting food waste for traditional feeds and accounting for ecosystem services provided by pasture-based farming systems within LCA.

Keywords: beef production, dairy production, life cycle assessment, regional food systems, greenhouse gas emissions, land use

1. Introduction

The global environmental burdens of ruminant production systems, particularly of cattle, are large and well documented (Herrero et al., 2015; Opio et al., 2013). The United States is the leading global producer of beef, slaughtering 11.8 million kg of carcass-weight beef annually (USDA-ERS, 2015a, 2015b). Accordingly, beef cattle account for over half (55%) of the greenhouse gas emissions from livestock in the U.S. (USDA-OCE, 2011). This burden is primarily associated with U.S consumer demand, since over 90% of U.S. beef is consumed domestically (USDA-ERS, 2015b). Beef is a significant contributor to environmental impact of the average U.S. diet. It accounts for over one-third of the U.S. dietary carbon emissions, despite being only 4% of food supply on a mass basis (Heller and Keoleian, 2015). Strategies are urgently needed to reduce beef production and consumption impacts.

Life cycle assessment (LCA) is a generally recognized method to compare the environmental and social performance of products or systems. Beef production systems in several U.S. regions, including the Great Plains, Upper Midwest, and California, have been evaluated using LCA (Lupo et al., 2013; Pelletier et al., 2010; Rotz et al., 2015; Stackhouse-Lawson et al., 2012). Regional analysis of these systems is warranted due to varied climate, terrain, and management practices across the country (Lupo et al., 2013). Applying LCA at the regional scale, however, may also be used as a lens to inform the development of food systems that are more spatially proximal to consumers (i.e., local or regional).

Developing more geographically proximal food systems may improve resilience and sustainability (Sundkvist et al., 2005). Domestic consumer demand for geographically identified foods is high, particularly in the Northeast region (Low and Vogel, 2011; Martinez et al., 2010). U.S. consumers are willing to pay a premium price for regional or source-identified beef (Abidoye et al., 2011; Umberger et al., 2009). In the Northeast, a growing number of state-level and sub-regional initiatives to increase production and improve market access suggest strong interest in regional beef systems from planning and policy perspectives (Donahue et al., 2014; Vermont Sustainable Jobs Fund, 2013)

Although beef is one of the most carbon and resource-intensive animal source foods, there is significant heterogeneity in the impacts of beef production systems (de Vries et al., 2015). Multifunctional systems that produce both milk and meat demonstrate substantial potential to reduce environmental burdens of beef from a life cycle perspective (de Vries et al., 2015). This multi-functionality may provide an ecological leverage point for producing beef in the Northeastern U.S., a dairy production center. The Northeast is 76% regionally self-reliant (RSR) in dairy products, despite its high population density (Griffin et al., 2014). In the U.S., dairy beef production has been subject to LCA in California and the Southern Great Plains, with varying coverage in impact categories (Rotz et al., 2015; Stackhouse-Lawson et al., 2012).

While not a major beef producing region (RSR of beef =16%; (Griffin et al., 2014)), landscape and market factors in the Northeast suggest current and future potential for grass-fed beef. A large proportion of agricultural land in the Northeast is used for pasture and roughage production (45%) (Conrad et al., 2016; Griffin et al., 2014). In Pennsylvania and West Virginia, states in the Northeast region, Evans et al. (2011) found consumers were willing to pay a premium for grass-fed, regional beef. For producers, a growing number of grazing-focused meetings in the region suggest interest in these production systems. Life cycle assessments of grass-fed beef have been performed in other U.S. regions (Lupo et al., 2013; Pelletier et al., 2010). However, these systems were modeled as finishing strategies for calves from conventional cow-calf herds, rather than as distinct production systems. Modeling whole herds managed with management intensive grazing (MiG), as is common in the Northeast (Steinberg and Comerford, 2009), maintained entirely on forage year-round, and bred for grass-based production, has not been yet been accomplished, to the best of our knowledge.

The environmental consequences of grass-fed and dairy beef systems in the Northeast region are unknown, but important given producer, consumer, and policy/planning trends. We conduct an ISO-compliant (International Standards Organization series 14040) LCA of MiG grass-fed beef (GF) and confinement dairy beef (DB) production systems in the Northeastern U.S[1] to identify hotspots and to inform regional food systems planning and policy.

2. Materials and Methods

The functional unit for this analysis was 1 kg hot carcass weight (HCW), due to differences in finished weights and to facilitate comparison with the literature. Dressing percentages for market DB, market GF, and all culled cattle (cows and bulls) were 59, 54, and 50% of final shrunk bodyweight, respectively, accounting for decreased carcass conversion of grass-fed (Duckett et al., 2013; Neel et al., 2007; Scaglia et al., 2012) and

[1] The Northeast region includes the following states: Connecticut, Delaware, Maine, Maryland, Massachusetts, New Hampshire, New Jersey, New York, Pennsylvania, Rhode Island, Vermont, and West Virginia (USDA-NIFA, 2012).

culled cattle (Stackhouse-Lawson et al., 2012). The system boundary extended from cradle to farm-gate, including feed production and processing, feed and water provisioning, manure management, cattle transport and facility operations. Although using HCW as a functional unit at the farm gate ignores both co-products from and emissions of processing, modeling this stage was beyond the scope of this study. Other North American beef LCAs have excluded the production and maintenance of capital goods (Beauchemin et al., 2010; Lupo et al., 2013; Pelletier et al., 2010). To compare with the literature, we excluded capital goods when modeling the foreground system. The Ecoinvent v. 3.1 data we used to model upstream supply chains, however, included capital goods (*Ecoinvent database*, 2013).

2.1. Data availability and approach

The foundation for the production system models was a U.S. livestock model developed by Peters et al. (2014) (hereafter, the LM), which calculates direct and indirect herd-level feed requirements, land use and food output for average systems. We calibrated the LM to represent Northeast regional systems and extended it to calculate resource use and emissions, applying it for the first time within an LCA. We followed a two-stage process to define the beef production systems. First, we parameterized models to the extent possible using published literature, special tabulations of two regionally-representative dairy surveys (Thoma et al., 2010; USDA-APHIS, 2007) and expert opinion. Second, we addressed data gaps and verified assumptions by interviewing regional grass-fed (n=9) and dairy beef (n=3) producers. Despite attempts to interview dairy beef producers in the Northeast, our final respondents were from Ohio, on the western border of the region. Due to the diverse nature of our data, all assumptions and their sources are fully summarized in the Appendices (Tables A1 and A2) when not stated directly in the text.

2.2. System descriptions

The GF calving phase is a herd of 30 breeding cows, plus associated breeding bulls and replacement stock (Table 2.1). The predominant breed is a smaller-frame variant of Angus. A yearling bull is imported to the system every 2.5 years, traveling 320 km by truck. Spring-born calves are weaned after 207 days (Peters et al., 2014). During the grazing season (200 d yr^{-1}), cattle are moved between small paddocks (0.4 – 6 times d^{-1}) divided by electric fence, using an all-terrain vehicle. During the winter, the breeding herd bale grazes on pasture, while backgrounding calves and finishing cattle are fed in a barn with access to an outdoor holding area. Manure and bedding from the barn are scraped, stored in piles, and applied to pastures during the growing season. Electricity is used year-round to pump ground water to meet cattle requirements. Mature steers and heifers are sent to market at 712 and 682 days of age, respectively. Mature cows and bulls are also culled to produce beef per breeding cycle (cull rates of 9.7 and 20.0%, respectively) (Peters et al., 2014).

The DB calving phase is a 328 cow dairy herd, plus replacement stock and one breeding bull (Table 2.2) (Thoma et al., 2010). The predominant breed is Holstein with a rolling herd average milk production of 10,732 kg $head^{-1}$ yr^{-1} (Thoma et al., 2010; USDA-APHIS, 2007). On-farm energy demand includes all electricity and other fuel, excluding household energy use, as estimated by Thoma et al. (2013b). Manure is managed as a slurry and stored in an earthen pond/tank system. In addition to meeting drinking water requirements, 25 liters cow^{-1} day^{-1} of water are used to wash milking facilities, the mean of a range reported for the U.S. (Safferman, 2008). Surplus calves are sold either for veal (47%) or dairy beef (53%) production per breeding cycle. Additionally, 20% of mature bulls and 28.2% of cows are culled for beef annually (Conrad et al., 2016; Peters et al., 2014). We allocated burdens between milk (90.2%) and meat (9.8%) at the dairy gate using a biophysical allocation equation developed for the U.S. (Thoma et al., 2013a). We then subdivided the meat burden on a mass basis between dairy beef (culls and calves) and veal to exclude veal from the system boundary (Table 2.3).

Newborn dairy beef calves are shipped 400 km by truck to starter operations, where they are fed in unheated, naturally ventilated barns. Water to hydrate milk replacer is pumped, heated, and mixed with milk powder that is shipped 450 km from a manufacturer. Bedded manure is scraped from the barn floor and lime is applied between each batch of calves. Weaned calves (181 kg) are shipped 400 km to feedlots for growing and finishing until 475 days of age. The animals are housed in a barn and open lot system, where manure is scraped and stored as a solid. Feeds are ground, mixed, and hauled to pens twice daily, and water is pumped to meet cattle requirements and wash facilities.

2.3. Feed production

We modified the LM to reflect regional GF and DB rations (Tables 1&2) and 2001-2010 mean regional feed crop yields, weighted by land area (Conrad et al., 2016; Griffin et al., 2014). We used the Dairy One database for feed nutrient compositions

(Dairy One, 2011; Rayburn, 2008). We developed representative mineral supplement mixes to meet micronutrient requirements for DB and GF cattle (*Dairy Reference Manual*, 1995; Rayburn, 2008). We estimated herd water requirements for beef and dairy cattle according to the National Research Council (NRC, 2001, 2000).

We adapted regionally representative processes from Adom et al. (2012) for the production of hays, silages, and corn. We reviewed these processes to ensure agreement with regional agronomic recommendations (Penn State, 2015). As a result, we reduced the application rates of phosphorus and potassium fertilizers used in hay and silage production to match estimated nutrient removal rates. For co-product feeds, we used process data from the Ecoinvent database for U.S. soybean meal (v. 3.1, cut-off system model) and distillers dried grains with solubles (DDGS) production (v. 2.2) (*Ecoinvent database*, 2013). We used version 2.2 for DDGS because the version 3.1 global warming potential of DDGS was over an order of magnitude lower than version 2.2, despite no reported changes to its LCI. Version 2.2 was also the basis of the DDGS process by Adom et al. (2012). To ensure consistency and relevance of allocation, we rescaled the DDGS and soybean meal processes and allocated burdens proportional to 2009-2013 average prices (USDA-ERS, 2015c, 2015d) (Table 2.4). For DDGS, we followed Ecoinvent's approach of adding detail to the system and only allocating flows that were not specific to DDGS (*Life Cycle Inventories of Bioenergy Data v2.0*, 2007). We modeled the production and shipment of mineral supplements for both systems (i.e., sodium chloride, dicalcium phosphate, and calcium carbonate (GF only)) following Lupo et al. (2013). Feeds and supplements were transported 100 km to farms when field- or mill-to-farm transportation was not accounted for in existing unit processes (Table S2).

We modeled milk replacer production as the shipment, evaporation, and drying of milk using Ecoinvent processes.

Grass-legume pasture in the GF system was manure fertilized, with 50% of annual aboveground biomass available for consumption (i.e., a harvest efficiency of 50%). We estimated Northeast pasture biomass production to be 5,394 kg ha^{-1} by adjusting hay yields, following Conrad et al. 2016. For phosphorus loss from grazed pastures, we used Soil and Water Assessment Tool model results from the USDA National Resources Conservation Service Great Lakes and Chesapeake Bay watershed studies (USDA-NRCS, 2013, 2011). For the sub-regions with the majority of area in the Northeast[2], we calculated area-weighted average P loads to water from grazed pastureland (2.12 kg ha^{-1}).

For greenhouse gas emissions on pasture, we followed IPCC (2006) Tier 1 protocols to estimate N_2O and CO_2 emissions, ensuring continuity with feeds data from Adom et al. (2012). We assumed pasture soil organic carbon (SOC) was at equilibrium due to considerable uncertainty related to SOC fluxes over time and to match other studies (Adom et al., 2012; Lupo et al., 2013; Pelletier et al., 2010; Thoma et al., 2013b). We calculated ammonia (NH_3) emissions as the difference between total volatilized N and direct N_2O-N emissions, and the fraction of N lost via leaching or runoff ($Frac_{leach}$) from pastures as a function of potential evapotranspiration (PE) and precipitation (Pr) (Rochette et al., 2008). For the 2000-2015 growing seasons across Allentown (PA), Binghamton (NY), and Burlington (VT) Northeast Regional Climate Center weather stations, mean Pr exceeded PE, yielding a $Frac_{leach}$ of 0.30 (NRCC, 2016). We estimated

[2] Includes the following sub-region codes: 0205, 0206, 0207, 0413, 0414, and 0415 (USDA-NRCS, 2013, 2011).

potential nitrate (NO_3) emissions to water as the fraction of N leached or runoff, minus indirect N_2O emissions from leaching. To convert potential to effective NO_3 emissions, we applied an adjustment factor of 0.80, following the method used by Ecoinvent (Nemecek and Kägi, 2007).

2.4. Manure and enteric emissions

We calculated manure volatile solids (VS) produced and nitrogen excreted by breed, cattle class, and phase following ASAE (2005). We used IPCC (2006) Tier 2 protocols to calculate greenhouse gas emissions from manure storage and methane emissions from manure deposited on pastures. We estimated direct N_2O emissions using N excretion rates and IPCC (2006) Tier 1 emissions factors (EF_3), which were specific to the manure management systems (MMS) (Table 2.4). We used Tier 1 emissions factors for indirect N_2O emissions from leaching and volatilization. For the fraction of N leached or runoff from manure storage ($Frac_{leach}$), we used the median of the range of total N added (10%) indicated by the IPCC (2006). We calculated NH_3 and NO_3 emissions following the same balance and adjustment processes as for pasture production. To account for additional NH_3 emitted from dairy cattle facilities, we added 7.17 kg NH_3 cow^{-1} yr^{-1} to the LCI of the dairy calving system (Pinder et al., 2004). We estimated CH_4 emissions from manure storage and deposition on pasture as a function of VS produced, maximum CH_4 production potential (B_0) by cattle class, and methane conversion factors specific to MMS and pasture (Table 2.4). As emissions associated with manure application were included in feed crop production, we considered any difference between feed crop nutrient requirements and nutrients available for application post-MMS to be a residual at the farm gate, and thus do not impose added burden to the system. Finally, we developed IPCC (2006) Tier 2 estimates of enteric methane emissions using feed intake and digestibility from the LM. We used default methane conversion factor values (Y_m) of 6.5% for grass-fed and dairy cattle, 3% for finishing DB cattle (≥90% concentrates), and 5.5% for growing DB cattle (<90% concentrates) following Pelletier et al. (2010).

2.5. Life cycle impact assessment (LCIA)

We conducted LCIA in the openLCA software (v.1.4) (*openLCA*, 2015). We used TRACI for estimating global warming potential (GWP based on 2007 IPCC factors), acidification, and eutrophication midpoint impacts (Bare et al., 2002). We also estimated acidification and eutrophication potential per hectare of agricultural land, as suggested by de Vries and de Boer (2010). This is justified because the majority of eutrophication and acidification impacts are produced on farm or locally, and nutrient loads per unit area and environmental responses are non-linearly related (see Supplementary Material). Finally, we used the ReCiPe Midpoint (H) method for fossil and water depletion impacts (Goedkoop et al., 2012). Finally, we quantified agricultural land use with the LM.

2.6. Sensitivity analyses

We conducted sensitivity analyses to explore the robustness of LCIA results. For GF, accounting for potential carbon sequestration in managed grazing systems has reduced the GWP of beef substantially in other U.S. regions (Lupo et al., 2013; Pelletier et al., 2010). We simulated a 410 kg C ha^{-1} yr^{-1} sequestration rate for pastures under MiG (Conant et al., 2003). Additionally, harvest efficiency can vary greatly under MiG (Pers.

Communication, J. Rowntree, 4.1.15). We simulated harvest efficiencies of ±15 percentage points from baseline (50%), with the lower bound corresponding to a rule of thumb reported during producer interviews (Table A1) and USDA guidance (Green and Brazee, 2012). Finally, Chiavegato et al. (2015) found rotationally grazed beef cows emitted 4.6% of gross energy intake as CH_4, driven by high forage quality. Assuming high forage quality during the grazing season (Y_m =4.6%), and lower forage quality over winter (Y_m=6.5%), we simulate a whole herd, seasonally weighted average Y_m of 5.5%, which also corresponds to the lower uncertainty bound for Y_m given by IPCC (2006).

For DB, we conducted sensitivity analyses on all co-product allocation decisions (Table 2.4). We used 2011-2015 producer prices for milk, cows and calves to develop an economic allocation ratio between milk, dairy beef, and veal at the dairy farmgate (USDA-NASS, 2015a, 2015b). For the co-product feed soybean meal, we allocated burdens between soybean oil and meal according to their relative masses (Omni Tech International, 2010). Emissions estimates and allocation factors for DDGS vary widely (Adom et al., 2012). In addition to mass allocation, therefore, we also tested the influence of using the economic allocation factors embedded in the original Ecoinvent DDGS process.

3. Results and Discussion

3.1. Global warming potential

The global warming potential of GF and DB was 33.7 and 12.7 kg CO_2-eq. per kg HCW, respectively. These results were in the range of LCAs of grass-fed and dairy beef in the U.S. and Europe (Lupo et al., 2013; Mogensen et al., 2015; Pelletier et al., 2010;

Stackhouse-Lawson et al., 2012). Grass-fed beef from the Northern Great Plains of the U.S. (NGP) resulted in a global warming potential of 31.5 kg CO_2-eq. per kg HCW with a shorter finishing time, lighter finished weight, and higher dressing percentage (DP) (DP: 55 vs. 51%) than the present work (Lupo et al., 2013). Grass-fed beef from the Upper Midwest U.S. resulted in a global warming potential of 14.8 CO_2 eq. per kg live weight (37.6 kg CO_2-eq. per kg HCW; DP: 51%). California dairy beef (Stackhouse-Lawson et al., 2012) produced 10.7 kg CO_2-eq. per kg HCW, while Swedish dairy bull calves finished at 19 mo resulted in 11.5 kg CO_2-eq. per kg HCW.

The leading contributor to GF emissions was enteric methane (57%), with N_2O emissions from grazed pastures the second largest contribution (Figure 2.1). For DB, enteric methane was less than half of emissions (37%), though it remained the largest single contributor. Amongst all DB feeds, corn grain had the greatest impact, producing 29% of the feed burden and 9% of system emissions.

3.2. Agricultural land use

GF used 122 m^2 of agricultural land per kg HCW, versus 17 m^2 for DB. Limited data is available to compare whole herd land use for grass-fed and dairy beef production systems. Mogensen et al. (2015) report land use for a mostly grass-based beef breed system in Denmark at 155 m^2 and a dairy beef system in Sweden (19 mo at slaughter) at 20 m^2. Compared to conventional U.S. beef production using beef breeds (Peters et al., 2014), GF and DB use 25% and 90% less land per kg HCW . Average U.S. beef breed systems include continuous grazing on low productivity range and pasture land (pre-feedlot finishing) (Peters et al., 2014), driving the higher land use relative to GF.

Although GF required more land than DB, its land base was entirely perennial forage, compared to 36% for DB. Almost half (43%) of the land required for DB was devoted to the production of corn-based feeds (corn grain, corn silage, and DDGS). Indirect feed needs to produce milk replacer accounted for 5% of the land use of the DB system.

3.3. Fossil and water depletion

The fossil depletion attributed to GF and DB was 1.10 and 1.33 kg oil-eq. per kg HCW, respectively. To our knowledge, no other estimates of fossil depletion in grass-fed and dairy beef production systems exist. The production of winter forages accounted for 75% of fossil depletion for GF (Figure 2.1). Forage production is heavily reliant on diesel fuel, which was a primary contributor to its fossil depletion (44%, for grass hay). For the DB system, concentrate feed production was a major contributor to fossil depletion (44%), particularly corn grain and DDGS, which require energy intensive drying or treatment processes. As these feeds are used heavily in growing and finishing, they accounted for 38% of DB fossil depletion.

Water depletion for GF and DB was 0.074 m^3 and 0.065 m^3, respectively. A recent review of beef production system LCAs indicated water use impacts were a knowledge gap (de Vries and de Boer, 2010); thus few comparisons are available. However, Rotz et al. (2015) found that including dairy beef (calves and culls) in an analysis of beef production systems in the Southern Great Plains reduced water use by 4% (Rotz et al., 2015). For the GF system, pumping groundwater to meet drinking requirements accounted for the majority of the depletion impact (90%). Drinking and wash water accounted for a smaller fraction of water depletion for DB (59%), with another 30% of the total burden attributed to concentrate feeds. The ReCiPe methodology attributes a higher depletion impact to the use of ground/well water compared to surface water (Goedkoop et al., 2012). All feeds were grown using a surface water source, except soybean meal, which was irrigated with ground water.

3.4. Eutrophication and acidification potential

Per kg HCW, the eutrophication potential of GF was 0.44 kg N-eq. versus 0.18 kg N-eq. for DB (Figure 2.2a). We converted N-eq. emissions to PO_4^{3-} -eq. emissions to compare to the literature (Baumann and Tillman, 2004), which yielded 184.8 g and 75.6 g PO_4^{3-} -eq. per kg HCW for GF and DB, respectively. No U.S. studies have examined the eutrophication potential of dairy beef systems, to our knowledge. Dairy bull calves raised to 16 mo in the EU produced a similar burden (73.7 g PO_4^{3-} -eq. per kg carcass weight) to the DB system (Nguyen et al., 2010). Phosphorus and nitrate losses from feed crop production accounted for the majority of the emissions for the DB system (86%), largely due to corn silage and grain production (76% of total).

The eutrophication potential of GF falls within a range of values for grass-fed systems reported in other U.S. regions. In the NGP, grass-fed beef production resulted in 35.1 g PO_4^{3-} -eq. per kg carcass weight (Lupo et al. 2013) compared to 142g PO_4^{3-} -eq. per kg liveweight in the Upper Midwest (278.4 g per kg carcass weight, DP: 51%) (Pelletier et al., 2010). The large disparity between studies may be partially driven by differences in modeled pasture systems. Emissions in the GF system were driven by nitrate leaching under grazed pastures (47%), with phosphorus loss from runoff and erosion on pastures the second largest contributor (26%). For phosphorus loss, we used pasture specific emissions from process modeling, whereas Lupo et al. assumed a loss rate of 2.9% of P excretion (Lupo et al., 2013). Although we used the same method as Lupo et al. to estimate nitrate leaching, our resulting rate was three times as large as that of the NGP (10%), due to Northeast climate conditions. Pelletier et al. (2010) also used a 30% leaching rate for the Upper Midwest, in combination with assumed synthetic fertilization of grazed pastures annually. Pastures in the NGP system were only fertilized with manure, similar to this study (Lupo et al. 2013). Finally, pasture legume content, and thus nitrogen excretion during grazing may have varied between studies. The 20% legume pastures in the GF system had a crude protein concentration (17% of DM) that exceeded beef cattle requirements, resulting in excess nitrogen excretion. One of the pastures modeled by Pelletier et al. (2010) was 40% legume, while the legume content of the pastures by Lupo et al. (2013) was not provided.

