Comparative life cycle environmental impacts of three beef production strategies in the Upper Midwestern United States

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ABSTRACT

We used ISO-compliant life cycle assessment (LCA) to compare the cumulative energy use, ecological footprint, greenhouse gas emissions and eutrophying emissions associated with models of three beef production strategies as currently practiced in the Upper Midwestern United States. Specifically we examined systems where calves were either: weaned directly to feedlots; weaned to out-of-state wheat pastures (backgrounded) then finished in feedlots; or finished wholly on managed pasture and hay. Impacts per live-weight kg of beef produced were highest for pasture-finished beef for all impact categories and lowest for feedlot-finished beef, assuming equilibrium conditions in soil organic carbon fluxes across systems. A sensitivity analysis indicated the possibility of substantial reductions in net greenhouse gas emissions for pasture systems under conditions of positive soil organic carbon sequestration potential. Forage utilization rates were also found to have a modest influence on impact levels in pasture-based beef production. Three measures of resource use efficiency were applied and indicated that beef production, whether feedlot or pasture-based, generates lower edible resource returns on material/energy investment relative to other food production strategies.

1. Introduction

Beef is an important animal husbandry food product, contributing roughly 30% of meat consumed in industrialized countries. The United States, a leader among beef-producing nations, was responsible for 20% of global beef production in 2007 (FAOstat, 2008). Beef production in the US is largely characterized by cow–calf herds maintained on pasture and (winter) hay, and mixed-ration feedlot finishing. Less than 1% of beef cattle are currently finished in pasture systems. Nonetheless, there is considerable variability in management strategies for both pasture and feedlot-finished beef, each with characteristic resource use and emissions patterns.

Life cycle assessment is an ISO-standardized biophysical accounting framework used to: (1) compile an inventory of the material and energy inputs and outputs characteristic of each stage of a product life cycle and (2) quantify how these flows contribute to specified resource use and emissions-related environmental impact categories (ISO, 2006). This allows the identification of key leverage points for reducing environmental impacts within supply chains, as well as comparisons of the resource dependencies and emission intensities of competing production technologies. Moreover, by bringing a suite of environmental accounting protocols under the umbrella of a single, rigorous framework, LCA facilitates evaluations of the environmental tradeoffs associated with different production strategies along multiple dimensions of environmental performance.

Researchers have previously called attention to the substantial feed, water and land requirements for ruminant production (Pimentel and Pimentel, 1996; Goodland, 1997; Gerbens-Leenes and Nonhebel, 2002). More recently, increased interest in the greenhouse gas intensity of food products has spurred a flurry of discussion in the popular media regarding the climate impacts of beef production and the comparative performance of feedlot and grass-based production systems.

LCA research has been used to examine the greenhouse gas intensity of conventional and organic beef production in Sweden (Cederberg and Darelius, 2000), Ireland (Casey and Holden, 2006a,b) and the UK; and the relative importance of the cow–calf and finishing phases for the farm-gate environmental impacts of Japanese beef production (Ogino et al., 2004, 2007). In the US, Kok-naroglu et al. (2007) have compared energy use in pasture and feedlot-based beef production and Phetteplace et al. (2001) investigated the influence of management strategies on greenhouse gas emissions in conventional beef production. However, full LCAs of US beef production strategies have not been reported to date. We contribute to this body of literature by using ISO-compliant LCA to evaluate four important measures of environmental performance (cumulative energy use, ecological footprint, greenhouse gas emissions and eutrophying emissions) for three distinct beef

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production strategies as currently practiced in the Upper Midwestern United States when weaned calves are either: sent directly to Iowa feedlots; sent to out-of-state small-grain (wheat and other) pastures (backgrounded) then finished in Iowa feedlots; or finished on pasture and hay in Iowa. We do not attempt to characterize optimal herds, but rather average contemporary production conditions for the strategies and region of interest as communicated to us by producers, beef researchers, and regional extension staff. We do, however, present sensitivity analyses to test the mitigation potential of soil organic carbon sequestration and improved forage utilization rates in pasture-based beef finishing.