Potential acidifying emissions were 30.2 and 13.1 moles H+-eq. per kg HCW for GF and DB (Figure 2.2b). We converted H+-eq. emissions to g SO_2-eq. emissions with a conversion factor from TRACI (Hischier and Weidema, 2010). The resulting emission for GF (593.6 g) was approximately twice the terrestrial acidification potential of the grass-fed system in the NGP (299.1 g) per kg HCW (Lupo et al., 2013). As most of the GF burden was due to NH_3 (94%) primarily from manure deposition and storage (88% of system total), differences in N excretion, manure management, and manure accounting may explain the larger emission compared to Lupo et al. (2013). For DB, SO_2-eq. emissions per kg carcass weight (258.3 g) were also higher compared to dairy bull calves raised to 16 mo in the EU (131 g) (Nguyen et al., 2010). Nguyen et al. (2010) used different methods to estimate emissions of NH_3, the pollutant accounting for 87% of DB emissions in this study. Additionally, Nguyen et al. (2010) excluded cull meat and burdens from their system boundary and allocated only the burden resulting from the production of feed to meet a cow's pregnancy requirements to the dairy bull calf. Conversely, we allocated 9.4% of the entire dairy burden to the DB system, resulting in 46% of the system's potential acidification (Figure 2.3). Non-feed burdens from the dairy system accounted for 28% of the DB impact (results not shown), highlighting the influence of allocating dairy system burdens beyond feeds.

Calculating acidification and eutrophication potential on an area basis roughly reversed the differential between GF and DB (Figure 2). For example, GF had a 2.4 times larger eutrophication impact than DB per kg HCW, whereas DB had a 3.1 times greater eutrophication impact than GF per ha of agricultural land. While comparing these impacts on an area basis helps assess risks to regional environmental quality, any future research examining extensification to reduce pollution must consider potential inducement of emissions (i.e. leakage) elsewhere.

3.5. Role of the breeding herd

Maintaining the breeding herd accounted for 60% and 55% of system impacts averaged across categories for GF and DB, respectively (Figure 2.3). In beef breed cattle systems, it is well established that the breeding herd is responsible for the majority of impacts due to low rates of reproduction (de Vries et al., 2015). For dairy beef, the relative contribution to total system burdens from the breeding herd is not well established, being complicated by differing allocation methods and system boundaries.

Research in Europe and California on the impacts of dairy bull calf fattening systems excluded the impacts and meat produced from culled dairy cattle, making direct comparison impossible (Mogensen et al., 2015; Nguyen et al., 2010; Stackhouse-Lawson et al., 2012). Despite a similar relative contribution to total system burdens, our results reinforce previous findings that the absolute contribution of the DB breeding herd to whole system impacts tends to be small compared to that of beef breed systems due to multi-functionality (de Vries et al., 2015). However, fossil and water depletion are notable exceptions. The breeding herd accounted for similar absolute water depletion (0.04 vs. 0.05 m^3) and fossil depletion (0.59 vs. 0.61 kg oil-eq.) per kg HCW, within DB and GF, respectively.

Strategies to reduce emissions of U.S. dairy farms has been an area of extensive research, and we refer readers to the literature for that discussion (Asselin-Balençon et al., 2013; Thoma et al., 2013b). At the same time, it bears noting that productivity enhancement has been touted as an key mechanism to reduce the sector's carbon footprint (Capper and Bauman, 2013). However, this approach may lead to less dairy beef production, calling into question the net benefits of further intensification and specialization (Zehetmeier et al., 2012). Proposed strategies to reduce the environmental consequences of dairy production should consider influences on dairy-sourced beef to assess whole system sustainability.

3.6. Sensitivity analyses

Full results of the sensitivity analyses are provided in Table A3. Simulating a harvest efficiency of 35% resulted in 25% more land used, 11% greater eutrophying emissions per kg HCW, but 11% lower eutrophying emissions per ha compared to GF at baseline. Decreasing harvest efficiency did not change acidifying emissions per kg HCW but reduced emissions per ha by 20% relative to baseline. Increasing harvest efficiency to 65% decreased land use by 14% and increased eutrophication and acidification potential per ha by 9 and 16%, respectively. For DB, using an economic value allocation between milk and beef reduced GWP by 9%, agricultural land use by 10%, and water depletion by 11%. Alternative co-product feed allocation scenarios did not influence any impact categories greater than 8% relative to baseline.

Including pasture C sequestration and an Y_m of 5.5% reduced the global warming potential of GF by 32 and 9%, respectively. Accounting for potential pasture carbon sequestration yielded a similar estimate for GWP of grass-fed beef (22.9 kg CO_2-eq. per kg HCW) compared to studies in other U.S. regions (21.6 – 23.9 kg CO_2-eq.) (Lupo et al., 2013; Pelletier et al., 2010). Combined, these two parameters resulted in 19.9 kg CO_2-eq. per kg HCW, which is near the low end of a range of U.S. beef breed systems previously studied (Lupo et al., 2013; Pelletier et al., 2010; Rotz et al., 2015; Stackhouse-Lawson et al., 2012). In addition to high uncertainty, the scope of this study precludes any conclusion about the ecological performance of the GF system compared to beef production systems outside the region. Given the influence these parameters had on GF global warming potential, however, experimental research is critically needed to determine long-term C sequestration rates and enteric methane emissions under high stocking density MiG grazing in the Northeast.

3.7. Implications for regional beef systems

Feed production was generally a substantial contributor to both systems' footprints across categories. Substituting waste byproducts into rations may be an opportunity to reduce impacts and enhance the human food supply (van Zanten et al., 2015, 2014). Importing waste byproducts into DB would likely reduce land requirements (van Zanten et al., 2014), but could potentially concentrate nutrients in systems. For GF, this strategy may be economically infeasible due to time costs and infrastructure constraints. Additionally, feeding waste byproducts may not fit within existing guidelines for 100% grass-fed beef or match consumer expectations for a 100% grass-fed product. Future research should explore the cost-effectiveness, consumer acceptance, and net environmental benefits of utilizing waste byproducts in these systems.

The DB system had a much lower carbon and land footprint, but much higher eutrophication and acidification impacts per unit land compared to the GF system at baseline. Optimizing across impact categories in these production systems may produce a more sustainable product. Given intense regional interest in and opportunity for grass-based systems, future research could explore the potential for integrating MiG into dairy beef systems. Management-intensive grazing Holstein steers during the growing phase can be an effective production system (Mouriño et al., 2003; Schlegel et al., 2000). However, calf health and ration composition reduced the effectiveness of this model in a regional trial, which would need to be addressed (Baker, n.d.; Fanatico, 2010).

Finally, differences in the type and amount of land used between the two systems require further analysis to assess their sustainability. The greater use of cultivated cropland by DB may be an inefficient allocation of resources. Redirecting human edible crops that are currently used to feed livestock to humans is a key opportunity to enhance global food security (West et al., 2014). Furthermore, reallocating highly productive cropland that is not currently producing human edible food (e.g., corn silage) would use land more efficiently from a food supply perspective (Garnett, 2009). At the same time, although GF does not rely on cultivated cropland, the suitability of its forage land base to produce human food is unclear. While producing beef on marginal grazing lands may provide a net positive contribution to the human food supply (de Vries et al., 2015), the potential suitability of grassland to produce human food should be taken into account (van Zanten et al., 2015). On the other hand, maintaining grassland in pasture-based farming systems may provide other benefits beyond human food production, such as biodiversity preservation, erosion regulation, and cultural value (Franzluebbers et al., 2012; Ripoll-Bosch et al., 2013). Understanding and balancing the desires of regional consumers, the tradeoffs between intensive and extensive beef production systems, and how regional beef production-consumption systems fit within the larger global context should be priorities for future research.

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Table 2.1. Northeast GF system characteristics

Life stage	Head[3]	Days	Beginning and end weight (kg)	Mortality (%)	Feed consumed (kg DM head^{-1} d^{-1}) GL pasture[1]	Grass hay	GL baleage[2]
Production cattle							
Pre-weaned calves	29	207	variable	3.6	included with cow		
Backgrounding heifers	11	165	217 - 308	0.7	-	3.1	3.0
Backgrounding steers	14	165	236 - 340	0.7	-	2.1	4.4
Finishing beef heifers	11	310	308 - 478	0.8	6.6	2.8	-
Finishing beef steers	14	340	340 - 544	0.8	6.5	3.6	-
Breeding herd cattle							
Breeding cows	30	365	454 - 544	1.5	6.9	5.3	-
Growing replacement heifers	3	249	224 - 354	2.1	4.6	2.8	0.1
Bred replacement heifers	3	274	354 - 454		5.9	4.4	-
Growing replacement bulls	0.3	523	236 - 590	2.1	6.0	1.7	3.0
Breeding bulls	1	365	590 - 892	1.5	9.3	7.0	-

[1] GL pasture, 80:20 mix of grass-legume pasture.

[2] GL baleage, 80:20 mix of grass-legume bale silage.

[3] Head in stage are the average of cattle entering and leaving the stage, after mortality losses.

Table 2.2. Northeast DB system characteristics

Life stage	Head[3]	Days	Beginning and end weight (kg)	Mortality (%)	Feed consumed (kg DM head^{-1} d^{-1}) Alfalfa silage	Corn grain	Corn silage	DDGS[4]	Hay[5]	Milk powder	SBM[6]
Production cattle											
Starter calves	80	175	44 - 181	9.4	-	1.0	-	0.1	1.2	0.2	0.3
Growing heifers	19	125	181 - 315	0.9	-	0.4	4.5	1.2	-	-	-
Growing steers	59	125	181 - 348		-	0.8	4.4	1.3	-	-	-
Finishing heifers	19	185	315 - 529	1.3	-	6.7	0.3	1.0	-	-	-
Finishing steers	58	185	348 - 602		-	7.3	0.7	0.9	-	-	-
Breeding herd cattle[1]											
Breeding cows[2]	328	409	650	6.1	4.4	1.9	9.9	0.4	2.0	-	1.6
Replacement calves	116	86	44 - 100	7.0	-	0.7	-	-	0.1	0.4	0.1
Growing replacement heifers	113	402	100 - 423	1.5	-	1.0	-	-	5.5	-	-
Bred replacement heifers	113	279	423 - 601		-	2.3	-	-	9.9	-	-
Growing replacement bulls	0.3	644	100 - 635	1.5	-	0.3	-	-	7.4	-	-
Breeding bulls	1	365	635 - 907	6.1	-	-	-	-	15.9	-	-

[1] Values are prior to milk:beef allocation and exclude indirect feed needs.
[2] Includes early, mid, and late lactation stages.
[3] Head in stage are the average of cattle entering and leaving the stage, after mortality losses.
[4] DDGS, dry distillers grains with solubles.
[5] Hay is grass for starter phase, 50:50 mix of grasses and legumes for breeding herd.
[6] SBM, soybean meal.

Table 2.3: Summary of allocation ratios used

Co-product and allocation method	Ratio[1]
Dairy beef/veal/milk	
Biophysical	9.4:0.4:90.2
Economic[2]	7.8:0.9:91.3
DDGS dry/corn ethanol	
Economic	20:80
Mass[3]	48:52
Ecoinvent, economic[4]	2:98
SBM/soybean oil	
Economic	65:35
Mass[5]	80:20

[1]The first allocation ratio under each co-product is used at baseline.

[2] USDA-NASS, 2015a;b.

[3] Adom et al. 2012.

[4] Ratio refers only to shared corn ethanol/DDGS processes (i.e., excludes stillage treatment). (Source: Life Cycle Inventories of Bioenergy Data v2.0, 2007).

[5] Omni Tech International, 2010.

Table 2.4: Manure management system-specific greenhouse gas emissions parameters

Manure management system	Percent herd VS[1]	MCF[2] (%)	N emissions parameters		
			EF_3 (%)[3]	$Frac_{gas}$[4]	$Frac_{leach}$[5]
GF Herd					
Solid storage	20	2	0.5	0.45	0.10
Pasture	80	1	2	0.20	0.30
DB Herd					
Slurry, crust	46	10	0.5	0.08	0.10
Slurry, crustless	23	17	0	0.40	0.10
Solid storage	25	2	2	0.45	0.10
Deep bedding	6	17	1	0.30	0.10

[1] VS, volatile solids production. Values for DB are post-allocation.

[2] MCF, methane conversion factors for regions with average annual temperatures of ≤10° C (Source: IPCC, 2006).

[3] EF_3, percent of manure N as direct N20-N volatile loss (Source: IPCC, 2006).

[4] $Frac_{gas}$, fraction of manure N as volatile NH_3-N and NO_x-N loss (Source: IPCC 2006, except "slurry, crust," which is estimated from Rotz, 2004).

[5] $Frac_{leach}$, fraction of manure N leached or runoff (Source: Rochette et al., 2008; IPCC, 2006).

Figure 2.1: Process contributions to global warming potential, agricultural land use, fossil and water depletion of GF and DB systems. Impact categories are normalized on a 0 to 1 scale, with 1 representing the largest value for that category. Eutrophication and acidification impacts are displayed separately in Fig. 2.2.

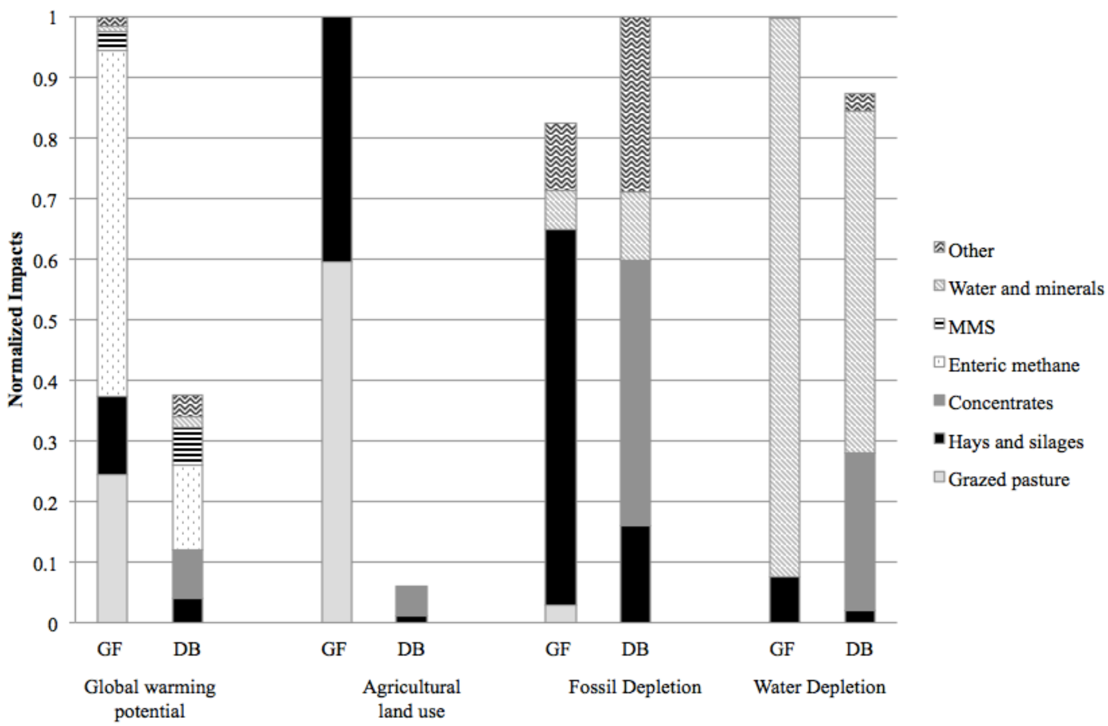


Figure 2.2: Process contributions for eutrophication (a) and acidification (b) potential of GF and DB systems on functional unit and area bases.

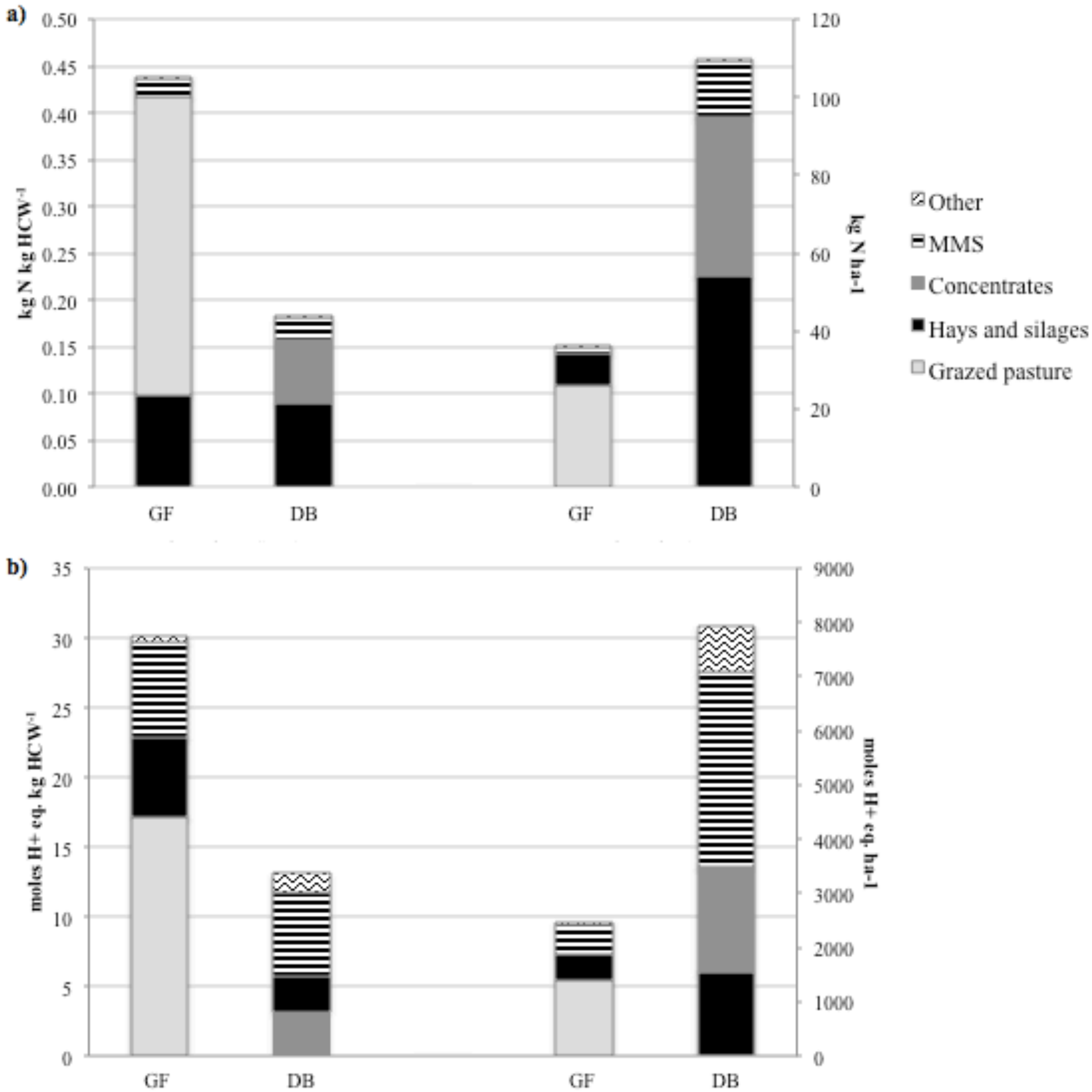
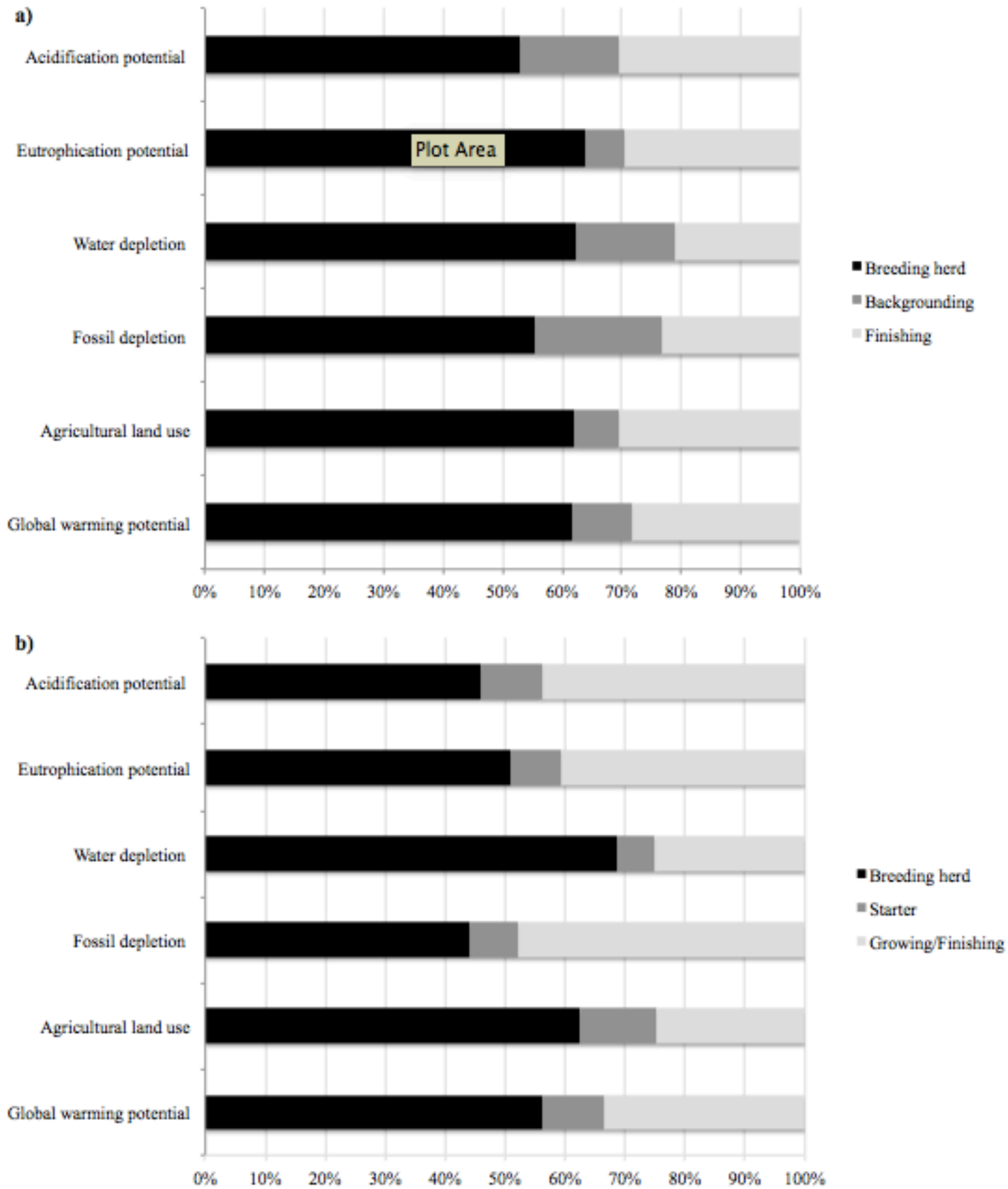


Figure 2.3: Herd contributions to LCIA impact categories for a) GF system and b) DB system.



Appendices

The Appendices address five topics in the following order: (1) data justifying the area-based comparison for eutrophication and acidification production system data for the (2) grass-fed beef system and (3) dairy beef system, (4) full results of the sensitivity analyses, and (5) a list of all references cited in the Supplementary Material. For the production system data, assumptions are categorized into farming system characteristics, feed systems, and herd performance metrics. Herd performance metrics are only reported if they were altered from assumptions reported by Peters et al. (2014) for U.S. beef cattle, dairy or dairy beef production systems. For each production system parameter, the value used in the livestock model is reported with the units of measurement, an explanation of the methods used to derive the parameter and relevant data sources. Measures of dispersion reported with parameter values or explanations are standard deviations, unless otherwise reported.

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1. Area-Based Comparison: Eutrophication and Acidification

We assume that harvested forages for both systems are produced either on farm or locally, due to the weight of forages and expense of transport. For GF, 100 and 98% of eutrophication and acidification impacts for GF were produced on farm or locally (MMS, grazed pasture, and harvested forages). For DB, 62 and 65% of eutrophication and acidification burdens were produced on farm or locally (MMS and harvested forages).

There are relationships for acidification and eutrophication between pollutant load and environmental response that are both non-linear and related to land area. For acidification, total sulfur and nitrogen deposition rates of less than 5 kg ha yr^{-1} are necessary to protect the most acid-sensitive ecosystem components in the Eastern U.S. (Burns, 2011). For eutrophication, for example, the ability of a soil to act as a sink for phosphorus declines as P accumulates (Kleinman et al., 2011), and soil P saturation thresholds have been demonstrated (Griffin et al., 2003).

2. Table A1: Grass-fed Beef Production System Data

2.1. Farming System Characteristics

Phase	Parameter	Units	Value	Explanation and Sources
	General			
All phases	Growth efficiency technology	n/a	None	Grass-fed beef is a niche product, often marketed as "hormone and antibiotic free." Confirmed by producer interviews (n=9).
	Breed	n/a	Small-frame Angus	Angus was the predominant breed in a regional GF beef survey (Steinberg and Comerford, 2009). Smaller frame variant would be likely (Pers. Communication, M. Baker, 3.28.15), which was confirmed by producer interviews (n=5).
	Winter manure management			
Calving	Winter manure management system (MMS)	n/a	Pasture	Calving in late winter or early spring requires little to no housing (Comerford et al., 2013). Confirmed by producer interviews: over 70% of producers with inventories of breeding cattle classes (n=7) used this system.
	Energy use: MMS	gal yr^{-1}	0	Management of manure on pasture for breeding herd does not require energy.
Backgrounding	Winter MMS	n/a	Solid storage	Backgrounding calves are kept in a barn or on a concrete pad with bedding and access to a small pasture (Pers. Communication, M. Baker, 3.28.15). Confirmed by producer interviews: 57% of producers with backgrounding calves used this

				system (n=7).
	Energy use: MMS	gal yr^{-1}	13.5	Manure scraping and piling takes 14 hours (3-4 times per year, 4 hours each time) using a skid steer loader (Pers. Communication, M. Baker, 3.28.15). Confirmed reasonable with producer interviews: total scraping duration was 12 ±11 hours with a skid steer loader or small tractor (n=5). Total fuel use for a 66 hp tractor (author's assumption) was estimated according to Grisso et al. 2010. Fuel use was allocated between backgrounding calves (1/3) and finishing cattle (2/3).
Finishing	Winter MMS	n/a	Solid storage	Finishing cattle are kept in a barn or on a concrete pad with bedding and access to a small pasture (Pers. Communication, M. Baker, 3.28.15). Confirmed by producer interviews: 83% of producers with finishing cattle used this system (n=6).
	Energy use: MMS	gal yr^{-1}	27	See "Backgrounding, Energy use: MMS" for details.
	Breeding			
Calving	Bull transport	t-km $head^{-1}$	189	Yearling bulls are purchased from within the region (Pers. Communication, M. Baker, 3.28.15). Author's assumption of 320 km (≈200 miles) traveled one-way confirmed reasonable with producer interviews: producers reported bulls transported 340 ± 206 km one-way (n=5). A 590 kg yearling bull is assumed.
	Bull import rate	Head yr^{-1}	0.4	Producers buy one yearling bull every 2 to 3 years (Pers. Communication, M. Baker, 4.23.15). Confirmed reasonable with producer interviews: producers purchased 1 bull every 2.9 ±1 years (n=5).
	Cow inventory	Head	30	A survey of grass-fed beef producers in the region reported a slaughter rate of 25.1 head $farm^{-1}$ yr^{-1} (Steinberg and Comerford, 2009). According to Peters et al. (2014), 0.85 head of cattle are produced cow^{-1} yr^{-1}, yielding a

				cow inventory of 30 on grass-fed beef operations by back calculation.

2.2. Feed: Rations, Pasture Production, and Provisioning

Phase	Category	Units	Value	Explanation and Sources
Calving and finishing	***Grazing season ration***			
	Pasture type	n/a	80% grass, 20% legume	81% of grass-fed beef producers in a regional survey reported grass-legume mixes composing 83% of total farm pasture (Steinberg and Comerford, 2009). Average Northeast pastures are 21% legume (Goslee, 2014). Author assumes a pasture mix of 80% grass, 20% legume. Confirmed reasonable with producer interviews, where producers estimated 28 ± 16% legume content of pastures (n=8). The baseline pasture mix assumption was unchanged due to high variability and lack of empirical farm data on pasture composition.
	Proportion of ration as grazed	%	100	Pasture should meet the herd's feed requirements during the grazing season (Personal Communication, M. Baker, 3.28.15).
	Grazing season duration	d	200	The grazing season in the Northeast is 200 days, on average (Pers. Communication, M. Sanderson, 9.12.14; Pers. Communication, M. Hall, 8.5.14). Confirmed reasonable with producer interviews: 207 ± 43 d grazing to meet the majority of the herd's forage requirement (n=8).
	Mineral supplementation rate	oz $head^{-1}$ d^{-1}	2	A mineral mixture of 48% salt, 31% dicalcium phosphate, 15% limestone and 3% selenium mix can be provided to pastured cattle at this rate (pg. 63) (Rayburn, 2008). Producer interviews confirmed that a mineral mixture supplement is administered (n=8).
All phases	***Non-grazing season ration components***			
	Grass hay	n/a	n/a	Dry hay is fed in Northeast grass-fed beef systems (Steinberg and Comerford, 2009) (Pers. Communication, B. Chedzoy and R.

				Brubaker, 3.28.15). Grass hay was assumed and confirmed reasonable with producer interviews (n=9).
	Grass-legume bale silage	n/a	80% grass, 20% legume	Bale silage is fed to cattle in grass-fed beef systems in the Northeast and upper Midwest (Rowntree, 2014; Shinn, 2013; Steinberg and Comerford, 2009) Grass-legume bale silage was assumed (Pers. Communication, M. Baker, 4.23.15). Two producers reported feeding bale silage, with extreme variability in estimated legume percentage (70 percentage points). Due to high variability and lack of empirical farm data, the baseline assumption was unchanged.
	Mineral supplementation rate	oz $head^{-1}$ d^{-1}	2	Same as grazing season.
All phases	***Water and feed provisioning***			
	Electricity use, water pumping for whole herd	kWh	72	Electricity use was estimated using total herd drinking water requirements, depth of the well, flow rates and the power requirement. The author assumes a flow rate of 2,271 L hr^{-1}, based on water provisioning scenarios for pastured cattle reported by Rayburn, 2007. A distance of 67 m is assumed, using average well depths reported by USDA (USDA-NASS, 2008). 0.34 kW is required to pump the flow rate the required distance ("Pumps, Fans and Turbines - Horsepower," nd).
	Water source	n/a	ground	75% and 50% of producers interviewed pumped drinking water to meet all or part of herd requirements during the winter and grazing season, respectively (n=8).
	Diesel use, winter feeding	L	1,034	Fuel use for delivering winter forage was estimated using total winter feed requirements, capacity and speed of delivery, and fuel efficiency. A small tractor or skid steer loader (66 hp) with a two bale fork has a point in

				time capacity of 907 kg ("Skidsteer Mount, Double Bale Spear," 2016). The authors assumed 1,814 kg could be fed per hour. On an animal unit (AU) basis, this is 1.33 hr AU^{-1}. Fuel use per hour was estimated at 11 L hr^{-1} according to Grisso et al. 2010. Total fuel use was estimated by multiplying time required to deliver all winter feed (94 hr) by fuel use per hour. Confirmed reasonable with producer interviews, where feeding required slightly less time (1.17 hr AU^{-1}) with slightly more powerful machinery (73 hp).
	Pasture production and grazing management			
Calving and finishing	Harvest efficiency (HE)	%	50	Average rotational systems may have a HE around 50% (Per. Communication, J. Rowntree, 4.1.15). Using this HE and estimated regional pasture yield, pasture intake was estimated as 2,697 kg ha^{-1}. This estimate was confirmed by producer interviews. Producer reported grazing animal inventories, including other species, were converted to animal unit equivalents and DMI d^{-1} was estimated according to NRCS standard forage demand (USDA-NRCS, 2003). Total DM demand was estimated by multiplying days on pasture by DMI d^{-1}. Total DM demand was then divided by land area grazed, and grazable yield was estimated as 2,674 ± 1,496 kg ha^{-1}, only 1% less than the baseline estimate, but with considerable variability.
	Fertilizer application rate	kg ha^{-1}	0	Permanent pastures are fertilized only with manure and biological fixation. Confirmed reasonable with producer interviews: 78% of farms did not apply N, K, or lime and 89% of farms did not apply P.
	Irrigation water use	L yr^{-1}	0	No irrigation mentioned in a regional survey of grass-fed beef producers

				(Steinberg and Comerford, 2009). Confirmed by producer interviews: no farms irrigated pastures.
	Diesel use, frost seeding	L yr^{-1}	23	Legumes are frost seeded into pastures every 2-4 years (Rayburn, 2006). Fuel use is calculated assuming a 30 hp tractor and 14 hour duration of reseeding estimated during producer interviews (n=4), according to Grisso et al. 2010.
	Diesel use, herd rotation	L yr^{-1}	195	88% of producers interviewed used an all terrain or utility vehicle to rotate their herd (n=8). Producers rotated their herd 348 times yr^{-1} (1.75 times dy^{-1} multiplied by days grazing), each rotation requiring approximately 20 min. The authors assume that the vehicle is running for 50% of rotation duration. An equation by Grisso et al. (2010) to calculate tractor fuel use is used as a proxy for utility vehicle fuel use, assuming a 22 hp vehicle.
	Electricity use, fencing	kWh yr^{-1}	88	88% of producers used electric fencing entirely or partly powered by AC electricity (n=8). An 8 output J energizer was used, on average. Energizers of this strength may use 0.01 kW of power, according to a fencing retailer (AB Custom Fencing Online Store, 2016). Fencing is assumed to be electrified year round (8,400 hr).
	Stored manure loading and spreading	kg	20,805	Total solids production was calculated as herd total volatile solids production divided by 0.80 (Hamilton and Zhang, nd). The ecoinvent 3.1 unit process “solid manure loading and spreading” for Canada, which has a mass based reference flow (kg), was used as a proxy.

2.3. Herd Performance Metrics

Phase	Category	Units	Value	Explanation and Sources
	Animal size			
Calving	Weaned weight of bulls	kg	236	Pers. Communication, J. Rowntree, 4.1.15
	Weaned weight of replacement heifers	kg	224	Extrapolated based on weighted of replacement heifers being 3% heavier than other heifers, according to Peters et al. (2014).
	Mature bull (2 yr. old)	kg	590	Extrapolated based on weaned weight of mature bulls being 2.5 times their weaned weight at 2 years of age, according to Peters et al. (2014).
Background-ing	Weaned weight of other heifers	kg	217	Extrapolated based on weaned weight of replacement heifers being 92% of weaned steers weight, according to Peters et al. (2014).
	Weaned weight of steers	kg	236	Pers. Communication, J. Rowntree, 4.1.15
Finishing	Heifer entering finishing	kg	308	Extrapolated based on weight of heifers entering finishing being 90.5% of steers entering finishing, according to Peters et al. 2014.
	Steer entering finishing	kg	340	Weight for steers in Michigan grass-fed beef system at beginning of grazing season (Rowntree, 2014).
	Finished heifer	kg	479	Extrapolated based on finished weight of heifers being 87.9% of finished steer weights, according to Peters et al. (2014).
	Finished steer	kg	544	Pers. Communication, J. Rowntree, 4.1.15. Confirmed by producer interviews: mean finished weights of steers were estimated at 542 ± 34 kg (n=6).
	Time spent in each life phase			
Background-ing	Steers	d	165	The authors assume that weaning occurs at the end of a 200 d grazing season and backgrounding occurs over the winter feeding period.
	Heifers	d	165	Same as steers.
Finishing	Steers	d	340	Steers were initially estimated to spend 293 d in finishing and 665 d total from birth to finish, based on finishing times

				of 20- 24 months in the region (Pers. Communication, M. Baker, 4.23.15). Producer interviews indicated a mean finishing time for all cattle of 712 ± 61 d, and the modeled time in finishing was increased accordingly.
	Heifers	d	310	Heifers were initially estimated to spend 265 d in finishing and 635 d total from birth to finish, assuming heifers finish 30 d earlier than steers (Pers. Communication, M. Baker, 4.23.15). Time in finishing was adjusted, as with steers.
	Mortality			
Finishing	Finishing cattle	%	0.75	Mortality rates of grass-finished cattle range from 0.5-1%, lower than national averages (1.3%) used by Peters et al. (2014) (Pers. Communication, J. Rowntree, 4.1.15).
	Reproduction			
Calving	Calving percentage	%	97.7	A calving percentage of 91.5% was assumed initially (Peters et al., 2014). Producer interviews indicated a mean calving percentage of 97.7% ± 3%, and the parameter was adjusted accordingly after it was confirmed to be reasonable by a regional expert (Pers. Communication, M. Baker, 4.13.16).

3. Table A2: Dairy Beef Production System Data

3.1. Farming System Characteristics

Phase	Parameter	Units	Value	Explanation and Sources
	General			
Calving	Growth efficiency technology, rBST	%	32.4%	32.4% of farms in region 1 reported using rBST in their herds (Thoma et al., 2010)
Starter	Growth efficiency technology	n/a	Medicated milk replacer	Calves on starter operations are assumed to be fed a medicated milk replacer, including a coccidiostat and antibiotic (Geyer, 2012) .
Growing and finishing	Growth efficiency technology	n/a	Ionophore	The data used to predict intake during growing and finishing is from a large sample of feedlot cattle implanted with an ionophore (Peters et al., 2014).
	Manure management			
Calving	Manure management system (MMS)	n/a	Earthen ponds/ tanks, slurry	95% of total manure DM on region 1 dairies was managed as a slurry (Authors' calculations, using Thoma et al., 2010). 43% and 24% of total manure DM on region 1 dairies was managed in "earthen ponds/tanks with a natural crust cover" and "earthen ponds/tanks without a natural crust cover" (*Statistical Analysis and Interpretation of Producer Survey, Section 7*, 2010; Thoma et al., 2010). These are the only two slurry systems in the top six ranked MMS in region 1 (*Statistical Analysis and Interpretation of Producer Survey, Section 7*, 2010). Thus, it is assumed that 66.7 and 33.3% of manure is managed in crusted and crustless slurry systems, respectively.
Starting	MMS	n/a	Deep bedding, stored > 1 mo.	Authors' assumption. Starter operations have dirt floors (Pers. Communication, D. Vermeire, 10.8.15) and may bed with straw (Pers. Communication, S. Boyles, 10.7.15, Pers. Communication, D. Vermeire, 10.8.15). Producers clean out the barn after calves move to the

				next phase (Pers. Communication, S. Boyles, 10.7.15, Pers. Communication, D. Vermeire, 10.8.15), which is 175 d in this analysis.
	Lime application	kg	34	Many small starter operations are Amish or Mennonite, relying heavily on manual labor (Pers. Communication, D. Vermeire, 10.8.15). After cleaning out the barn manually, lime would be applied to the floor (Pers. Communication, S. Boyles, 10.7.15, Pers. Communication, D. Vermeire, 10.8.15),
Growing and Finishing	MMS	n/a	Solid storage	Manure is scraped from lots and stored as a solid (Pers. Communication, L. Geyer, 4.30.15; Pers. Communication, S. Boyles, 10.7.15). Producer responses on MMS (n=2) were mixed, but included solid storage.
	Energy use: MMS	L	602	Each manure scraping and piling takes four hours using a tractor and front loader (Pers. Communication, S. Boyles, 10.7.15). Using the mean of frequency estimates from two experts, scraping is assumed to occur nine times per phase (Pers. Communication, L. Geyer, 4.30.15; Pers. Communication, S. Boyles, 10.7.15). Total fuel use for a 100 hp tractor (authors' assumption) was estimated according to Grisso et al. 2010.
	Cattle transport			
Starter	Calves to starter phase	km	400	Calves that are raised in the greater region may make one medium/long distance trip and one short distance trip. For example, up to half of calves that are born in New York may stay in New York at times (Pers. Communication, B. Buchanan and Northern NY Cornell Cooperative Extension Agents, 2.11.15), and then could be shipped to Ohio or
Growing and Finishing	Weaned calves to growing and finishing	km	400	

				Pennsylvania (Pers. Communication, M. Broccoli, 1.28.15; Pers. Communication, B. Buchanan and Northern NY Cornell Cooperative Extension Agents, 2.11.15; Pers. Communication, L. Geyer, 4.30.15). Calves could also be shipped a medium/long distance first, and then be sent to a local feedlot for growing and finishing. The authors assume 400 km per trip based on the mean of summed potential local (150 km) and medium/long distance (650) trips from central New York, which was chosen as a regional midpoint.

3.2. Feed: Rations and Provisioning

Phase	Category	Units	Value	Explanation and Sources
	Ration components			
Calving	Corn grain Grass-legume hay Alfalfa silage Corn silage Soybean meal DDGS, dry Mineral supplement	n/a	n/a	The dairy module of the livestock model, which excludes grazing, is considered a reasonable proxy for the Northeast (Conrad et al., 2016). Composition of the ration was refined using non-grazing season rations for region 1 from Thoma et al. 2013, which included 27 aggregate feeds. Feeds were excluded from the present analysis if they a) accounted for < 5% of total herd DMI *and* b) accounted for < 5% of the carbon footprint of the ration (Thoma et al., 2010). The sum of all feeds excluded accounted for 8% of total herd DMI, 3% of which was various silages.
	Milk powder	n/a	n/a	86% of operations feed milk replacer to calves (USDA-APHIS, 2012, p. 201). This was confirmed reasonable for a 300 cow dairy in the region (Pers. Communication, B. Buchanan, 1.25.2016).
Starter	Milk powder	n/a	n/a	Dairy beef calves consume milk replacer prior to weaning (*Dairy Reference Manual*, 1995). Confirmed with producer interviews.
	Corn grain	n/a	n/a	Corn grain is the primary energy source in calf feeds on starter operations (Pers. Communication, M. Baker, 1.26.15; Pers. Communication, L. Geyer, 4.30.15; Pers. Communication, D. Vermeire, 10.8.15). Confirmed reasonable with producer interviews.
	Protein mix	n/a	n/a	Soybean meal is the primary protein source in calf feeds on starter operations (*Dairy Reference Manual*, 1995; Geyer, 2012) (Pers. Communication, D. Vermeire, 10.8.15). The protein mix modeled is 80% soybean meal, 20% DDGS dry. DDGS may also be used as a protein source (Pers. Communication, M. Baker, 1.26.15), with a maximum inclusion rate of 30% DMI (Lehmkuhler

				and Burris, 2011).
	Grass hay	n/a	n/a	Grass hay is a common forage source during the starter phase (*Dairy Reference Manual*, 1995; Geyer, 2012) (Pers. Communication, M. Baker, 1.26.15; (Pers. Communication, D. Vermeire, 10.8.15). Confirmed reasonable with producer interviews.
	Mineral supplement	kg head^{-1} dy^{-1}	0.06	Starter calves should be supplemented with di-calcium phosphate and salt at this rate (Dairy Reference Manual, 1995).
Growing and Finishing	Corn grain	n/a	n/a	Corn grain is the primary energy source for growing and/or finishing (Pers. Communication, M. Baker, 1.26.15; Pers. Communication, B. Buchanan, 2.11.15; Pers. Communication, L. Geyer, 4.30.15). Confirmed by producer interviews.
	Corn silage	n/a	n/a	Corn silage is a primary roughage source for growing and/or finishing (Pers. Communication, M. Baker, 1.26.15; Pers. Communication, B. Buchanan, 2.11.15; Pers. Communication, L. Geyer, 4.30.15; Pers. Communication, E. Richer, 10.12.15) Confirmed reasonable with producer interviews.
	DDGS, dry	n/a	n/a	Corn byproducts, such as DDGS, are fed during growing and finishing (Pers. Communication, E. Richer, 10.12.15). DDGS are common protein feeds (Pers. Communication, M. Baker, 1.26.15). Confirmed reasonable with producer interviews.
	Mineral supplement	kg head^{-1} dy^{-1}	0.09	A salt and mineral supplement is fed to cattle to meet nutrient requirements (Pers. Communication, M. Baker, 1.26.15; Pers. Communication, B. Buchanan, 2.11.15) The supplement is assumed to be fed at the indicated rate (Ohio State University, 2014) and is assumed to be 67% dicalcium phosphate and 23% salt (*Dairy Reference Manual*, 1995)
	Water and feed provisioning			
Calving	Water source	n/a	Ground	Authors' assumption.
Starter	Electricity use, water	kWh	5.4	Water is pumped to meet calf requirements (including milk replacer

	pumping			hydration) and wash equipment. Producers estimated 88% of total water pumped was used to meet cattle requirements (n=3), which is used to upscale requirements to total use. The same procedure used to calculate the energy required for pumping in the GF system is used.
	Water source	n/a	Ground	Authors' assumption, which was confirmed by producer interviews.
	Gasoline use, milk replacer mixing	L	25	A small diesel engine (approx. 5 hp) and steel bulk tank would be commonly used to mix milk replacer (Pers. communication, D. Vermeire, 10.8.15). A 5 hp Honda engine burns 2.2 L of diesel hr^{-1} at rated power with the load from a generator (Pers. Communication, Honda US Headquarters, 11.12.15). Calves are fed twice per day, with each mixing and heating requiring 0.1 h (Pers. communication, D. Vermeire, 10.8.15). Total fuel use is estimated by multiplying the total hours mixing and heating by the fuel efficiency.
	Milk replacer heating	hr	11.5	The authors assume that a diesel generator is used to heat milk replacer. The total hours mixing and feeding is used with the ecoinvent 3.1 process "machine operation, diesel < 18.64 kW generators " which has duration as a reference flow. The inventory is modeled for the US.
Growing and Finishing	Electricity use, water pumping			See starter phase.
	Water source	n/a	Ground	Authors' assumption, which was confirmed by producer interviews.
	Diesel use, feed mixing	L	101	Operations use mixer-grinders and gravity bins of different ingredients on farm, including supplements (Pers. Communication, L. Geyer, 4.30.15). A mixer grinder grinds at a rate of 17 bu min^{-1} (Pers. Communication, H&S Manufacturing Company Incorporated, Marshfield, WI, 9.25.15). The total volume of feed to mix (except corn

				silage) was divided by the mixing rate to derive time required. Total fuel use for a 100 hp tractor (authors' assumption) was estimated according to Grisso et al. 2010.
	Diesel use, feed delivery	L	10,429	Cattle are fed twice daily from a feeder wagon pulled by a tractor (Pers. Communication, S. Boyles, 10.7.15). Each feeding is assumed to require 1 hr, yielding total duration of 620 hr^{-1} $phase^{-1}$ (2*310). Total fuel use for a 100 hp tractor (authors' assumption) was estimated according to Grisso et al. 2010.
	Feed transport to farms			
Starter	Milk replacer	km	450	Authors' assumption based on distances from large milk replacer manufacturers (NRV, Inc. and Grober Nutrition) to central New York. Central New York was chosen as a regional midpoint.
	Soybean meal	km	100	Soybean meal is used in a protein mix with DDGS, which is shipped 100 km to farms in the Adom et al. (2012) feed process set. The same distance is assumed here.
	Corn grain	km	100	Same distance as assumed for soybean meal.
	All other feeds	km	0	No additional transport added, as feed unit processes from Adom et al. (2012) were "at farm."
Growing and finishing	Corn grain	km	100	See above.
	Supplements	km	100	See above.
	All other feeds	t-km	0	See above.