2. Methods

We used ISO-compliant life cycle assessment to compare the cradle-to-farm gate cumulative energy use, ecological footprint, greenhouse gas emissions and eutrophying emissions associated with one managed pasture and two feedlot-finished beef production strategies in the Upper Midwestern United States. Our beef production models were developed in consultation with producers, beef researchers and extension specialists, and all rations were formulated by staff at the Iowa Beef Center at Iowa State University using the BRANDS performance model (Dahlke, 2004). We do not account for resource use and emissions associated with the production and maintenance of capital goods in any of the systems modelled.

2.1. Cow–calf system

The herd modelled for the cow–calf component of the beef life cycle comprises 100 cows, 15 heifers and three bulls. We assume a 90% annual calving rate, of which 15 are retained as replacement heifers. Seventy-five spring-born calves are sent to finishing in November at 216 kg and 15 cows to slaughter at an average weight of 636 kg (contributing to the total beef production for each system). Local bulls enter the system at 545 kg, having been raised on a diet of hay and grain equivalent to ¾ of the feedlot finishing diet. They are slaughtered after three years at a weight of 727 kg. We assume a similar cow–calf system provides calves directly from weaning to feedlots, out-of-state backgrounding on wheat pastures (see below) or pasture finishing. The cow–calf herd is maintained on legume frost-seeded (i.e. no tillage) pasture forage and hay, with small amounts of grain fed to the cows and heifers. No housing is provided.

2.2. Finishing systems

Calves sent directly to Iowa feedlots with hormone implants finish in 303 days at 637 kg. This strategy represents close to 50% of production in the US Upper Midwest.

Backgrounding refers to the feeding and management of steers and heifers from weaning until they enter a feedlot and are placed on a high-concentrate finishing ration. Wheat pastures are wheat fields that are seeded at 50–100% above normal planting rates, producing a high quality forage which can be grazed from early winter to late spring, depending on seeding date, then subsequently harvested for wheat at season's end (this allows wheat farmers to generate additional revenue per hectare without compromising yields) (KSU, 1993). In this scenario, Iowa cattle that are backgrounded prior to feedlot finishing are trucked to Oklahoma and then Kansas after weaning in November, for grazing on wheat pasture, before returning to an Iowa feedlot at 455 kg. These cattle finish in 450 days (300 days on pasture and 150 days in feedlots) at 637 kg. Hormone implants are employed during the feedlot stage only. This strategy similarly represents close to 50% of production in the US Upper Midwest.

Calves weaned to pasture in Iowa finish at 505 kg in 450 days on a ration of forage and hay. Hormone implants are not typically used in grass-finished beef production in the Upper Midwest, which largely serves niche markets demanding, among other things, hormone-free meat. Following consultations with regional producers and Upper Midwest beef researchers, we assumed a pasture utilization rate of 60% for both grass finishing and in the cow–calf system. This estimate falls mid-way between the 30–90% range suggested by Gerrish (2002) for pasture utilization rates in the Midwest.

Corn feed, which does not require processing and is typically sourced locally, is assumed to be transported 30 km by truck. All other non-pasture feed inputs are assumed to be transported 100 km. This reflects additional transport to and from processing in the case of soy co-products or lower Iowa production volumes relative to corn in the case of hay and wheat. In the absence of production strategy-specific on-farm direct energy input data, we apply average on-farm energy use for Minnesota beef production as reported by Ryan and Tiffany (1998) in proxy across all three production systems. This represents energy inputs associated with feed mixing and delivery in the feedlot systems, and with the movement of hay, fences, and cattle between paddocks in the pasture finishing system.