3.3. Herd Performance Metrics

Note: Parameters listed are changes from "Dairy_system_assumptions" tab in the livestock model published by Peters and colleagues (2014).

Phase	Category	Units	Value	Explanation and Sources
	Reproduction and herd management			
Calving	Veal calves sent to market	Number cow^{-1}	0.212	Based on the available data, the number of veal calves was calculated as the ratio of mean annual calf slaughter (2010-2014) to the mean dairy cow inventory (Jan 1, 2011-2015) in the Northeast (USDA-NASS, 2015c, 2015d).
	Animal size			
Calving	Heifer at first calving	kg	601	Geometric mean value for surveyed region 1 farms (Thoma et al., 2010).
Starter	Weaned calf entering growing/finishing	kg	181	The mean value from a range of estimates (136 – 227 kg) by regional experts was used (Pers. Communication, M. Baker, 1.26.15, Pers. Communication, M. Broccoli, 1.28.15, Pers. Communication, B. Buchanan, 2.11.15, Pers. Communication, L. Geyer, 4.30.15). Producer interviews (n=2) confirmed this was reasonable.
	Time spent in each life phase			
Calving	Dry period	d	59	Geometric mean value for surveyed region 1 dairy farms (Thoma et al., 2010).
	Calving interval	d	409	Geometric mean value for surveyed region 1 farms (Thoma et al., 2010).
	Days in milk	d	350	Days in milk is calculated by subtracting dry period from the calving interval (Peters et al., 2014).
Starter	Calves on post-weaning diet	d	118	Calves may reasonably be on a starter operation for 175 d (Pers. Communication, L. Geyer, 4.30.15). The duration on a post-weaning diet is the difference between 175 and the 57 d weaning age (Peters et al., 2014).
Growing and finishing	Growing, heifers	d	125	Growing phase duration is assumed to match a two phase, stage 2 feeding program for Holstein steers,
	Growing, steers	d	125	

				as outlined by Grant et al. (1993).
	Finishing, heifers	d	185	The time in finishing is estimated by subtracting days in all other phases from a 485 d finishing age for dairy beef (Peters et al., 2014).
	Finishing, steers	d	185	
	Age at finish, heifers and steers	d	485	Consultation with regional experts revealed considerable variation in estimated finishing age (395 – 476 d) (Pers. Communication, M. Baker, 1.28.14, Pers. Communication, L. Geyer, 4.30.15). Therefore, the estimate is unchanged from Peters et al. (2014), which was confirmed reasonable with finishing time estimated during producer interviews (n=2).
	Mortality			
Calving	Cow mortality rate	%	6.1	Cow mortality is estimated as the ratio of reported mortality of cows and first-calf heifers to the total number of cows for region 1 farms, multiplied by 100 (Thoma et al., 2010).
	Replacement heifer mortality rate	%	1.5	Dairies in PA, NY, and VT have average replacement heifer mortality rates of 1.5% (USDA-APHIS, 2007).
Calving and starting	Preweaned replacement heifer calf mortality rate	%	7.0	Dairies in PA, NY, and VT have average pre-weaned replacement heifer mortality rates of 1.5% (USDA-APHIS, 2007).
Starting	Pre-weaned bull calf mortality rate	%	10.2	The authors used reported cattle inventories and sales from Thoma et al. 2010 for region 1 and the livestock model to attempt to estimate a bull calf mortality rate. A dairy cattle health expert deemed the resultant rate (13.4%) too high, but acknowledged that mortality rates of bull calves are likely higher compared to heifer calves (Pers. Communication, J. Lombard, 6.17.15). The authors assume a mortality rate that is the mean of pre-weaning heifers (7.0%) and the calculated rate (13.4%). This falls

				within a range estimated by a dairy beef expert (0-12%, at times up to 40%) (Pers. Communication, D. Vermeire, 10.8.15).

4. Table A3: Sensitivity Analysis Results

				GWP	Acidification	Acidification	Eutrophication	Eutrophication	Ag land use	Fossil depletion	Water depletion
System	Scenario	Value	Units	kg CO_2-eq. kg HCW^{-1}	moles H+ eq. kg HCW^{-1}	moles H+ eq. ha^{-1}	kg N kg HCW^{-1}	kg N ha^{-1}	m^2 kg HCW^{-1}	kg oil-eq. kg HCW^{-1}	m^3 kg HCW^{-1}
GF	Baseline	-	-	33.73	30.15	2480.10	0.44	35.96	121.57	1.10	0.07
GF	C sequestration	1502.00	kg CO_2 ha^{-1}	22.88	-	-	-	-	-	-	-
GF	Ym	5.50	%	30.77	-	-	-	-	-	-	-
GF	Harvest efficiency	35.00	%	33.79	30.20	1980.27	0.49	31.83	152.52	1.12	0.07
GF	Harvest efficiency	65.00	%	33.71	30.15	2873.93	0.41	39.25	104.90	1.09	0.07
DB	Baseline	-	-	12.69	13.12	7922.20	0.18	110.02	16.56	1.33	0.06
DB	Milk:beef econ allocation	See table 4	-	11.52	12.13	8177.74	0.17	112.55	14.83	1.24	0.06
DB	Feeds mass allocation	See table 4	-	12.99	13.57	7720.91	0.19	106.31	17.58	1.40	0.07
DB	DDGS:ethanol EI allocation	See table 4	-	12.54	12.86	7964.94	0.18	111.16	16.15	1.30	0.06

Chapter 3: Land use efficiency of beef systems in the Northeastern U.S. from a food supply perspective

Written in the style of *Agricultural Systems*

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Abstract

One widely recognized strategy to meet future food needs is reducing the amount of arable land used to produce livestock feed. Of all livestock products, beef is the largest land user per unit output. Whether beef production results in feed-food competition or a net positive contribution to the food supply, however, may depend largely on whether marginal land is used to grow forage. Van Zanten et al. (2015) developed the land use ratio (LUR) to identify livestock systems that produce more human food directly than would be produced by converting their associated feed land to food production – a perspective that is not addressed within LCA. While Van Zanten et al. (2015) used FAO country-level data, higher resolution crop suitability and yield estimation at the regional level is warranted. We present a method that integrates geospatial data for crop suitability and yield estimation at multiple scales into the LUR. We applied this approach for a grass-fed beef (GF) system and a dairy beef (DB) system in the Northeastern USA,

including multiple scenarios limiting land conversion. All systems had LURs greater than one, indicating they were inefficient land users from a food supply perspective. Because a large fraction of the forage land used in the GF system was suitable for crop production and moderately productive, its LUR was 3-6 times larger (less efficient) than the DB system. Future research should explore mechanisms to improve LUR and life cycle environmental burdens of these regional production systems.

Keywords: Land use ratio, beef production, food security

1. Introduction

Over one-third of global land is used for agriculture, the majority of which (75%) is dedicated to livestock (Foley et al., 2011). Of all livestock products, beef is the largest land user per unit output (de Vries and de Boer, 2010). Land occupation varies tremendously between beef production systems, however, largely due to differences in calf origin, cattle diets, and management system (e.g., upland grazing) (de Vries et al., 2015). Producing beef from dairy cattle (calves and culls) requires less land than suckler beef production systems, primarily because land use is allocated between milk and beef (de Vries et al., 2015). However, dairy beef calves are often fattened on large quantities of cereals and/or oilseeds (Mogensen et al., 2015; Nguyen et al., 2010; Stackhouse-Lawson et al., 2012; Tichenor et al., in review); feeding food crops to livestock instead of humans is an inefficient use of edible calories (West et al., 2014), with the magnitude of inefficiency dependent on the species On the other hand, beef systems that rely heavily on forages require a large land base (Mogensen et al., 2015; Tichenor et al., in review) but convert substantial quantities of human inedible products (e.g., grasses) into edible nutrients. For example, upland suckler beef production in the UK, which included small amounts of concentrates (e.g., maize or soymeal) produced more human edible protein than it consumed (Wilkinson, 2011). In the Upper Midwest USA, grass-finished beef returned 69.1% of the human edible energy it consumed – over an order of magnitude more efficient than a feedlot-based system in the same region (Pelletier et al., 2010). By addressing feed efficiency from a human food supply perspective, these analyses are more nuanced assessments of the role of livestock in future food security compared to solely focusing on land use. However, they do not address the opportunity costs of using land to produce feed instead of human food. To meet future food needs in a sustainable way, it is necessary to critically examine the current allocation of land to livestock (Foley et al., 2011; Garnett, 2009).

Evaluations of land use by livestock systems must consider both quantity and quality (Ridoutt et al., 2014). Van Zanten et al. (2015) developed the concept of land use ratio (LUR) to identify livestock systems that produce more human food directly than would be produced by converting their associated suitable feed land to food production. Systems that utilize byproducts and/or land unsuitable for producing human food directly, thus, can be efficient in terms of human protein production (van Zanten et al., 2015). For example, beef production systems that rely on marginal land may result in a net positive contribution to the food supply (de Vries et al., 2015; Eisler et al., 2014; van Zanten et al., 2015). However, roughly half of global pasture land is marginal, with the other half suitable for human food production (van Zanten et al., 2016). Whether the production of beef results in feed-food competition or a net positive contribution to the food supply may depend largely on whether marginal land is used for forage production.

To estimate the land use ratio (LUR) for two types of dairy systems in the Netherlands, Van Zanten et al. (2015) combined data on land suitability and yields at the country level for purchased feeds and at the farm level for grassland. Country-level data may be appropriate for the Netherlands, which has a small and relatively homogenous agricultural land base, and for imported feeds when the region of origin is unknown. However, this approach is less informative for production systems in large countries with diverse land uses, geographies, and climates, such as the USA. Even if such a country is suitable for production of a human food crops, its regions may have quite different capabilities. Additionally, farm level data is not always available, further suggesting a need for an intermediate approach between farm and country scale estimation.

We enhance the land use ratio (LUR) concept developed by Van Zanten et al. (2015) by incorporating geospatial analyses of land suitability and yield potential of food crop production on different land cover types at multiple scales (e.g., field-scale, regional). We illustrate this approach with two case studies in the Northeast USA: a management-intensive grazing (MiG) grass-fed beef system, and a confinement dairy beef system. Additionally, we expand the system boundary for dairy beef to include milk production to estimate results for the whole dairy and beef system.

2. Methods

2.1. Existing and Enhanced Land Use Ratio

The land use ratio (LUR) measures the land-use efficiency of human protein production of livestock systems from a food supply perspective (van Zanten et al., 2015). The LUR is estimated using Equation 1:

$$LUR = \frac{\sum_{i=1}^{n}\sum_{j=1}^{m}(LO_{ij} \times HDP_j)}{HDP_a}$$

where LO_{ij} is the whole herd land requirement for the production of feed ingredient *i* (i=1,n) in country *j* (j=1,m), resulting in the production of one kg of animal source food (ASF), and HDP_j is the maximum amount of human-digestible protein that could be produced per year from conversion of suitable land to human food crop production in country *j* (van Zanten et al., 2015). This sum is divided by the HDP_a, the amount of human-digestible protein from one kg of ASF produced by the system. We enhanced the

LUR by including multiple sub-country scales of production (i.e., field and region scale) and assessing yield potential for food crop production on different land cover types at those scales using publicly-available geospatial data in the USA. Our enhanced LUR is estimated using Equation 2:

$$LUR = \frac{\sum_{i=1}^{n}\sum_{j=1}^{m}\sum_{k=1}^{p}(LO_{ijk} \times HDP_{jk})}{HDP_a}$$

where all variables an indices are as in Eq. 1, except *k* (k=1,p), which indicates the livestock feed land requirement at the sub-country scale for country *j*. As in Van Zanten et al. (2015), the enhanced LUR is computed in four major steps, which are described sequentially in the following sections.

2.1.1. Quantifying Land Requirements of Feed Production

We used data from a recent life cycle assessment of Northeast grass-fed beef (GF) and dairy beef (DB) as the basis of the production systems (Tichenor et al., in review). We defined the Northeast region in accordance with the U.S. Department of Agriculture's National Institute of Food and Agriculture.[3]

The GF system is based on a 30 cow herd, which produces approximately 24 market-weight steers and heifers per breeding cycle (Table 3.1). Producers practice management-intensive grazing (MiG), moving cattle between paddocks 0.4 – 6 times per day. During the grazing season, herd feed requirements are met with grass-legume pasture, milk from

[3] The Northeast includes the following states: Connecticut, Delaware, Maine, Maryland, Massachusetts, New Hampshire, New Jersey, New York, Pennsylvania, Rhode Island, Vermont, and West Virginia (USDA-NIFA, 2012).

the dams, and a mineral mixture. During the winter, cattle are fed grass hay or grass-legume bale silage and a mineral mixture.

The DB system is a combination of two production systems: dairy production and finishing of dairy beef calves (Tables 3.2 and 2.3). The dairy system is a 328 cow herd, plus associated breeding bulls and replacement stock, which produces milk, culled cows and bulls, and surplus calves to be raised for either veal or dairy beef (Table 3.2). Cattle are fed a mixture of harvested forages (hay and silage), grains, oilseeds, and associated coproducts, and a mineral mixture year round. Allocation between milk and beef was performed using a biophysical allocation equation developed for the USA (Thoma et al., 2013a), as described in (Tichenor et al., in review). Newborn calves destined for dairy beef are first sent to starter operations to be weaned and then shipped to grower/finisher operations to be raised to market weight on a high concentrate ration (Table 3.3).

Whole herd feed requirements and crop yields (Table 3.4) were used to calculate feed land occupation within the aforementioned LCA (Tichenor et al., in review). Land requirements for feeds were adjusted for harvest efficiency (pasture only) and losses during storage and feeding. Similar to Van Zanten et al. (2015), land requirements for co-products were allocated based on economic value (Tichenor et al., in review).

2.1.2. Estimating Suitability for Human Food Crop Production

We estimated the suitability of livestock feed land to produce the same five human food crops as Van Zanten et al. (2015), which included maize, soybeans, wheat, potatoes, and rice. For the Northeast region, food grains and oils (including soybeans) are the highest ranking categories of food consumption on a mass basis, while starchy vegetables are the highest ranking subcategory of vegetables (Griffin et al. 2014).

For the GF system, we collected field-level data from a sample of Northeast grass-fed beef producers practicing MiG (n=9). We mapped all forage land (i.e., grazed pasture, hay and bale silage) owned, leased or managed for the maintenance of their herds using the Google My Maps application (Google, 2015). We exported farm parcels from the My Maps application to ArcMap version 10.2 for geospatial analysis (Esri, 2014). To assess arability, we used the Gridded Soil Survey Geographic (gSSURGO) Database, which includes a 10m raster layer of the highest resolution classification of soils in the U.S. and their associated attributes (Soil Survey Staff, 2014a). We used the non-irrigated Land Capability Class (LCC) attribute data within gSSURGO to classify each cell within the farm parcels as either arable (LCC 1 though 4) or non-arable (LCC 5 – 7) (Soil Survey Staff, 2014a; USDA-SCS, 1961). For forages purchased beyond the boundaries of the farms, we used a similar method to estimate arability of soils at the regional level. We used the 2014 Cropland Data Layer (CDL), a raster agricultural land cover dataset, to spatially classify land used for hay and pasture production (hereafter, hay/pasture) (USDA-NASS, 2014). We again used the non-irrigated LCC dataset at the regional level to classify hay/pasture land as arable or non-arable.

For the DB system, field-level data was not available. However, we assumed all forages and most maize (except the fraction used in starter feed, which is purchased from companies) in the ration were produced regionally. We assumed dairies sourced maize regionally as the Northeast is 94 percent self-reliant in the production of livestock feed energy, and maize is a major energy feed produced in the region (Conrad et al., 2016). As with regional hay/pasture, we used the CDL to spatially identify regional cultivated cropland and the LCC to classify cropland as arable or non-arable. Cultivated cropland includes alfalfa, which is commonly planted in rotation with cultivated crops (e.g., maize silage) (Cornell University, 2016) and likely shares the same land base.

The LCC indicates whether groups of soils with similar management limitations can be used for arable agriculture, not whether those soils are suitable for production of a particular crop (USDA-SCS, 1961). Suitability for production of individual food crops is addressed by productivity indices, as described in the following section. Productivity indices are only applied to land categorized as arable according to the LCC. We assume no suitability for rice at the field or regional scale, because it is a sub-tropical or tropical crop and thus could not be grown under normal circumstances in the temperate Northeast (Samanta et al., 2011).

Two of the coproduct feeds in the DB system, soybean meal and distillers dried grains with solubles (DDGS), were globally traded commodities. The USA is the top global producer and exporter of soybeans, with less than one percent of the domestic average annual supply (2000-2010) of soybean and soybean meal imported (USDA, Economic Research Service (ERS), 2015; USDA-ERS, 2012). As such, we assumed all soybean meal was produced domestically. Similarly, the USA is the top global producer of maize, which is used in ethanol production, the primary source of the co-product DDGS within the country (USDA-ERS, 2015d, 2015e). Less than one percent of the total domestic average annual supply of DDGS (2000-2010) is imported (USDA-ERS, 2015d). Thus, we assume all DDGS were produced domestically. We used the country level suitability assessment from Van Zanten et al. (2015) for the land used to grow these feeds.

2.1.3. Estimating Human-digestible Protein Production from All Suitable Land

We followed the approach of Van Zanten et al. (2015) to estimate protein production from suitable land, multiplying food crop output by protein content and adjusting for human digestibility (Table 3.5). We also calculated results on an energy basis for an additional perspective, as did Van Zanten et al. (2015). For each feed crop type (e.g., maize silage), we chose the food crop producing the maximum amount of human digestible protein or energy to include in the LUR calculation. To estimate food crop yields, we used a tiered approach at the field, regional and national scales. At the field scale (GF only), we used the National Commodity Crop Productivity Index (NCCPI) version 2.0 within gSSURGO to estimate maize, soybean, and wheat yields on arable land used for perennial forage production (Soil Survey Staff, 2014b). The NCCPI corn and soybeans (hereafter, corn/soy) module and small grains (i.e., wheat, oats, barley and rye) module estimate non-irrigated yields as a function of soil, climate and landscape factors (USDA-NRCS, 2012). We calculated weighted average NCCPI corn/soybean and small grains values on arable pasture and arable hay/bale silage land within ArcMap, and then used those values to adjust the following maximum yields: 15,063 kg ha^{-1} maize, 4,705 kg ha^{-1} soybeans, and 8,065 kg ha^{-1} wheat (Dobos, Personal communication). A detailed example of this process and the previous step of calculating suitability for human food production is provided in Figure 3.1. For potato, we developed an alternative approach, as no productivity index was available (see below).

At the regional scale, we followed a similar procedure, calculating weighted average NCCPI corn/soybean and small grains values on arable hay/pasture land and cultivated cropland. For potatoes, we did not have a similar index to apply at the field and region scales. Instead, we used region average yields (2000-2010), weighted by land area from the data of Griffin et al. (2014) as a proxy for yields on all cultivated cropland. We assumed potato yields on regional hay/pasture land were 15.4 percent lower than on cultivated cropland, based on the proportional difference between overall NCCPI values on arable cultivated cropland and hay/pasture land. At the country level, we estimated U.S. average yields (2000-2010) for all five food crops using the U.S. Department of Agriculture's National Agricultural Statistics Service (NASS) annual survey data (USDA-NASS, 2011).

2.1.4. Estimating Human Digestible Protein of Animal Source Food

For GF and DB we use conversion factors of 0.35 edible weight per liveweight accounting for decreased carcass conversion of grass-fed, Holstein breed, and culled cattle within these estimates (Duckett et al., 2013; Neel et al., 2007; Scaglia et al., 2012; Stackhouse-Lawson et al., 2012). We adopted the nutrient conversion data from raw ASF output to human digestible protein and energy of Van Zanten et al. (2015), with the exception of milk and white potatoes (Table 3.5).

2.2. Scenario Analysis

Cropping is possible on arable land but not necessarily advisable from a conservation or potential profitability perspective. Clearing and cropping woodland pasture, for example, may have a high economic opportunity cost, carbon emission, and biodiversity impact. Additionally, economic viability of cropping on pasture may be limited by low yields. At the regional scale, it might be prudent to retain part of the land base in perennial crops to promote agricultural resilience and sustainability; however, the minimum quantity of land required for this purpose is unknown. Fully addressing these tradeoffs is complex and beyond the scope of this analysis. However, to explore how such considerations may impact our results, we developed "risk averse" (RA) scenarios for field and region-scale hay/pasture land used in the production systems. For the GF system, we first reclassified all arable land currently managed as woodland pasture on farms as non-arable (RA1). For both systems, we used NCCPI statistics on cultivated cropland at the region-scale to define thresholds that limited the conversion of all hay/pasture land cover based on potential productivity. These thresholds were the mean NCCPI corn/soy and small grains values on regional cultivated cropland minus 2, 1, and 0.5 standard deviations (RA2 – 4). All other steps in the LUR calculation were the same for these scenarios, as outlined in the previous sections.