2.3. Nutrient management and gaseous emissions

For the cow–calf system and grass-finishing systems modelled, housing is not utilized hence all manure is assumed to be deposited directly to pasture. Manure production rates for feedlot finishing are estimated using the Excel-based Manure Nutrient and Solids Excretion Estimator model provided by the Iowa Beef Center at Iowa State University (Koelsch and Power, 2005). Manure in the feedlot-finishing systems is assumed to be scraped from feedlots and applied to agricultural land within a 5 km radius. Nitrogen and phosphorus emission rates are calculated based on feed composition and consumption, assuming that 2.6% of beef cattle body mass is nitrogen and 0.69% is phosphorus following Koelsch and Lesoing (1999). Nitrogen excretion estimates are used to calculate direct nitrous oxide, ammonia and nitric oxide emissions from manure management and indirect nitrous oxide emissions from nitrate leaching following IPCC (2006) protocols and Tier I emission factors. Methane emissions from manure management and enteric fermentation are calculated following IPCC (2006) Tiers I and II protocols respectively. Tier I protocols are applied for manure management given the trivial methane emissions associated with solid manure management, which is common to all systems modelled. Tier II protocols are applied for calculating enteric methane emissions due to the sensitivity of emissions to diet composition and throughput, and the relative importance of methane emissions to overall GHG emissions in ruminant production. Tier II protocols stipulate a 3 ± 1% methane conversion factor for feedlot diets containing >90% concentrates and a 6.5 ± 1% conversion factor for forage diets. We applied a 5.5% conversion factor for feedlot-finished cattle, since their rations contained high levels (but less than 90%) of concentrates. A 6.5% conversion factor was applied for the pasture phase and a 3% conversion factor for the feedlot stage of the backgrounded/feedlot-finished cattle, which have a feedlot finishing diet containing >90% concentrates. A 6.5% conversion factor was applied for the cow–calf phase and for grass-finished cattle, due to their forage and hay-based diet.

2.4. Fodder production

Inventory data for fodder production (see Supporting information Tables S1 and S2) were derived from the US National Agricultural Statistics Service (NASS), Iowa State extension publications

and peer-reviewed literature. Yields are based on 5-year averages for 2003–2007 calculated from NASS (2008) data. Fertilizer mixes correspond to average US consumption as reported by NASS (2008). Application rates of pesticides and fertilizers used in conventional soy and corn are based on 2005 data for Iowa (NASS, 2006). Energy inputs are based on Iowa averages for 2001 (NASS, 2004). Inputs to wheat production correspond to national averages (NASS, 2004, 2006). Fertilizer inputs and yields for hay and pasture production in Iowa follow Iowa State Extension statistical bulletins and production recommendations (ISU Extension, 1997, 2002, 2008). For out-of-state wheat pastures, they follow Oklahoma and Kansas State University Extension recommendations (OSU, 2008; KSU, 1993). Energy inputs for pasture and hay production are derived from EcoInvent (2008) models for farm machinery operation. All fertilizers and pesticides are assumed to be transported 1000 km by truck and all seed inputs 100 km by truck. Processing of crops applies inventory data reported by Pelletier and Tyedmers (2007) and Pelletier (2008).

Field-level emissions of carbon dioxide, nitrous oxide, ammonia, nitric oxide and nitrate related to nitrogen fertilizer application, biological nitrogen fixation and crop residues were calculated following IPCC (2006) Tier 1 protocols. Additional ammonia–nitrogen emissions at a rate of 5 kg/ha for all field crops was assumed following Andersen et al. (2001), and a standard atmospheric nitrogen deposition rate of 15 kg/ha was assumed across production regions. Indirect nitrous oxide emissions from nitrate leaching to water were calculated based on a standard leaching rate of 30% of surplus nitrogen following a nitrogen balance calculation as per IPCC (2006) guidelines. A 2.9% surplus phosphorus leaching rate was assumed and phosphate emissions calculated using a phosphorus balance following Dalgaard et al. (2008).