3. Results

3.1. Land Use of the GF System

GF operations managed 603 ha of forage land. The majority of this land was grazed grass-legume pasture (80%), with the remainder managed for harvested forage (13%) or woodland pasture (7%). Most land was used either for grazing or harvesting forage, not both (≈95%). Eighteen percent of the harvested forages fed on GF operations were produced on the farm.

3.2. Land Suitability and Yield

The majority of forage land at the field and region scales is arable (Table 3.6). Estimated yields on pasture and hay/bale silage land at the field and regional scale were similar. For example, when field scale pasture and hay/bale silage land were combined, the weighted average NCCPI corn/soy was 0.39 (results not shown) compared to 0.41 at the region scale. Compared to national averages, estimated food crop yields on regional cropland were lower for all crops except wheat. Unsurprisingly, estimated food crop yields on arable regional hay/pasture land were lower than on regional cultivated cropland. However, the difference was not that large, ranging from 12% lower for wheat to 16% lower for maize and soy.

3.3. Land Use Ratio

A livestock system with a LUR less than one provides more protein or energy than would be produced from converting its suitable land base to food crops. The baseline LURs on human digestible protein (HDP) and energy (HDE) bases for both beef systems were much greater than 1 (Figure 3.2). For the GF system, the high arable fraction of forage land and moderate estimated crop productivity resulted in a large LUR. The DB system used 86% less land per unit output compared to GF, partially explaining why the LUR was much lower despite its high reliance on cultivated cropland. The protein based LUR of GF and DB systems means that converting their arable feed land bases to food crops (in this case, soybeans) could yield 52.9 and 9.2 times more human digestible protein than is currently produced. Overall, the protein based LUR of the GF system was about six and sixteen times greater than the DB and DB plus milk systems at baseline, respectively.

Expanding the system boundary to include milk production from the dairy calving system resulted in a lower LUR compared to DB, though it was still greater than 1 (Figure 3.2). Although 55% of the land required for the dairy system was cultivated cropland, high milk productivity and HDP output partially compensated for this. Adding milk to the DB system resulted in a more than a 17-fold increase in HDP output with only seven times more land. We also estimated the LUR of the dairy system alone, using same system boundaries and edible product conversions as Van Zanten et al. (2015) for comparison purposes. On a protein basis, the LUR of the dairy system at baseline was 3.26.

None of the RA scenarios resulted in a LUR less than one (Figure 3.2). For GF, removing woodland pasture from the arable land base at the farm scale (RA1) reduced the arable fraction to 0.83, decreasing the LUR by 5% (HDP and HDE bases). In each scenario, the GF system had the largest LUR, followed by DB, then DB plus milk. However, the magnitude of the differences between GF and the dairy based systems decreased with each additional scenario. Limiting the fraction of forage land that could be converted based on productivity potential (RA 2 - 4) had a more significant influence on the GF system, due to its complete reliance on forages. Major reductions in the LUR only occurred in RA 3 and 4 for the GF system, where pasture and hay land conversion was limited to land with productivity greater than or equal to the mean cropland productivity, minus 1or 0.5 standard deviations. This illustrates substantial overlap between the distributions of potential food crop productivity of pasture/hay land and cultivated cropland at the regional level. Although the LUR of the GF system was 53% lower than baseline in RA4, the differences between the production systems remained large. The protein-based LUR of the GF system, for example, was approximately three and eight times larger than the DB and DB plus milk systems, respectively.

It bears mentioning how difficult it might be for beef systems to have a LUR < 1 in this region. While the small reduction in the land use ratio of the GF system in RA1 is partially a function of the small fraction of total pasture as woodland, pasture accounts for only 59% of the land requirements of the GF system. If all grazed pasture in the GF system was assumed to be unsuitable woodland pasture, the LUR for this scenario would still be 19.9 due to the quantity of cut forages fed (results not shown). If all on-farm forage land and off-farm hay land in the GF system was considered unsuitable, and if off-farm bale silage land (2.3 ha) was subject to the restrictions in RA4, this would produce a protein-based LUR < 1 (0.99) (results not shown). Thus, the GF system would have a LUR < 1 if 4% or less of its forage land base was suitable for human food crop production. For DB, the high use of cultivated cropland means that even if all forage land was considered unsuitable, the LUR on a protein basis would still be 6.7 (results not shown).

4. Discussion

Our enhanced land use ratio method simulates potential human food crop productivity at multiple sub-country scales, on multiple land cover types. We illustrate a higher resolution approach to suitability and yield estimation than was possible with the data used by Van Zanten et al. (2015). For land used to grow feed off-farm, Van Zanten et al. (2015) used the UN Food and Agricultural Organization's Global Agroecological Zone (GAEZ) suitability index on cultivated cropland at the country-level and country average yields to estimate potential human food crop production. Assuming countries with suitability ratings of > 55 (good or better) are suitable for the production of a crop might underestimate suitability, whereas applying country average yields may overestimate yield potential. The net balance of these uncertainties is unclear, however. For the dairy systems of Van Zanten et al. (2015), the only grassland used was on the farms, with suitability and potential food crop yield determined by broad soil type (sand versus peat). In our case, we sought to estimate suitability and yield potential of cropland and grassland at one or more sub-country scales. Using a productivity index made this possible and increased the specificity of food crop yield estimates. Given the latter, as well as differences found between national and estimated regional food crop yields on cropland and grassland, our approach increased the accuracy of estimating the LUR for ruminant systems in the USA. While the datasets we used were specific to the USA, agricultural land cover data and productivity indices are likely available in other countries, making this approach broadly applicable.

Due to the novel nature of the LUR, limited data exist for comparison. The LUR has only previously been applied to Dutch egg and dairy production systems (van Zanten et al., 2015). The baseline LUR on a protein basis of the dairy system in this study is 55% greater than Dutch dairy production on sandy soils (LUR: 2.10) (van Zanten et al., 2015). Although it is impossible to identify which factors account for these differences due to varying methodologies between the two analyses, differences in feeds may provide a partial explanation. Grassland, for example, provided 39% of energy intake for the Dutch dairy system (van Zanten et al., 2015) versus 21% in the Northeast USA dairy system. Due to lower grass yields, three times as much land would be required in the Northeast region to produce an equivalent amount of grass compared to the Netherlands (9,544 kg DM ha^{-1}) (van Zanten et al., 2015). Although Dutch grassland was only suitable to grow white potatoes and wheat, the maximum HDP yield per hectare (conversion to white potatoes) was 33% higher than that of Northeast U.S. regional grassland (conversion to soybeans). To produce equivalent amounts of grass for dairy production, therefore, the forgone HDP potential from crop production was 2.3 times greater in the Northeast versus the Dutch dairy system. Furthermore, rations in the Dutch dairy system included significant quantities of crop residues or byproducts with no associated land use (e.g., citrus pulp), which do not add to the numerator of the LUR (van Zanten et al., 2015).

A striking finding was that most hay and pasture land at the field (GF only) and region scale is arable with only modestly lower potential for select food crop production than cultivated cropland. Historical land use change and agricultural production dynamics may provide insight into why this is the case. The agricultural land base in the Northeast declined by 60% between 1929 and 2012, driven by the relocation of production due to comparative advantage in other regions and improvements in productivity (Conrad et al., 2016). Additionally, urban land use has been increasing, and now accounts for a disproportionately larger fraction of regional land compared to other U.S. regions (Conrad et al., 2016). As such, land that was previously agricultural has largely been reforested or developed (Conrad et al., 2016). When agricultural land use changes, lower quality lands are more likely to shift to less intensive uses, such as forestry (Lubowski et al., 2006). As such, it is logical that the remaining agricultural land after a period of marked contraction due to market forces is generally of relatively high quality. While land quality is an important factor in the allocation of land use, however, it is insufficient to fully describe these complex dynamics (Lubowski et al., 2006). For example, for products like forages, which have a low economic value relative to their weight, distance to markets is an important profitability consideration (Lubowski et al., 2006). The Northeast has long been a dairy production center, which requires large amounts of high quality forages (Conrad et al., 2016). Whether the forage demand of the industry shaped land use patterns or the quality of the land base encouraged dairying (or both) is unclear.

We have demonstrated that even when beef production is completely reliant on forages, there may be a significant opportunity cost of land use regarding human food production. However, our findings are specific to these case studies in the Northeastern USA. Raising grass-fed beef on arid rangeland in the Western U.S. likely has a starkly different LUR, which is an area for future research. Furthermore, as was mentioned earlier, although food production is technically possible on regional hay/pasture land, it is not necessarily economically feasible. We addressed one aspect of economic feasibility by limiting the converted land base based on productivity potential in the RA scenarios. However, farmers allocate land to uses they believe will result in the greatest benefit over time, estimating expected returns to land as a function of, for example, output value, input costs, current policies, land quality, skills, and personal preferences (Lubowski et al., 2006). In New York, Peters et al. (2012) estimated low and negative weighted average land use values for the production of grains and meat, respectively. The land use values, which were calculated assuming average quality land and conventional production systems, were also found to be highly sensitive to yield changes (Peters et al., 2012). Producing a high-value, niche product like grass-fed beef, therefore, may be an attempt to establish a business model that generates positive returns on less productive land in the region.

Significant opportunities may exist to reduce the LURs of these systems. Given the substantial fraction of system footprints across categories attributed to feed production (Tichenor et al., in review), substituting food waste or crop byproducts for feeds may be a promising strategy to reduce burdens and LUR. There may also be policy momentum for this strategy in the region, as landfill bans for organic waste are increasingly being adopted. Thus far, three Northeast states (Massachusetts, Connecticut, and Vermont) and New York City have banned commercial entities from landfilling organic waste (Edwards et al., 2015). Future research should explore the net benefits of incorporating food waste into these regional systems. In addition to process improvement, examining these systems with a more holistic lens may reduce their LURs, particularly for the GF system. Ecosystem services, such as carbon sequestration, erosion regulation, and cultural value, provided by the maintenance of grasslands (Franzluebbers et al., 2012; Ripoll-Bosch et al., 2013) are not currently accounted for in the LUR. Partial accounting for this multi-functionality has produced starkly lower life cycle burdens compared to only considering marketed products in Spanish case studies (Ripoll-Bosch et al., 2013). Similar results could be true for the LUR, which merits further research.

Achieving a LUR < 1 is a high threshold to set for livestock systems. In fact, the crop(s) that maximize HDP per hectare within the LUR may provide more protein per unit land than many other plant-based proteins. For example, soybeans grown in the Northeast are very high yielding and protein-dense (Tables 5 & 6). Compared to soybeans, kidney beans grown in the region have 38% lower yields and 33% lower protein concentration (DM basis) (Griffin et al., 2014; USDA-ARS, 2015). As a result, kidney beans provide 59% less protein than soybeans per unit land. By choosing the crop(s) that maximize HDP per hectare, the LUR illustrates how far away current land allocation within a livestock system is from maximizing protein production. While this is largely a theoretical objective, it is a helpful benchmark given global food security and sustainability challenges.

While the enhanced LUR is an important metric to understand land use efficiency from a food supply perspective, there are limitations to this method. Our suitability and productivity estimates rely on national soil survey data that is continuously updated on a project basis, though the currency of full soil surveys at the county-level varies tremendously (Soil Survey Staff, 2014a; USDA-NRCS, 2016). Although the NCCPI accounts for many soil attributes that could impair productivity, such as erosion class, actual erosion could vary from site to site due to the nature of the data. In many cases, it is uncertain how much erosion has occurred, since erosion is estimated using general class ranges at a point in time and because the actual starting condition of the soil surface is uncertain (Pers. Communication, S. Finn, 5.19.16). That being said, this dataset is the highest resolution classification of soils in the USA, and thus, was best possible option for our analysis.

Additional limitations relate to the nutrient based functional unit used (van Zanten et al., 2015). Adjusting protein content for digestibility is necessary but insufficient to assess protein quality, which is also a function of essential amino acid content and bioavailability (Gilani et al., 2005). Digestibility values were derived using the fecal balance method, which may overestimate digestibility compared to ileal methods (Gilani et al., 2005). However, little ileal digestibility data currently exists (Ghosh, 2016), making this an area for future innovation with the LUR (Gilani et al., 2005). Additionally, the single nutrient approach does not account for the fact that beef provides other essential micronutrients such as iron (van Zanten et al., 2015), nor does it address differences in the fatty acid profiles between beef and plant-based proteins. To account for multiple nutrients would require an index (van Zanten et al., 2015), which may be more appropriately applied at the diet level to assess overall healthfulness of consumption. Furthermore, ChooseMyPlate.gov, the consumer education website for the U.S Dietary Guidelines classifies beef as a "protein food," which supports using a protein basis for this analysis (USDA, 2016).

5. Conclusion

The enhanced LUR provides a high-resolution approach to estimating the opportunity costs of land used in livestock systems for human food production. The case study systems, MiG grass-fed and confinement dairy beef production in the Northeastern U.S. have LUR > 1, meaning they produce less protein and energy than conversion of their land base to food cropping would. However, the LUR does not consider the ecosystem services provided by regional grasslands, which are likely important both from a conservation and social value standpoint. At the very least, results of the LUR provide additional clarity on the tradeoffs regarding different regional beef production systems.

Furthermore, coupling LUR and LCA results highlights leverage points to reduce environmental burdens and enhance the food supply. This knowledge can inform an ongoing dialogue about increasing self-reliance of food production in the Northeastern U.S. and will hopefully catalyze conversations about how these systems fit within the global context of sustainably meeting future food needs.

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Table 3.1 Technical data for Northeast grass-fed beef system

Techincal parameter[a]	**Value**
Breeding stock	
Number of mature cows	30
Number of mature bulls	1
Number of replacement heifers and bulls	7
Calving percentage (%)	97.7
Cull rates (%)	
Breeding cows	9.7
Breeding bulls	20.0
Mortality rates (%)	
Mature cows and bulls	1.5
Replacement heifers and bulls	2.1
Pre-weaned calves	3.6
Growing and finishing cattle	0.5
Length of grazing season (days)	200
Birth to market weight (days)	
Steers	712
Heifers	682
Market weights (kg)	
Steers	544
Heifers	478
Beef production (kg liveweight/year)	
Market steers and heifers	12,463
Culled cattle	1,822
Feed intake parameters (kg DM)	
Grass-legume pasture[b]	142,887
Grass hay	103,781
Grass-legume bale silage[b]	15,874
Mineral mix[c]	1,440

[a] Source: Tichenor et al. (2016)

[b] 80:20 grass-legume on a dry matter basis.

[c] No associated agricultural land use.

Table 3.2 Technical data for Northeast dairy system, system boundary at the dairy farm gate

Techincal parameter[a]	**Value**
Breeding stock	
Number of cows	328
Number of mature bulls	1
Number of replacement heifers and bulls	232
Calving percentage (%)	86
Cull rates (%)	
Breeding cows	28.2
Breeding bulls	20.0
Mortality rates (%)	
Mature cows and bulls	6.1
Replacement heifers and bulls	1.5
Pre-weaned replacement calves	7.0
Milk production	
Milk production per cow (kg/year)	10,732
Total milk production per farm (kg/year)	3,522,404
Beef production (kg liveweight/year)[b]	
Calves (veal and dairy beef)	6,742
Culled cattle	60,398
Feed intake parameters (kg DM)	
Maize silage	1,328,928
Grass-legume hay	842,964
Alfalfa haylage	596,284
Shelled maize	382,969
Soybean meal	212,468
DDGS, dry	53,117
Milk replacer[c]	4,261
Mineral mix[d]	123,283

[a] Source: Tichenor et al. (2016).
[b] Includes all outputs of the dairy. Culled cattle and 53% of calves enter the dairy beef system, and thus are included in Table 3.3.
[c] Includes land requirements for producing dry milk for milk replacer.
[d] No associated agricultural land use.

Table 3.3 Technical data for Northeast dairy beef production system, system boundary at the growing/finishing gate

Techincal parameter[a]	Value
Calves from dairy system	86
Biophysical allocation beef (%)	9.4
Mortality rates (%)	
Pre-weaned calves	9.4
Growing and finishing cattle	1.1
Birth to market weight (days)	485
Market weights (kg)	
Steers	602
Heifers	529
Beef production (kg liveweight/year)	
Market steers and heifers	44,523
Culled cattle	60,398
Feed intake parameters (kg DM)	
Shelled maize	122,134
Maize silage	51,457
DDGS, dry	26,905
Mostly-mixed grass hay	16,278
Soybean meal	4,240
Milk replacer[b]	2,935
Mineral mix[d]	2,705

[a] Source: Tichenor et al. (2016).
[b] Includes land requirements for producing dry milk for milk replacer.
[c] No associated agricultural land use.

Table 3.4 Feed crop yields at region and country scales 2000-2010 (USDA-NASS)

Crop and production scale	Feed crop yield (kg DM/ha)[a]
Northeast region	
Grass-legume pasture[b]	2,697
Grass-legume bale silage	4,641
Hay (mostly mixed grass; grass-legume)	3,166
Maize silage	12,166
Alfalfa haylage	5,966
Shelled maize	6,794
Country (USA)	
Shelled maize	8,204
Soybean meal	3,762
DDGS, dry	13,523

[a] Yields are adjusted for storage and feeding losses, according to Rotz 2014 (grains and oilseeds), Rayburn 2008 (hay and bale silage), and Behling 2014 (corn silage and alfalfa haylage). Economic value allocation is used to further adjust yields for soybean meal and DDGS, dry.
[b] Includes 50 percent harvest efficiency.

Table 3.5. Dry matter, energy and protein conversion factors for animal source food and food crops (Adapted from Van Zanten et al., Table 7)

Product	Product code	DM (kg DM/kg edible product)	HD energy (MJ/kg DM)	Protein (g/kg DM)	Protein digestibility (%)
Beef	13002	0.418	27.9	418.3	94
Milk[a]	01078	0.123	21.8	252.1	95
Maize	20014	0.896	17	105.1	85
Soybeans	16111	0.915	20.4	399	78
Wheat	200474	0.904	15.8	125.1	87
Potatoes white[b]	11354	0.184	15.7	91.2	78
Rice	20052	0.867	17.3	75	89

[a] Energy content in milk from USDA, ARS 2016 and protein content is from Thoma et al. 2010.
[b] Protein digestibility value from Kies and Fox, 1972.

Table 3.6. Suitability, productivity index values, and estimated yields for feed land

Feed land and production scale	Percent arable	NCCPI corn/soy	NCCPI small grains	Food crop yield on arable land (kg/ha)				
				Maize	Soybeans	Wheat	White potatoes	Rice
Field scale								
Grass-legume pasture	89.1	0.43	0.32	6,538	2,042	2,600	26,272	-
Hay and bale silage	89.1	0.34	0.31	5,062	1,581	2,486	26,272	-
Regional scale								
Pasture and hay[a]	83.4	0.42	0.36	6,333	1,978	2,878	26,272	-
Cultivated crops[b]	100.0	0.50	0.41	7,552	2,359	3,280	31,049	-
Country scale	100.0	-	-	9,271	2,729	2,814	43,294	7,582

[a] Pasture and hay land includes: grass-legume bale silage, grass-legume hay, mostly-mixed grass hay

[b] Cultivated cropland includes: maize silage, alfalfa haylage, shelled maize

Figure 3.1: Example calculation of potential HDE and HDP from feed land conversion to maize production at the field scale. Conversion factors used in HDP and HDE calculations are provided in Table 3.5.

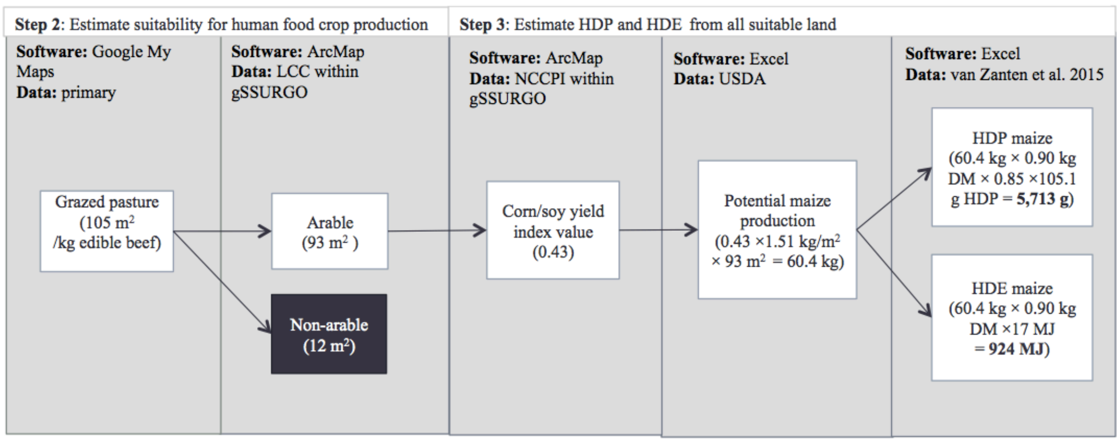
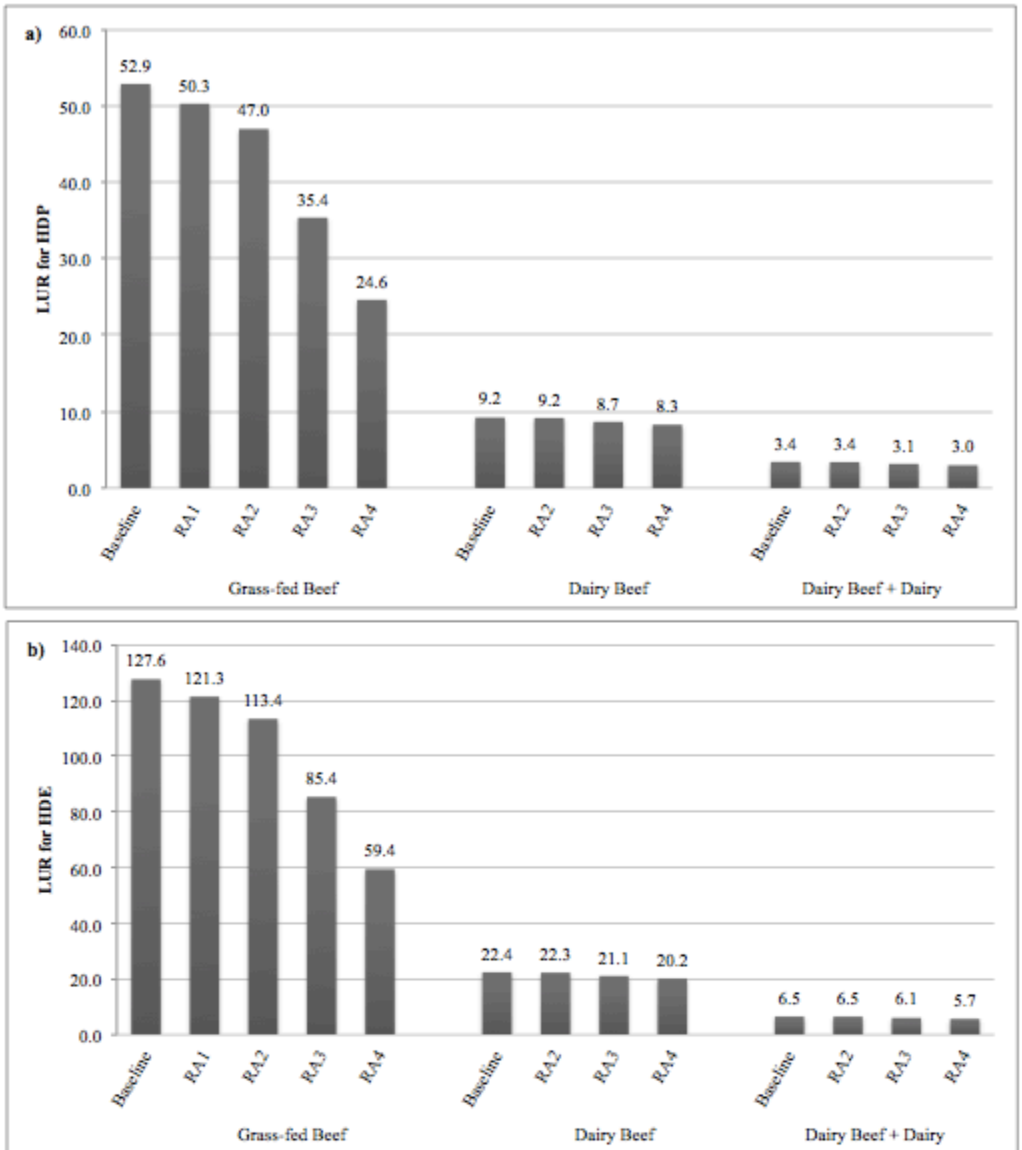


Figure 3.2: Land use ratios (LUR) on a) human digestible protein (HDP) and b) energy (HDE) bases for Northeast beef systems. Baseline and risk averse (RA) scenarios are shown.



Chapter 4: Feed or feedstock? Environmental and food supply consequences of feeding food waste to cattle in the Northeastern U.S.

Written in the style of *The Journal of Industrial Ecology*

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Abstract

Interest in local and regional meat production coupled with high population density provides a significant opportunity to repurpose food waste as livestock feed in the Northeastern U.S. However, food waste may also be used in waste-to-energy programs, providing cleaner power than the status quo. We used consequential life cycle assessment (C-LCA) to assess the net environmental and food supply impacts of substituting food waste for corn in Northeast dairy beef cattle (DB) rations, instead of using it for a substrate in anaerobic digestion. The impact scope was global warming potential, acidification potential, and the land use ratio (LUR), which addresses feed-food competition. We accounted for the changes in processes and marginal products required as a result of the proposed shift. The functional unit was the amount of food waste that could reasonably replace corn in rations of growing and finishing DB cattle (71,492 kg dry matter), subject to several constraints. Converting this quantity of food waste to feed instead of using it as a substrate for anaerobic digestion reduced net global warming potential and acidification potential by 9,975 kg CO_2-eq and 48,785 moles H+-eq, respectively. While sensitivity analyses demonstrate the reduction in GHGs is less certain, acidification reductions were robust. Feeding food waste to dairy beef cattle also keeps food in the food supply, reducing feed-food competition by 11% as measured by the LUR. Regional policies to divert food waste from landfills ought to prioritize transformation to livestock feed to reduce ecological burdens and improve regional self-reliance of food.

Key words: beef production, waste management, consequential life cycle assessment

1. Introduction

In the Northeastern U.S., there is intense interest in local and regional meat production (Tichenor et al., 2016b, in review). As shown by Tichenor et al. (2016b, in review), dairy beef has several ecological advantages compared to grass-fed beef by leveraging the robust dairy sector in the region. However, dairy beef relies on significant quantities of corn-based feeds (i.e., grain, silage, and DDGS), which account for 21% of the global warming potential, 34% of acidification potential, 40% of fossil depletion, and 43% of agricultural land use of this system. Furthermore, despite requiring far less land than regional grass-fed or a conventional U.S. beef system, there is still a significant opportunity cost of human food production associated with Northeast dairy beef. The agricultural land required to feed dairy beef cattle could produce 9.2 times as much human digestible protein if it were converted to food crops. Incorporating food waste into dairy beef production systems may be a strategy to reduce environmental burdens and the opportunity cost of land use from a food supply perspective.

Significant opportunities may exist to feed food waste to livestock in the Northeast, a densely populated region with a shrinking agricultural land base (Conrad et al., 2016; Griffin et al., 2014). Approximately 65.6 million people reside in the region, with population projected to increase 3% by 2030 (Conrad et al., 2016). At the retail level alone, non-animal source foods (i.e., grains, fruit, vegetables, nuts, sweeteners) are lost at a per-capita rate of 36 kg yr^{-1} (Buzby et al., 2014), producing 1.1 billion kg yr^{-1} of potential feed for livestock at the regional scale. Connecticut, Massachusetts, Vermont, and New York City, NY have recently banned the landfilling of organic waste from commercial entities to varying degrees (Edwards et al., 2015). The U.S. Environmental

Protection Agency's Food Recovery Hierarchy prioritizes diverting food waste to animal feed over anaerobic digestion, indicating its superior benefits to ecosystems and society (US-EPA, 2016). At the same time, landfill bans and energy policy (e.g., The Clean Power Plan) encourage anaerobic digestion of organic wastes (Edwards et al., 2015), increasing competition for these materials. Most anaerobic digestion in the U.S. occurs on farms, particularly on dairies, where manure may co-digested with other feedstocks (Edwards et al., 2015). While the most prevalent use of source-separated food waste in the U.S. is aerobic composting, anaerobic digestion is becoming more common (Hodge et al., in review). For example, Stop and Shop, a New England supermarket chain owned by Ahold USA, Inc., the fourth largest grocery retailer in the U.S. (Martinez, 2007), opened an anaerobic digester in Massachusetts in spring of 2016 to generate electricity from its food waste (O'Connor, 2016).

There are two general approaches to life cycle assessment (LCA): attributional and consequential. Attributional LCA computes a life cycle inventory by "attributing" to specific products a portion of the pollution from, and resources into, multi-product systems. Consequential LCA attempts to directly model the consequences or system-wide changes induced by a decision or direct change. In practice, attributional LCA results are often used to infer the impact of shifting from one alternative to another (Plevin et al., 2014). This may produce misleading conclusions, however, due to the data used (i.e., average instead of marginal) and boundaries drawn within attributional LCA.

A C-LCA framework that addresses substitutions and knock on emissions is necessary to determine the optimal management of food waste. Very few life cycle assessments of food waste management have been conducted in the United States (Hodge et al., in review; Levis and Barlaz, 2011), all of which have used the attributional approach and did not include livestock feed as an option. Several European LCAs have assessed livestock feed as a potential food waste management strategy, with varying approaches, boundaries, and impact scopes (Eriksson et al., 2015; Salemdeeb et al., 2016; Styles et al., 2015; Tufvesson et al., 2013; Vandermeersch et al., 2014; van Zanten et al., 2014). However, none accounted for compensatory changes in rations that may occur when major feed substitutions are made. Additionally, a critical consideration for feeding food waste to livestock is the opportunity to reduce agricultural land use and feed-food competition. While a few European LCAs have assessed waste-to-feed impacts on land use (Vandermeersch et al., 2014; van Zanten et al., 2014), none have quantified how waste-to-feed strategies affect the efficiency of livestock systems' land use from a human food supply perspective. The objective of this study is to assess the net environmental and food supply impacts of substituting food waste for corn grain in Northeast dairy beef cattle rations, instead of using it for a substrate in anaerobic digestion.

2. Methods

The system boundary for this C-LCA included changes in food waste processing and utilization, as well as changes in the production and utilization of marginal replacement products when switching from anaerobic digestion to dairy beef cattle feed. We modeled food waste composition based on grocery byproduct feed (GBP), a high-moisture energy feed of fruit, vegetable, and bakery waste that is collected from grocery stores and processed on farms (Froetschel et al., 2014). The baseline cattle system was a dairy beef production system (DB) modeled in the Northeastern U.S. (Tichenor et al., 2016b, in review). The functional unit was the amount of source-separated food waste that could reasonably replace corn in rations of growing and finishing DB cattle, subject to the maximum inclusion rate [54% dry matter (DM)] recommended by Froetschel et al. (2014) and without changing the other feed types comprising the ration. With these constraints, we developed the reference flow within the livestock model (LM) (Tichenor et al., in review). GBP could replace 100% and 60% of the corn (DM basis) for growing and finishing DB cattle. The resulting reference flow was 71,492 kg DM of food waste (408,525 kg wet weight).

The impact scope included global warming potential, acidification potential, and the land use ratio (LUR) developed by Van Zanten et al. (2015). We included global warming potential because climate change is a primary concern in the adoption of alternative energy generation technologies. Acidification potential is also important because modeled anaerobic digestion of organic waste produced greater acidifying emissions than landfilling in the UK (Evangelisti et al., 2014) and the greatest acidifying emissions among multiple food waste treatment systems in the U.S. (Hodge et al., in review). Finally, we included LUR as an assessment of the impact of this change in terms of the human food supply. As the LUR is an attributional metric (i.e., the land use ratio of the beef production system), the system boundary for this calculation was the cradle-to-farm gate dairy beef production system (Tichenor et al., in review). Therefore, while the emissions analysis addressed the net impact of using food waste for dairy beef cattle feed instead of anaerobic digestion, the LUR analysis highlighted how this change influences the food supply opportunity cost of feed land use for dairy beef production.

2.1. Overall C-LCA calculation

We use the C-LCA framework for changing the application of a co-product of Van Zanten et al. (2014). The C-LCA is estimated according to the following equation, which is adapted from Van Zanten et al. (2014):

$$\text{Net Impacts} = -\,I1 + D1 - \Delta B1 + I2 + \Delta B2 - \Delta D2 \qquad (1)$$

where I1 is an avoided anaerobic digestion process; D1 is the production of marginal replacements for the products of anaerobic digestion; ΔB1 is the change in emissions from utilization of marginal products; I2 is the processing of food waste to feed to cattle; ΔB2 is the change in emissions from utilization of food waste by cattle; and ΔD2 is the change in cattle feed production resulting from feeding food waste. Each of these steps is detailed in subsequent sections, and the full system is depicted in Figure 4.1.

As most food waste in the U.S. is landfilled, it could be argued that the framework Van Zanten et al. (2014) highlighted for a byproduct becoming a product instead of a waste (Scenario 0) should instead be used here. However, we focus on commercial food waste generators, and as stated earlier, landfilling is no longer an option for many of these entities throughout the region due to state or municipal regulations. Furthermore, once separated, organic waste would be unlikely to be landfilled due to the cost of separation. Finally, our framing facilitates long-term decision support by analyzing the tradeoffs associated with two different applications of food waste.

2.1.1. Avoided anaerobic digestion (I1)

The anaerobic digestion process is based on a continuous, single-stage digester operating at mesophilic temperatures. This technology has been previously modeled for the digestion of food waste in the U.S and Canada (Hodge et al., in review; Levis and Barlaz, 2011; Sanscartier et al., 2012). We model the digestion of food waste (as GBP) primarily using parameters adapted from these studies (Table 4.1). GBP has a moisture content of 82.5%, which is higher than the food waste moisture content of Levis and Barlaz (2011) and Hodge et al (in review) (70% and 57-77%, respectively). However, the moisture content of GBP is well within that of existing food waste methane yield studies (47-95%) (Hodge et al, in review) and within a range of values for organic waste in a similar U.K. digestion study by Evangelisti et al. (2014) (65-85%).

We assumed that transport of food waste to the digester or to dairy beef operations was equidistant; that is, the collection transport burden was excluded at baseline. Although transport has a negligible impact on the burden of anaerobic digestion (Evangelisti et al., 2014), we ran sensitivity analyses regarding this assumption (Sec. 2.3).

Food waste digestion produces biogas and digestate, the latter of which may be used as a fertilizer. For the avoided digestion process, we accounted for emissions from fugitive biogas losses, biogas flaring, biogas combustion to power the digester, surplus biogas combustion for electricity, wastewater treatment of the liquid digestate fraction, and capital goods. Nitrous oxides (NO_x), sulfur dioxide (SO_2), and hydrogen chloride (HCl) emissions from biogas flaring and electricity generation via an internal combustion engine were from Hodge et al. (in review). After digestion, the digestate is dewatered, which reduces the cost of transport to farms (Levis and Barlaz, 2011; Sanscartier et al., 2012) but also results in the loss of fertilizer nutrients that are retained in the liquid digestate fraction (see Sec. 2.1.2). Emissions from the transport and treatment of the liquid digestate are from Levis and Barlaz (2011). We estimated the capital goods burden of running the digester using the global market for biowaste in Ecoinvent 3.1, following Hodge et al. (in review).

2.1.2. Production of marginal replacements for AD products (D1)

If food waste is used to feed cattle instead of as a substrate for digestion, marginal products must be produced to replace the avoided digestate fertilizer (*D1a)* and surplus electricity from biogas (*D1b)*. For D1a, we assume that the fertilizer nutrients in digestate (N, P, K) are replaced with the marginal synthetic fertilizers ammonium nitrate, triple superphosphate, and potassium chloride (De Vries et al., 2012; Styles et al., 2015). We assumed N, P, and K concentrations for food waste were equivalent to GBP according to Froetschel et al. (2014) (Table 4.1). We assumed that 28%, 33% and 18% of the incoming N, P, and K in food waste were retained in the solids fraction of the digestate, using the upper bound for a screw press dewaterer (Lukehurst et al., 2010). Screw press dewatering is the technology used at the Dufferin plant in Toronto, Canada (Gorrie, 2015), which was the foundation of the digestion models for Levis and Barlaz (2011), Sanscartier et al. (2012), and Hodge et al (in review). We estimated quantities of marginal fertilizer needed to replace digestate according to mineral fertilizer equivalents (MFE) (Hansen et al., 2006). We assume a MFE of 0.60, 1.0, and 1.0 for N, P, and K in digestate, respectively (Sanscartier et al., 2012). For example, 0.6 kg of ammonium nitrate-N is required to replace 1 kg of N in digestate. We used ammonium nitrate, triple superphosphate, and potassium chloride production processes from the Ecoinvent 3.1 database, long-term consequential system models for emissions from marginal fertilizer production (*Ecoinvent database*, 2013).

For avoided electricity from surplus biogas combustion (*D1b),* we assume electricity is replaced with a marginal grid mix for the Northeastern U.S. The Northeast includes two North American Electric Reliability Corporation (NERC) regions, NPCC (New England) and RFC (Mid-Atlantic) (Siler-Evans et al., 2012). We created a grid mix of 78% RFC and 22% NPCC based on their relative electricity generation totals in 2007, from Siler Evans et al. (2012). We replaced the total kWh produced from biogas combustion, net electricity needed for digester operations and losses, with this grid mix. We used the markets for high-voltage electricity from the respective sub-regions from the Ecoinvent 3.1. database, long-term consequential system models for emissions from marginal electricity production.

2.1.3. Change in emissions from utilization of marginal products (ΔB1)

Replacing digestate and surplus electricity with marginal products requires that any changes in the emissions of utilizing these products are also captured. For electricity (*ΔB1b)*, we assume no change in the emissions of using marginal electricity, following Van Zanten et al. (2014). For the change in emissions from using marginal fertilizers instead of digestate (*ΔB1a)*, we considered changes in product transport to the field and application emissions. We assumed both fertilizers and digestate were transported 20 km to the field, the digestate transport distance assumed by Hodge et al. (in review). Fertilizer application required 2.9, 2.3, and 1.6 L of diesel fuel ton^{-1} N, P and K, respectively (Hodge et al., in review). Digestate application required 2.7 L of diesel fuel dry ton^{-1} (Sanscartier et al., 2012). We estimated N_2O emissions from digestate and fertilizer application following IPCC (2006) Tier 1 protocols. Emissions factors for direct and indirect N_2O emissions and the leaching fraction ($Frac_{leach}$) were the same for both digestate and fertilizer. The fraction of applied N that was volatilized ($Frac_{gas}$) was 0.2 for digestate and 0.1 for synthetic fertilizer (IPCC, 2006). We assumed NH_3 emissions were the difference between volatile N loss and indirect N_2O emissions from N deposition, as in Tichenor et al. (in review).

2.1.4. Food waste processing pre-feeding (I2)

As was mentioned in section 2.1.1, we assumed the burden of transporting food waste to the dairy beef operation was equal to transport to a digester. Food waste is processed into GBP using a tractor powered grinder-mixer wagon on the farm and may require several grinds prior to feeding (Froetschel et al., 2014). We assumed food waste was ground twice in the feed grinder-mixer modeled for dairy beef (Tichenor et al., in review), but conducted a sensitivity analysis for this parameter. We approximated the change in the capital goods burden of the mixer-grinder using Ecoinvent 3.1 markets for agricultural machinery and four-wheel tractors.

2.1.5. Change in cattle feed utilization (ΔB2)

The DB system is based on a 328 cow dairy herd, with 53% of surplus calves sold to dairy beef starter operations per breeding cycle (Tichenor et al., in review). After being fed and weaned on starter operations, 78 calves are shipped to feedlots for growing and finishing (Tichenor et al., in review), the latter of which is the focus of this analysis (Table 4.2). We calculated the change in emissions and land use from substituting GBP for corn in growing and finishing rations using the LM. The LM was previously extended to calculate resource use and emissions, in addition to feed crop and land use of the entire DB system (Tichenor et al., in review). We extracted baseline results for the growing and finishing phases from that version of the LM (Tichenor et al., in review). We then changed the composition of the ration in the LM to include the reference flow for this analysis to simulate the corresponding feed crop and land use, enteric and manure CH_4, and manure N emissions (N_2O and NH_3). As in Tichenor et al. (in review), we estimated enteric CH_4 emissions as a function of feed intake and digestibility, following IPCC (2006) Tier II protocols. We assumed no change in enteric methane conversion factors (Y_m) for growing (5.5%) or finishing cattle (3.0%) (Tichenor et al., in review). Similarly, we followed the same procedure as Tichenor et al. (in review) to estimate greenhouse gas (CH_4 and N_2O) and NH_3 emissions from manure storage. We assumed all emissions factors (default or specific to the MMS), leaching and gaseous fractions were unchanged. Emissions are also a function of N excretion and volatile solids (VS) production, however, which were determined by the resulting ration composition from the LM.

2.1.6. Change in feed production (ΔD2)

We extracted feed and land requirements for DB growing and finishing cattle from the version of the LM parameterized for Tichenor et al. (in review). After changing the ration composition to include GBP, we quantified the changes in quantities and associated land requirements of corn grain, corn silage, and DDGS, dry for growing and finishing DB cattle. Less DDGS were needed, as they are a protein supplement, and GBP has a higher crude protein content than DDGS (Dairy One, 2011; Froetschel et al., 2014). For corn silage, the first step in LM ration formulation balances roughages and energy feeds to meet total digestible nutrient (TDN) requirements. The quantity of corn silage required in the ration slightly increased due to differences in TDN concentration between

GBP and shelled corn (89.8 vs. 88.2 % DM, respectively) (Dairy One, 2011; Froetschel et al., 2014). We assumed any required increase in corn silage was accommodated by shifting land that was previously used for corn grain in the ration to silage. Thus, no land use change (LUC) needed to be modeled. Unit processes for corn grain and silage production were adapted from Adom et al. (2012), as outlined in Tichenor et al. (in review). Similarly, the unit process for DDGS was adapted from Ecoinvent (v.2.2), rescaled, and subject to economic allocation using 2009-2013 prices, as described in Tichenor et al. (in review). Feed production processes did not initially include capital goods in the foreground. Therefore, we estimated their required capital goods using proxy feed production processes in Ecoinvent v 3.1 (i.e., maize grain production – US) and added their burdens accordingly.

2.2. Life cycle impact assessment

We used TRACI within the openLCA software (v. 1.4) to estimate per unit GWP (100 yr) and acidification impacts of relevant processes (e.g., diesel fuel use) (*openLCA*, 2015). We modified the GWP characterization factors for CH_4 and N_2O within TRACI to reflect IPCC (2007) recommendations. We used the modified TRACI characterization factors for GWP and acidification to also conduct LCIA for cattle utilization emissions. We modified the LUR Microsoft Excel (v. 14.6.5) workbook of Tichenor et al. (2016b) to calculate the change in the LUR of the beef system (*Microsoft Excel for Mac 2011*, 2011). We combined impacts from both sources and conducted the full LCIA in Microsoft Excel.

2.3. Life cycle interpretation

We tested the robustness of our results using parametric sensitivity analyses (Table 4.3). We constructed low and high efficiency scenarios for the anaerobic digestion process (AD-low and AD-high, respectively), following the approach of Hodge et al. (in review) and Sanscartier et al. (2012). Parameters modified in the scenarios included electricity generation efficiency and fugitive biogas emissions, which have been documented as having significant influence on the burdens of anaerobic digestion (Evangelisti et al., 2014; Hodge et al., in review; Levis and Barlaz, 2011). We also tested the sensitivity of the results with respect to the methane yield of food waste, following Hodge et al (in review) and Levis and Barlaz (2011). The marginal fuel source chosen can also significantly affect the ecological efficiency of anaerobic digestion (Bernstad and la Cour Jansen, 2012; Evangelisti et al., 2014; Hodge et al., in review). The electricity grid mix is quite different between two sub-regions in the Northeast, New England and the Mid-Atlantic (Siler-Evans et al., 2012). Thus, we simulated the impact of replacing electricity from the baseline digester with either the New England (NPCC) or Mid-Atlantic (RFC) mixes from the Ecoinvent 3.1 long-term consequential database.

On the cattle side, we tested the influence of three parameters. First, we accounted for potential differences in transport of food waste to farms versus anaerobic digestion. Hodge et al. (in review) indicated that doubling the distance and time from initial collection of waste to unloading increased fuel use from 5.3 to 9.7 L Mg^{-1} of waste. We used this difference (4.4 L Mg^{-1}) to simulate transport to farms from the collection point as 50 or 200% as fuel intensive as to a digester (TR-low and TR-high, respectively). Second, we varied the fuel use for grinding and mixing by 50% and 100% to account for the aforementioned uncertainty in the amount of grinding necessary for GBP (GR-low and GR-high, respectively). Finally, the concentrate fraction of the DB finishing ration reduced when adding GBP, due to increased corn silage. The concentrate fraction for steers dipped just below the 90% threshold (87%) and heifers were at the threshold (90%) that corresponds to the 3% IPCC methane conversion factor (Y_m). We therefore simulated a Ym of 4%, which is the upper bound for feedlot cattle on 90% or greater concentrates according to the IPCC (2006).

To provide context, we estimated normalized results for the proposed change. We adopted the normalization factors for the U.S. using TRACI by Lautier et al. (2010). Per capita annual normalization factors were 5,110 moles H+-eq. and 24,100 kg CO_2-eq. for acidification and global warming potential, respectively.

3. Results and Discussion

3.1. Global warming potential

Converting 71,492 kg DM of food waste into GBP instead of using it as a substrate for anaerobic digestion reduced global warming potential by 9,975 kg CO_2-eq (Figure 4.2). Avoiding anaerobic digestion (I1) reduced emissions by 15,703 kg CO_2-eq, 83% of which was due to avoided fugitive biogas and 16% of which was due to avoided capital goods. Replacing electricity from biogas with a marginal grid mix (D1a) was the largest contributor to emissions, responsible for 45,636 kg CO_2-eq. This reinforces results from attributional waste management LCAs of the importance of the electricity offset in the footprint of anaerobic digestion versus other waste management strategies (Evangelisti et al., 2014; Hodge et al., in review; Levis and Barlaz, 2011).

The increased burden from producing synthetic fertilizers (D1b) was more than offset by the emissions reduction from applying synthetic fertilizers instead of digestate (ΔB1a). This was due to several factors, primarily that less synthetic fertilizer needed to be produced and transported compared to digestate. Only 18-33% of the fertilizer nutrients in the digested GBP are retained in the solid digestate fraction, and the N in digestate has only 60% of the fertilizer value of a mineral fertilizer. For N, for example, this means that only 280 kg of synthetic N was needed (1,670 kg N digested x 28% retained in solid digestate x 60% MFE = 280 kg) to replace lost N from avoided digestate. This, in combination with the relatively high moisture content of solid digestate (74%) means that a much smaller quantity of synthetic fertilizer (1.61 Mg) must be transported to the field to replace digestate (138 Mg of digestate). Additionally, organic fertilizers have volatile N loss rates double those of synthetic according to IPCC, which results in higher indirect N_2O emissions.

For dairy beef, increases in feed processing for GBP on farms (I2) and cattle utilization emissions (ΔB2) had modest impacts compared to the change in feed production emissions (ΔD2). Avoiding the production of 59.5 and 15.0 Mg corn grain and DDGS, respectively, reduced emissions by 44,810 kg CO_2-eq. An additional 9.5 Mg of corn silage was required, which produced 2,383 kg CO_2-eq. emissions.

Limited data exist for comparing our results to the literature. Working in the Netherlands, Van Zanten et al. (2014) found that substituting a food processing waste (beet tails) for barley in dairy cattle rations instead of using it as a substrate for digestion resulted in a net emission of greenhouse gasses, excluding emissions from LUC. These results may diverge with ours due to multiple factors, such as: differences in marginal electricity mixes, GHG intensity of the marginal feeds, or methodological choices. On the other hand, Salemdeeb et al. (2016) found that converting municipal food waste into a wet pig feed produced lower GHG emissions than management with anaerobic digestion or composting. Although conducted in the UK with a different assumed livestock species, these results are in line with the findings of the present research.

3.2. Acidification

Changing the application of food waste to GBP also reduced acidification potential by 48,785 moles H+-eq. (Figure 2). Avoided anaerobic digestion (I1) reduced acidifying emissions by 13,171moles H+-eq., primarily due to avoided biogas combustion (89%) that releases SO_2, NO_x and HCl. Although producing marginal electricity (D1a) and fertilizer (D1b) (total: 19,334 moles H+-eq.) outweighed acidifying emissions savings from avoided anaerobic digestion (-13,171 moles H+-eq.), the change in digestate/fertilizer application emissions more than closed this gap. The majority of the change in application emissions was due to avoided field application of digestate (10,733 moles H+-eq.), due to aforementioned differences in fertilizer efficiency and higher volatile N losses for organic N application versus synthetic according to IPCC. The digestion side alone of eq (1), therefore, resulted in -2,100 moles H+-eq. emissions.

For dairy beef, changing feed production (ΔD2) resulted in the largest net impact of all categories. Within changing feed, avoided production of corn grain was responsible for majority (92%) of avoided acidifying emissions. This echoed the significant influence of avoided or induced feed production on environmental burdens found in European waste management LCAs that included waste-to-feed (Salemdeeb et al., 2016; Tufvesson et al., 2013).

3.3. Land use ratio

Compared to the baseline growing and finishing phases from the dairy beef system in Tichenor et al. (in review), substituting GBP for corn in rations required 41% less agricultural land (9.0 ha for the whole herd). Land used for corn grain and DDGS was reduced by 8.7 and 1.1 ha, respectively, while corn silage acreage increased by 0.74 ha. The change in the land area required at the same level of beef output resulted in an 11.4% reduction of the land use ratio (LUR) for human-digestible protein of the dairy beef system (from 9.2 to 8.2). This means that the GBP-based dairy growing and finishing system competes less with the human food supply, but it still competes, as the LUR is far > 1. The modest impact is largely due to the fact that growing and finishing only accounts for 25% of the total agricultural land use of dairy beef production (Tichenor et al., in review). As most of the land for dairy beef (62%) is for the maintenance of the breeding herd (Tichenor et al., in review), feeding GBP on dairies could potentially make a more significant impact both on the LUR of milk and meat. In fact, dairies may be a more feasible target for food waste-based feeds due to their proximity to cities and higher feed throughput compared to feedlots (Pers. Communication, M. Froetschel, 4.29.16). Future research should explore the net impacts and cost-effectiveness of feeding food waste feeds on regional dairy operations.

3.4. Life cycle interpretation

Full results of the sensitivity analyses are provided in Tables 4.4 and 4.5. For global warming potential, nine out of twelve scenarios resulted in net emissions reductions from feeding food waste to cattle instead of using it as a substrate for anaerobic digestion. Results were highly variable, however, and all but one scenario (feed grinding) resulted in a change of ± 55% from baseline. The low efficiency anaerobic digestion scenario further increased emissions savings by 159% compared to baseline (to -25,874 kg CO_2-eq.); whereas, the high efficiency scenario resulted in a net emission of 1,298 kg CO_2-eq. Implementing best management practices such as technologies to reduce fugitive biogas loss, therefore, may make it more advantageous from a climate perspective to digest food waste with a digester instead of with dairy beef cattle. Changing the methane yield of food waste by ± 1 SD from baseline resulted in a ± 136% change overall GWP. The high methane yield scenario resulted in an overall net emission of 3,549 kg CO_2-eq. Changing the marginal electricity source to either NPCC or RFC maintained a net emissions reduction for the proposed change. However, simulating NPCC electricity resulted in a net reduction of -38,947 kg CO_2-eq., an emissions savings 290% greater than baseline. Switching from feedstock to a GBP feed in New England versus the Mid-Atlantic may, therefore, create larger emissions savings. Simulating transport of food waste to farms as half or double that required to transport to a digester changed results by ± 56%, indicating that differential transport between the two options may have a significant influence on the magnitude of emissions savings. However, the high transport to farms scenario still resulted in overall emissions savings (-4,438 kg. CO_2-eq.).

The global warming potential result was highly sensitive to a one percentage point increase in the enteric methane conversion factor, resulting in an overall net emission of 234 kg CO_2-eq. Between the growing and finishing phases, the weighted-average Y_m at baseline was 4%, higher than average values predicted using mechanistic models for U. S. feedlot cattle by Kebreab et al. (2008). Increasing the finishing Y_m to 4% resulted in a weighted average Y_m of 4.6%, beyond the upper bound of the range for feedlot cattle (4.56%) given by Kebreab et al. (2008). Additionally, adding fat to rations is well known to reduce methane emissions (Kebreab et al., 2008), and GBP has a higher fat content than shelled corn (11.3 vs. 6.7% DM) (Dairy One, 2011; Froetschel et al., 2014). Thus, current evidence suggests that results produced by the high Y_m scenario are unlikely. However, further research is needed to explore these dynamics.

Results for acidification were far less sensitive than those for global warming potential. Each scenario resulted in a net emissions reduction from feeding food waste to cattle instead of to a digester, with only one scenario showing a change greater than ± 8% from baseline (NPCC). Simulating marginal electricity from New England only (NPCC), resulted in a total emissions reduction of 61,614 moles H+-eq. for the proposed change. Again, this suggests that switching food waste from digester substrate to dairy beef cattle feed in New England may provide even greater benefits than in the Mid-Atlantic.

Normalized LCIA results indicated that swapping GBP for corn in the rations of a small herd of dairy beef cattle instead of using it for anaerobic digestion could offset the annual acidification and global warming potential of 9.5 people and 0.41 people, respectively. All of New England and most of the Northeast is classified as an acid-sensitive ecosystem by the U.S. EPA (Burns, 2011). SO_2 and NO_x emissions, deposition, and their resulting ecological effects have improved dramatically in recent years as a result of increased EPA regulation and market-based programs (Burns, 2011). However, continued emissions and deposition reductions are required to restore sensitive ecosystems in the Northeastern U.S. (Burns, 2011). Implementing this change in the use of food waste from energy to feed source, therefore, may contribute to this goal.

4. Conclusion

In conclusion, changing the application of food waste from anaerobic digestion to a high-energy dairy beef cattle feed in the Northeast U.S. may reduce acidification and global warming potential. While sensitivity analyses demonstrate the reduction in GHGs is less certain, acidification reductions were robust to changes in multiple parameters. Additionally, feeding food waste to dairy beef cattle keeps food in the food supply, reducing feed-food competition by 11% as measured by the land use ratio (LUR). Our findings provide empirical support for feeding food waste to livestock as a preferred strategy to energy production according to the U.S. EPA's Food Waste Hierarchy. Regional policies to divert food waste from landfills ought to prioritize transformation to livestock feed in an effort to reduce ecological burdens and improve regional self-reliance of food.

Table 4.1. Baseline anaerobic digestion parameters

Parameter	Value
Food waste/GBP composition[1]	
Dry matter (%)	17.5
N (% DM)	2.3
P (% DM)	0.3
K (% DM)	1.7
Food waste methane yield (m^3 dry Mg^{-1})[2]	369
Biogas methane content (%)[3]	63
Biogas flared (%)[2]	5
Fugitive biogas loss (%)[2]	3
Methane lower heating value (MJ $m^{3\ -1}$)[2]	37.7
Electrical conversion efficiency (%)[2]	36.5
Digestate[4]	
Incoming food waste as solid digestate (%)	34
Dry matter (%)	26

[1] Froetschel et al. (2014). N is estimated assuming crude protein is 16% N.

[2] Hodge et al., in review.

[3] Evangelisti et al. (2014).

[4] Sanscartier et al. (2012).

Table 4.2. Northeast growing and finishing dairy beef herd characteristics

	Head	Days	Beginning and end weight (kg)	Feed intake, no GBP[1] (kg DM head^{-1} d^{-1})			Feed intake, with GBP (kg DM head^{-1} d^{-1})			
				Corn grain	Corn silage	DDGS[2]	Corn grain	Corn silage	DDGS	GBP
Growing heifers	19	125	181 - 315	0.4	4.5	1.2	0.0	4.6	1.2	0.4
Growing steers	59	125	181 - 348	0.8	4.4	1.3	0.0	4.6	1.1	0.8
Finishing heifers	19	185	315 - 529	6.7	0.3	1.0	2.9	0.8	0.0	4.3
Finishing steers	58	185	348 - 602	7.3	0.7	0.9	3.1	1.2	0.0	4.6

All values except feed intake with GBP are from Tichenor et al. (2016a)

[1] GBP, grocery byproduct feed.

[2] DDGS, dry distillers grains with solubles.

Table 4.3. Parameters used for scenario and sensitivity analyses

			Anaerobic digestion efficiency scenarios		Food waste methane yield		Marginal electricity		Food waste transport		GBP grinding		Enteric CH_4
Parameter	Units	Base	AD-low	AD-high	CH_4 - low	CH_4 - high	NPCC	RFC	TR-low	TR-high	GR-low	GR-high	Ym
Fugitive biogas loss[1]	% produced biogas	3.0	5.0	2.0	-	-	-	-	-	-	-	-	-
Electrical efficiency[2]	%	36.5	32.9	40.2	-	-	-	-	-	-	-	-	-
Methane yield of food waste[3]	m^3 dry Mg^{-1}	369	-	-	266	472	-	-	-	-	-	-	-
GWP, electricity[4]	kg CO_2-eq kWh^{-1}	0.66	-	-	-	-	0.24	0.78	-	-	-	-	-
Acidification, electricity[4]	moles H+-eq. kWh^{-1}	0.27	-	-	-	-	0.08	0.32	-	-	-	-	-
Diesel use, truck food waste[5]	L Mg^{-1} food waste	-	-	-	-	-	-	-	-4.4	4.4	-	-	-
Diesel use, GBP grinding[6]	L	608	-	-	-	-	-	-	-	-	253	1,317	-
Capital goods, grinder[7]	kg	36.1	-	-	-	-	-	-	-	-	15.1	78.3	-
Capital goods, tractor[7]	kg	3.4	-	-	-	-	-	-	-	-	0.4	9.5	-
Methane conversion factor[8]	% GE intake	3.0	-	-	-	-	-	-	-	-	-	-	4.0

[1] Sanscartier et al. 2012

[2] Hodge et al, in review.

[3] Mean methane yield ±1 SD from Hodge et al., in review.

[4] LCIA results from high voltage electricity markets in Ecoinvent 3.1. consequential long term system models. Baseline mix is 78% RFC and 22% NPCC, based on the generation fractions of Siler-Evans et al. 2012.

[5] Assumed using data of Hodge et al, in review.

[6] Fuel use estimated using grinder-mixer model from Chapter 1, this thesis.

[7] Capital goods burdens were estimated using global markets for agricultural tractors and agricultural machinery from Ecoinvent 3.1 cutoff system models.

[8] GE, gross energy. IPCC, 2006.

Table 4.4: Global warming potential sensitivity analysis results, in kg CO2-eq. emissions

Category	Baseline	AD -low	AD - high	CH4 -low	CH4 - high	NPCC	RFC	TR- low	TR- high	GR -low	GR- high	Ym -4%
I1: Avoided AD process												
Fugitive methane	-12,979	-21,632	-8,653	-9,346	-16,613	-12,979	-12,979	-12,979	-12,979	-12,979	-12,979	-12,979
Wastewater transport and treatment	-282	-282	-282	-282	-282	-282	-282	-282	-282	-282	-282	-282
Capital goods	-2,442	-2,442	-2,442	-2,442	-2,442	-2,442	-2,442	-2,442	-2,442	-2,442	-2,442	-2,442
B1a: Change in fertilizer application												
Avoided digestate field emissions	-3,120	-3,120	-3,120	-3,120	-3,120	-3,120	-3,120	-3,120	-3,120	-3,120	-3,120	-3,120
Avoided digestate application	-538	-538	-538	-538	-538	-538	-538	-538	-538	-538	-538	-538
Avoided digestate transport	-1,460	-1,460	-1,460	-1,460	-1,460	-1,460	-1,460	-1,460	-1,460	-1,460	-1,460	-1,460
Fertilizer field emissions	1,741	1,741	1,741	1,741	1,741	1,741	1,741	1,741	1,741	1,741	1,741	1,741
Fertilizer application	7	7	7	7	7	7	7	7	7	7	7	7
Fertilizer transport	17	17	17	17	17	17	17	17	17	17	17	17
D1a: Production of marginal fertilizers	2,906	2,906	2,906	2,906	2,906	2,906	2,906	2,906	2,906	2,906	2,906	2,906
D1b: Production of marginal electricity	45,636	38,390	52,583	28,478	62,794	16,664	53,766	45,636	45,636	45,636	45,636	45,636
I2: feed transport and processing	2,110	2,110	2,110	2,110	2,110	2,110	2,110	-3,427	7,647	871	4,588	2,110
B2: change in cattle emissions												
MMS	856	856	856	856	856	856	856	856	856	856	856	856
Enteric methane	0	0	0	0	0	0	0	0	0	0	0	10,209
D2: change in feed production												
Corn grain	-24,629	-24,629	-24,629	-24,629	-24,629	-24,629	-24,629	-24,629	-24,629	-24,629	-24,629	-24,629
DDGS	-20,181	-20,181	-20,181	-20,181	-20,181	-20,181	-20,181	-20,181	-20,181	-20,181	-20,181	-20,181
Corn silage	2,383	2,383	2,383	2,383	2,383	2,383	2,383	2,383	2,383	2,383	2,383	2,383
Net impact	**-9,975**	**-25,874**	**1,298**	**-23,500**	**3,549**	**-38,947**	**-1,846**	**-15,512**	**-4,438**	**-11,214**	**-7,497**	**234**

Table 4.5: Acidification potential sensitivity analysis results, in moles H+-eq. emissions

Category	Baseline	AD -low	AD -high	CH4 -low	CH4 -high	NPCC	RFC	TR- low	TR- high	GR -low	GR- high	Ym -4%
I1: Avoided AD process												
Biogas flaring	-830	-812	-839	-598	-1,063	-830	-830	-830	-830	-830	-830	-830
Biogas combustion, electricity surplus	-8,669	-8,091	-9,070	-5,410	-11,929	-8,669	-8,669	-8,669	-8,669	-8,669	-8,669	-8,669
Biogas combustion, AD operation	-2,974	-3,299	-2,700	-2,974	-2,974	-2,974	-2,974	-2,974	-2,974	-2,974	-2,974	-2,974
Wastewater transport and treatment	-84	-84	-84	-84	-84	-84	-84	-84	-84	-84	-84	-84
Capital goods	-614	-614	-614	-614	-614	-614	-614	-614	-614	-614	-614	-614
B1a: Change in fertilizer application												
Avoided digestate field emissions	-10,733	-10,733	-10,733	-10,733	-10,733	-10,733	-10,733	-10,733	-10,733	-10,733	-10,733	-10,733
Avoided digestate application	-290	-290	-290	-290	-290	-290	-290	-290	-290	-290	-290	-290
Avoided digestate transport	-470	-470	-470	-470	-470	-470	-470	-470	-470	-470	-470	-470
Fertilizer field emissions	3,220	3,220	3,220	3,220	3,220	3,220	3,220	3,220	3,220	3,220	3,220	3,220
Fertilizer application	4	4	4	4	4	4	4	4	4	4	4	4
Fertilizer transport	5	5	5	5	5	5	5	5	5	5	5	5
D1a: Production of marginal fertilizers	799	799	799	799	799	799	799	799	799	799	799	799
D1b: Production of marginal electricity	18,536	15,592	21,357	11,567	25,504	5,666	22,147	18,536	18,536	18,536	18,536	18,536
I2: feed transport and processing	1,456	1,456	1,456	1,456	1,456	1,456	1,456	-2,652	5,565	604	3,160	1,456
B2: change in cattle emissions												
MMS	3,940	3,940	3,940	3,940	3,940	3,940	3,940	3,940	3,940	3,940	3,940	3,940
D2: change in feed production												
Corn grain	-48,048	-48,048	-48,048	-48,048	-48,048	-48,048	-48,048	-48,048	-48,048	-48,048	-48,048	-48,048
DDGS	-9,007	-9,007	-9,007	-9,007	-9,007	-9,007	-9,007	-9,007	-9,007	-9,007	-9,007	-9,007
Corn silage	5,013	5,013	5,013	5,013	5,013	5,013	5,013	5,013	5,013	5,013	5,013	5,013
Net impact	**-48,745**	**-51,417**	**-46,059**	**-52,222**	**-45,268**	**-61,614**	**-45,134**	**-52,854**	**-44,636**	**-49,597**	**-47,041**	**-48,745**

Figure 4.1: Process flow for C-LCA of using food waste as dairy beef feed instead of as a substrate for anaerobic digestion.

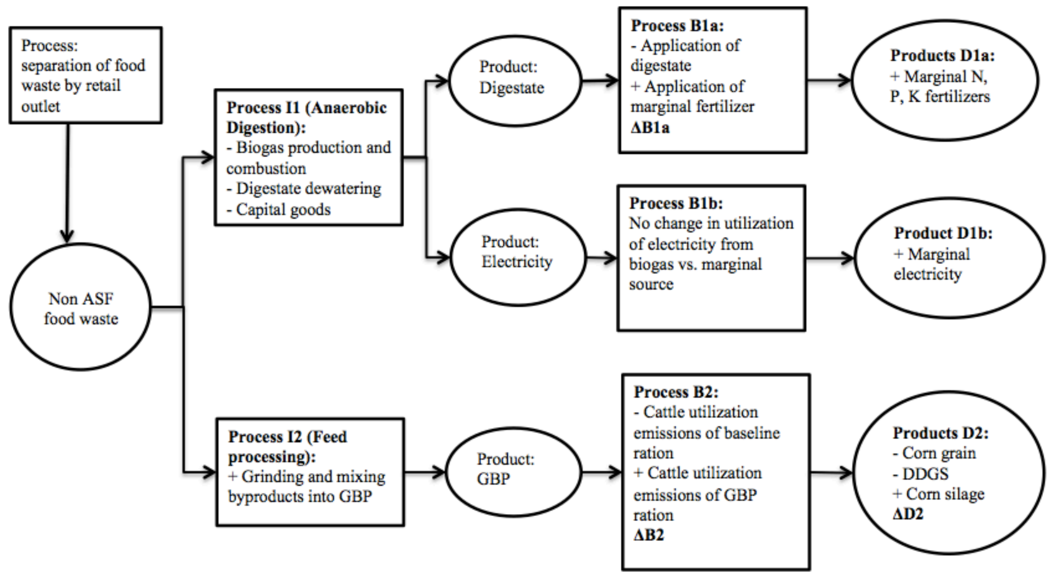
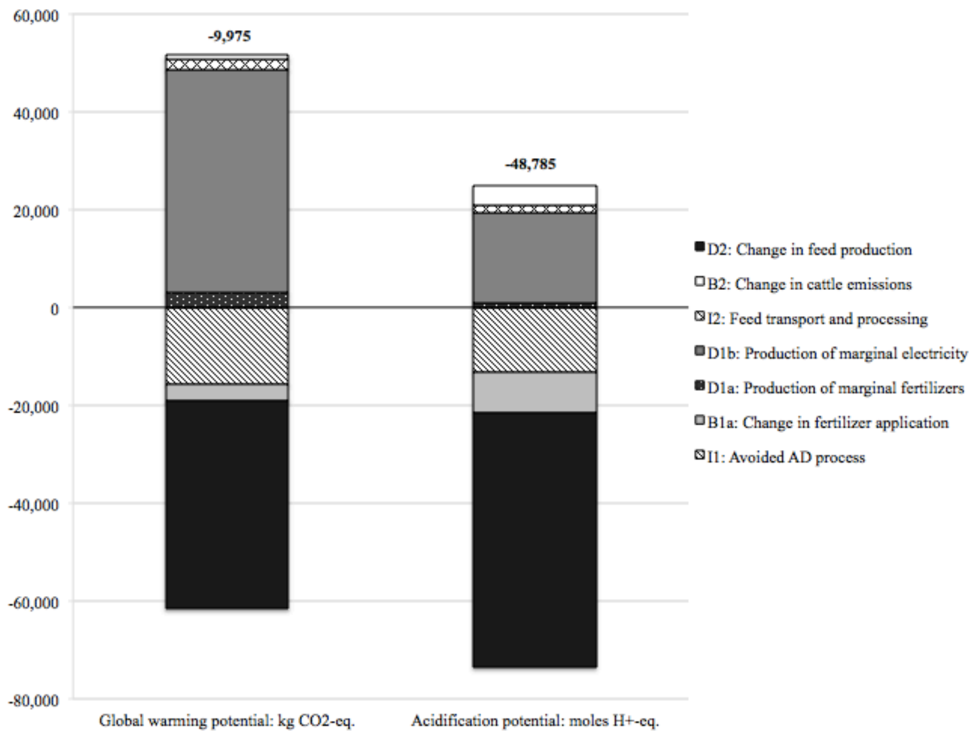


Figure 4.2: Environmental consequences of swapping food waste for corn in dairy beef rations instead of using it as a substrate for anaerobic digestion. Negative results at the top of the bars indicate net emissions savings. The reference flow is 71,492 kg DM (408,525 kg wet weight) food waste.



Chapter 5: Could we build a better burger by thinking regionally?

Written in the style of the blog *The Conversation*

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Cattle are responsible for a large fraction of global land use and livestock-related greenhouse gas emissions. Of all animal-based foods, beef has the largest carbon footprint per pound and in total when global production is summed (Gerber et al., 2013). In addition to being the leading global producer of beef (USDA-ERS, 2015a), the United States ranks among the top five countries in per-capita consumption (OECD, 2016). As a consequence, North America is the global region with the second highest GHG emissions of beef production (Gerber et al., 2013). Furthermore, over one-third of the carbon footprint of the U.S. food supply is attributable to beef (Heller and Keoleian, 2015). Given this state of affairs, the U.S. should lead the charge in developing innovative strategies to reduce the environmental burdens of the beef system.

Approaches to improve sustainability have largely focused on the ends of the supply chain – either consume less or produce better. However, both ends are part of a chain, part of a *system*. We could instead view the current state of affairs as an imbalance, where system transformation is the only route to sustainability (Garnett, 2014). The structure of the beef system itself may be an underlying cause of environmental degradation and source of future vulnerability. For example, over 80% of beef packing in the U.S. is controlled by four firms, whose facilities and immediate supply chains are concentrated in the Great Plains (Ward, 2010). Concentrations of feedlot cattle at the county-level have increased over time (Kellogg et al., 2000), calling into question the ability of surrounding ecosystems to assimilate excess manure nutrients. Enhancing regional self-reliance could be a strategy to address these trends. Increasing self-reliance of beef in major consuming regions (e.g., the Northeastern U.S.) could theoretically shift some of these nutrients and their resulting environmental burdens, such as eutrophication, closer to consumers. Restoring this feedback loop between production and consumption could, in theory, lead to more sustainable levels of both. Additionally, regional beef systems may rely on the ecological "leftovers" of a region – the marginal land, crop residues, and waste – that do not compete with the food supply. Such systems could complement the cultivation of plant-based foods for more sustainable diets.

Working in the Northeastern U.S., we asked whether regional beef production could enhance environmental sustainability and the food supply. We focused on two types of production systems. The first was 100% grass-fed beef, produced using management intensive grazing, which some evidence suggests may increase carbon sequestration. Furthermore, grass-fed beef is entirely reliant on forages, which may be grown on land otherwise unsuitable to produce food. Because the Northeast has a robust dairy industry, we also studied dairy beef, which includes the fattening of surplus calves from dairies and mature cattle (cows and bulls) that are culled from the herd.

Per pound of beef, Northeast dairy beef had a smaller carbon footprint than grass-fed beef, even when potential pasture carbon sequestration was included (Tichenor et al., in review). Because dairies produce both milk and beef, the environmental burden of maintaining the whole dairy herd can be divided between those products, which results in a smaller footprint compared to most other types of beef production systems. Northeast dairy beef is produced in an intensive system, where cattle are confined and fed high concentrate rations (e.g., corn, dry distillers grains). As such, producing dairy beef depleted more fossil fuel resources compared to grass-fed beef, and resulted in greater acidifying and eutrophying emissions across the landscape. At the same time, dairy beef also required dramatically less land than grass-fed beef. While comparing the quantity of land used by livestock systems is an overly simplistic measure of sustainability, we also estimated the extent to which the land used to feed cattle in these systems could instead be used to directly produce human food – the food supply opportunity cost of using that land to feed livestock instead of humans (Tichenor et al., 2016b). Because dairy beef uses much less land, and most of the land that feeds grass-fed beef is arable and moderately productive, dairy beef had a lower food supply opportunity cost compared to grass-fed. However, both systems produce less protein than could be produced by converting their feed land bases to human food crops.

For both systems, a large contributor to multiple categories of pollution and the determinant of the food supply opportunity cost was feed. Since dairy beef producers already own infrastructure for processing feed and feeding, we were curious if the burden of that system could be lowered further by feeding something we have in abundance in the Northeast: food waste. Some cattle feeders in the region partner with food processors and institutions to secure food waste and byproducts as low-cost feeds. Rutgers University, for example, has a longstanding partnership with a local beef producer to pick up processed cafeteria food scraps for feed (U.S. EPA, 2009). While recent organic waste landfill bans at state and municipal levels in the region increase the food waste supply available for feeding to livestock, they also increase competition for these materials by encouraging anaerobic digestion (Edwards et al., 2015).

We found that using food waste as a dairy beef cattle feed instead of as a feedstock for anaerobic digestion reduced greenhouse gas and acidifying emissions (Tichenor et al., 2016a). It also reduced feed land requirements, and in turn the human food supply opportunity cost of dairy beef. Even with substantial amounts of food waste substituted for corn feeds on dairy beef feedlots, the system still produced less protein than conversion of its feed land base to human food crops could. Since most of the land required for dairy beef is for maintaining the breeding herd, feeding large amounts of food waste on dairies may be a strategy to make dairy beef a net positive contributor to the human food supply.

So, what does all of this mean for the future of beef in the region? Proponents of grass-fed beef may argue that our research does not tell the whole story – and they would be correct. For one, pasture-based farming systems may provide other benefits beyond food production, such as biodiversity preservation, erosion regulation, and cultural value (Ripoll-Bosch et al., 2013). Estimating this value is difficult, however, and standardized methods to do so within a life cycle framework have yet to be developed. From a theoretical standpoint, the issue of who decides the worth of non-market goods and services is a pernicious one in and of itself. We ought to be making these questions and tradeoffs explicit as we consider the future of beef and food production more broadly in the region.

As the region looks to build a more sustainable future, innovations in dairy, beef, and waste management systems should be part of the vision. Dairy has historically been the backbone of many rural communities, particularly throughout New England and upstate New York. As long as there is dairy, there will be dairy beef. Dairy beef is often sold in mainstream or commodity markets, but there may be potential to develop a niche product. Dairy beef is less seasonal than beef produced with beef breeds, as dairy calves are born year-round. Using small and mid-scale plants in the region to slaughter dairy beef cattle could help smooth out seasonal trends that challenge processors (Lewis and Peters, 2012). Given this potential and what we have learned from our research, a state or municipal-level pilot program that facilitates production and marketing of regional waste-fed dairy beef should be a priority. Ideally, closed-loop, value chain partnerships could be developed, where food retailers or institutions would partner with dairy beef operations to provide both food waste for feed and a market to sell source-identified, regional beef. Cattle would need to be raised and slaughtered entirely in the Northeast region to be eligible. Establishing a brand and educating consumers at the point of purchase could help develop a market niche. Creative, public-private partnerships like these could go a long way toward more sustainable beef.

Food waste is an asset for regional dairy beef production, but it could also be used to feed other types of livestock. While the top priorities for food waste should be minimizing waste and maximizing recovery for direct human consumption, our research and similar work done with swine the UK (Salemdeeb et al., 2016) demonstrates that waste-to-feed should be preferred over other management strategies such as composting and anaerobic digestion. This is in line with the U.S. EPA's Food Recovery Hierarchy (U.S. EPA, 2016). To facilitate more sustainable management of food waste, states and municipalities in the region should develop policies and support structures that encourage waste-to-feed. First, finding and securing contracts to source food byproducts or waste can be extremely time consuming for farmers. Additionally, information on regulations or best practices for food-waste feeding can be difficult to locate, as each state may be different. State agencies and cooperative extension should provide support to farmers and businesses interested in waste-to-feed partnerships. Waste management and animal agriculture are typically handled by separate state agencies, which may be relatively siloed. Working across agencies to develop accessible, consistent information on waste-to-feed regulations and best practices is critical. Furthermore, states should develop incentive programs for waste-to-feed. For Massachusetts, Vermont, Connecticut, and New York City, which have already banned landfilling of organic wastes by commercial entities to varying degrees, these programs could be built into existing compliance processes (e.g., permitting or audits). State and federal incentives exist to develop waste-to-energy facilities, such as anaerobic digesters (Fitzgerald, 2013). It is time to consider policies that promote the utilization of food waste by the four-legged, living digesters we have reared in the region for centuries.

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Appendices: Data collection instruments

PRODUCER QUESTIONNAIRE

Life Cycle Assessment of Grass-fed Beef Production in the Northeastern U.S.

FARM ID: ___________________________________

Introduction
Thank you for agreeing to participate in this interview. We're going to discuss characteristics of your grass-fed beef herd, grazing management, pasture and forage production, winter feeding and management, water and supplements provided and slaughter metrics. I want to remind you that participation in this study is voluntary and you may decline to answer any questions or terminate your participation at any time without penalty. Let's get started.

Typical Production Timeline

Phase	1. Birth	2. Weaning	3. Slaughter	4. Total Months
Month			a. Steers: b. Heifers:	

Notes:

Herd Characteristics

5. What breed/breed(s) are your cattle?

6. Do you use any growth-promoting technology with your herd?

7. What are typical January inventories for the following cattle classes:
   Note: If typical unknown, try January 2015 and see if representative.

   a. Cows (incl. first calf heifers): _______ head

b. Bred heifers: ______ head

c. Mature bulls (2+ years): _______ head

8. Annual reproduction, herd management and mortality:

	Estimate	Head or rate (H/R)? Give units for rate.
a. Live calves (incl. twins		
b. Calf deaths (pre-wean)		
c. Culled cows		
d. Cow deaths		
e. Culled bulls		
f. Mature bull deaths		
g. Finishing cattle deaths		

Typically, at what ages are:

9. Cows culled: ______________ months OR years

10. Bulls culled: _______________ months OR years

11. Heifers first bred: ___________ months OR years

12. What is the source of fertility for your herd?

a. All bulls raised on this operation (closed herd)

b. Bulls purchased from another operation (open herd)

c. Artificial insemination

d. Combination of a, b and/or c

i. Please explain:

13. If bulls PURCHASED FROM ANOTHER OPERATION:

a. How often do you purchase? Every _______ years

b. How many bulls per purchase? _______ head

c. How old are the bulls when you purchase them? ______ months

d. How far away is the bull operation (one way)? ______ miles

14. How much do your cattle typically weigh in the following phases? *If unknown, indicate as such.*

b. Calves, at weaning	
e. Yearling heifer let out on pasture	
h. Yearling steer let out on pasture	
i. Finished heifer	
j. Finished steer	
k. Mature cow (2 year old)	
l. Mature bull (2 year old)	

15. Do you track average daily gains (ADG) of your herd? (Y/N) ______

If YES, please share ADGs for any of the following cattle classes:

	ADG (lb/day)
a. Backgrounding heifers (on winter feed)	
b. Backgrounding steers (on winter feed)	
c. Yearling heifers (on pasture)	
c. Finishing heifers (late stage/on winter feed)	
c. Yearling steers (on pasture)	
c. Finishing steers (late stage/on winter feed)	

16. What other metrics do you use to track the productivity of your cattle? For example, BCS.

Grazing Management

17. Do you keep any written records regarding grazing? For example, grazing plans: (Y/N): ________

    a. If YES, ask if we can review together and I can take photo(s).

18. Do you contract any of your grazing out to other producers? (Y/N): ______

    If YES, how much of the herd and for how long:

    a. __________ head

    b. ____ cattle class(es)

    c. __________ weeks OR months (circle)

19. How many animals are grazing on your pasture at any given time during the grazing season?

    a. Cows/cow-calf pairs: __________ head

    b. Bred replacement heifers: _________ head

    c. Yearling heifers: _________ head

    d. Yearling steers: ________ head

    e. Young replacement bulls (< 2 yrs): _________ head

    f. Mature bulls (2+ yrs): ___________ head

    g. Sheep, mature ewe or ram: __________ head

    h. Yearling sheep: ___________head

    i. Goat: ___________ head

j. Horse: __________ head

k. TOTAL: _________head

20. Do the above animals graze in one herd? (Y/N) ________

    a. If more than one, please describe the sizes of the herds:

21. What is your average stocking density for your grazed pastures (point in time, not cumulative)?

    ____________ head/acre OR ___________AU/acre

22. How do you decide when to move cattle from one paddock to another? *Ask about rules of thumb or measurements used if not mentioned up front.*

23. How long do your cattle typically stay in one paddock during the grazing season?

    _________ hours OR days

24. What, if any, machinery do you use to rotate your herd? *Either make/model, horsepower, or PTO drive estimates for equipment.*

    a. If machinery is used, how long does it take to rotate your herd?

    _____ hours

25. Is your fencing electrified? (Y/N) _______

a. If YES, what is the power source (e.g., battery)?

b. What are the output joules? ___________ J

c. IF BATTERY, how many times per year do you need to replace or recharge the battery?

____________ times replace OR recharge

Winter Feeding and Management

How much of the following feeds do you typically PURCHASE in a year and where do they come from?

Type	a. Quantity (tons or bales)	b. % legume	c. County of origin
26. Dry hay			
27. Haylage			
28. Baleage/ bale sileage			
29. Other			

30. If you produce or purchase baleage/bale silage for winter feeding, how do you dispose of plastic wrapping?

31. How do you manage your herd in the winter months *(place X in one per row)*:

Cattle class	On pasture, away from barn	On pasture, with barn access	Unvegetated/concrete pad	In barn

			pen	
a. Cows				
b. Mature bulls				
c. Bred heifers				
d. Unbred replacement heifers				
e. Backgrounding calves				
f. Finishing cattle				

32. For cattle on pasture and away from the barn, how many acres are used over the winter? _______________ acres

33. How do you manage manure that accumulates in the barn area (if applicable)?

a. Mixed with bedding? (Y/N) ______

b. Machinery used for removal (make/model, hp or PTO drive):

c. Frequency of removal: ______________ times per month

d. Removal time: _______ hours

e. Storage duration: _________________ months

f. All manure applied on farm? (Y/N) _______

34. What energy source provides power to your barn (if applicable)?

35. How do you provide feed to your herd during the winter months:

    a. Fed from feeder (e.g., ring feeder)? (Y/N) _______

    b. Frequency of feed delivery: every ____________ days

    c. What machinery is used to delivery feed? *Either make/model, hp or PTO drive estimates for equipment.*

    d. How long does each feeding take? _________ hours

Water and Supplements

36. What is the winter water source for your herd?

    Ground OR Surface OR Municipal

    a. Average distance from water source to herd: ____________

    b. Power source for pump (e.g., battery): ____________

37. What is the grazing season water source for your herd?

    Ground OR Surface OR Municipal

    a. Average distance from water source to herd: __________

    b. Power source for pump (e.g., battery): ____________

38. Do you feed a mineral supplement to your herd? (Y/N): ______

a. If YES, how much per head per day? __________ ounces/grams/lbs (circle)

Slaughter

39. Does your processor provide you the following cutout data for your grass-fed cattle?

    a. Dressing percentage? ______% SBW or LW or not sure

    b. Dressed weight? __________ lbs

    c. Grade?

        i. Ungraded

        ii. Select

        iii. Choice

        iv. Prime

Pasture and Forage Production

40. How many days of the year does grazed forage account for the majority of your herd's feed intake (including stockpiled pasture, if applicable)?

__________ days

How much land does your herd typically graze per year, and what is its composition (including stockpiled pasture)? *Place species in descending order of predominance.*

Crop type	a. Acres	b. % legume	c. Species 1	d. Species 2
41. Perennial pasture				
42. Annual pasture				
43. Forest/ woodland				
44. Crop residues				
45. Other:				

46. How much pasture do you stockpile to extend the grazing season? _________ acres

    a. If you stockpile pasture, how long does this practice extend the grazing season? _________ days

47. When you began your operation, which of the following did you use to establish your pastures (Y/N)?

    a. Tillage: _____

    b. Seeding grass and legumes: ______

    c. Seeding legumes only: ______

    d. Seeding grass only: ______

    e. N application: _______

    f. P application: ________

    g. K application: _______

    h. Lime application: ______

48. How often do you re-seed your pastures for maintenance? Every ________ years

49. What method (e.g., frost seed) and equipment do you use to re-seed your pastures? *Either make/model or PTO drive estimates for equipment.*

    a. If machinery is used, how long does it take you to re-seed your pastures: __________ hours

What fertilizers do you apply to your grazed pastures and how often?

Fertilizer	a. Tons/acre	b. Type	c. Frequency
50. N			
51. P			
52. K			
53. Lime			
54. Stored manure from winter		n/a	
55. Other:			

56. Other:			

What pesticides do you apply to your grazed pastures and how often?

Pesticide	a. Tons/acre	b. Frequency
57.		
58.		
59.		

60. Do you irrigate your pastures? (Y/N) ______

If YES, please describe your irrigation system:

a. Type (e.g., center pivot):

b. Water source:

Ground OR Surface

c. Frequency and duration of irrigation OR estimate of seasonal water use:

d. Power source (e.g., battery):

61. Excluding land your herd also grazes, how much forage land do you harvest for stored feed for your herd? This would be land dedicated to hay/haylage/baleage ONLY.

_______ acres

What fertilizers do you apply to your dedicated hay/haylage/baleage land and how often?

Fertilizer	a. Tons /acre	b. Type	c. Frequency
62. N			
63. P			
64. K			
65. Lime			

66. Stored manure from winter		n/a	
67. Other:			
68. Other:			

What pesticides do you apply to your dedicated hay/haylage/baleage land and how often?

Pesticide	a. Tons/ acre	b. Type	c. Frequency
69.			
70.			
71.			

How do you store your harvested forages and what is their composition?

Type	a. Acres harvested	b. % legume
72. Dry hay, in barn		
73. Dry hay, in open		
74. Baleage/bale silage		
75. Haylage		

76. What machinery do you use for mowing, tedding, raking, and baling (if applicable)? *Either make/model, hp, or PTO drive estimates.*

77. Do you keep track of how much forage you harvest per field/paddock each season? (Y/N) _________

    a. If YES, ask them to describe hay/baleage yields during the mapping exercise.

78. Have you tested the quality of your forages in the last 3 years? (Y/N) _________

If YES, please list the following by forage type:

Forage type	a. DM%	b. TDN (% DM)	c. CP (%DM)
79.			
80.			
81.			

PRODUCER QUESTIONNAIRE

Life Cycle Assessment of Dairy Beef Production in the Northeastern U.S.

FARM ID: ____________________________________

Introduction
Thank you for agreeing to participate in this interview. We're going to discuss characteristics of your dairy beef herd, feeding practices, housing/facilities, water and energy use. I want to remind you that participation in this study is voluntary and you may decline to answer any questions or terminate your participation at any time without penalty. Let's get started.

Facility Energy and Water Use

1. Are you able to share information regarding electricity use for your facility? Y/N

2. Is your facility on the same or a different electric meter than your house?

3. What is your annual or monthly electricity use?
    a. Annual:

    b. Monthly (1 summer, 1 winter month):

4. How much of each of the following inputs do you use annually for your dairy beef operation?

    a. Diesel:

    b. Gasoline:

    c. Natural gas:

    d. Fuel oil:

    e. LPG:

2. What is your annual or monthly water use/withdrawal ?

    a. Annual:

    b. Monthly (1 summer, 1 winter month):

Starter/Calf-Raiser Phase

1. How many calves are you typically feeding at one time? _________ head

2. What type of housing do you have for calves at your facility?

3. How many houses do you have for calves and large are they (sq. ft. or width and length)?

4. Is this an all in all out system? (Y/N)

5. Is your housing heated? (Y/N)

    a. What type of fuel is used for the heating system (natural gas, LP gas, fuel oil, electric)?

6. Are the floors of your facility slatted or covered with bedding?

    a. If bedding, how deep? What material?

    b. How often and how are they cleaned out?

7. How is manure stored in your facility, and for how long?

8. What percentage of manure is used on-farm for feed production? ________%

Feeding Questions

9. Do you use an automated system for milk replacer mixing? (Y/N)

10. Do you use an automated system for milk replacer feeding? (Y/N)

    a. IF YES, Which company manufactures this equipment? (e.g., Calf Star, Lely, Rombouts)

11. How long are the machines running per feeding? __________ minutes

12. How many times per day are calves fed? __________times

13. What machinery is used to clean equipment?

14. How often is equipment cleaned? __________ times per DAY/WEEK/MONTH

15. Is starter feed mixed on site or bought mixed?

16. What machinery is used to deliver starter feed?

17. How often do you deliver starter feed, and how long does it take?

Water Questions

18. Is the barn washed out with water to remove manure? (Y/N)

    a. If YES, how often is the barn washed out?

19. What other activities do you perform that require water on your calf operation?

    a. Cleaning mixing and feeding equipment?

    b. Washing out pens after calves leave?

    c. Other activities (list)

Performance Questions

1. What is your pre-weaning mortality rate for calves?

2. What is your post-weaning mortality rate for calves?

Growing and Finishing Phases

20. How large are your facilities for feeder cattle (in terms of head, at a point in time)?

21. What type of housing do you have at your facility for feeder cattle? (Options may include pen lot, confinement facility, barn with a concrete run)

22. Are the floors of your feeder cattle facility slatted or covered with bedding?

    c. If bedding, how deep? What material?

    d. How often and how are they cleaned out?

23. How is manure stored in your feeder cattle facility, and for how long?

24. What percentage of manure is used on-farm for feed production? ________%

Feeding Questions

25. How often are cattle fed? ____________ times per day

26. What machinery is used for feeding?

27. How long does each feeding take? ___________ minutes

Performance Questions

28. What mortality rates do you typically have for your finishing Holstein cattle?

29. How old are your Holstein cattle when they are sent to slaughter?

_________ months