2.5. Co-product allocation

Co-product allocation is required to apportion resource use and emissions among the co-products of multi-output systems. Since the purpose of this analysis was to describe the biophysical environmental dimensions of a food production system, it was deemed appropriate to base allocation decisions on an inherent biophysical characteristic of crop co-products which both reflects the efficiency of the process and is relevant to the underlying causal impetus of the production system. To this end, the gross chemical energy content of co-product streams was chosen as the basis for all allocation decisions because (1) producing caloric energy is the root driver of all food production activities and (2) the chemical energy of food products present in raw materials is apportioned between processed outputs in a quantifiable manner which speaks directly to the efficiency with which the system provides food energy. For a detailed discussion of this rationale, see Ayer et al. (2007) and Pelletier and Tyedmers (2007).

2.6. Life cycle impact assessment

Impact assessment in LCA involves calculating the contributions made by the material and energy inputs and outputs tabulated in the inventory phase to a specified suite of environmental impact categories. We considered two resource use impact categories (energy use and ecological footprint) and two emissions-related categories (greenhouse gas emissions and eutrophying emissions) that we believe are of global relevance for considering environmental performance in animal husbandry. All impacts were calculated using the SimaPro 7.1 LCA software package from PRé Consultants (PRé, 2008). Energy use (MJ) was quantified following the Cumulative Energy Demand method (Frischknecht et al., 2003), which takes into account the conversion efficiencies of primary energy

carriers. Whereas ecological footprints have historically been calculated using a stand-alone methodology, the recent incorporation of this method as an impact assessment option in the SimaPro software package now facilitates its use alongside more standard impact assessment methods. The ecological footprint method is unique among LCA impact assessment methods in that it provides a direct estimate of the ecological dependence of economic activity by expressing the resource inputs and waste assimilatory services underpinning specific economic goods and services in terms of the area of productive ecosystem required to furnish them (Rees and Wackernagel, 1994). This includes direct land occupation for producing resources as well as the forest land required to sequester emissions. Since productive ecosystem is ultimately a limited resource, this metric facilitates management of cumulative demand relative to biocapacity (Rees and Wackernagel, 1994). The ecological footprint was calculated following the EcoInvent 2.0 method (EcoInvent, 2008). This method was modified to include methane and nitrous oxide emissions. We believe this method to be of value to our analysis because it facilitates quantification of the ecosystem support (as measured in area of productive ecosystem) required to underpin human activities, which are typically ignored in LCA research. We also believe this measure to be more relevant than simple estimates of land use, which are sometimes quantified in LCA research but are not sensitive to the quality of land use. Greenhouse gas emissions (expressed as CO_2-equiv. units) were quantified using the IPCC (2007) method, assuming a 100-year time horizon. Eutrophying emissions (all emissions of nitrogen and phosphorus-containing compounds, expressed as PO_4-equiv. units) were quantified following the CML 2001 method (Guinee et al., 2001). These assessment methods follow the problem-oriented mid-point approach, meaning that results are expressed in terms of total resource use and emissions rather than actual impact levels.

2.7. Life cycle interpretation

Impacts were calculated on a whole-herd basis and per kg of live-weight production in each system. Cradle-to-farm gate supply chain impacts were assessed to identify impact hotspots and key leverage points for environmental performance improvements. Comparative impacts between production systems were also evaluated. Although our models assumed equilibrium conditions in soil organic carbon (SOC) flux associated with feedlot and grass-based beef production (pers. comm., Keith Paustian, Colorado State University, and Cindy Cambardella, National Soil Tilth Laboratory), we conducted a sensitivity analysis to test the potential impact any such differences might have on overall GHG emissions. Specifically, we applied estimates of 0.12 tonnes C sequestered/ha/year for improved cow–calf pastures and 0.4 tonnes C sequestered/ha/year for previously unmanaged pastures subjected to management-intensive grazing for pasture finishing following Phetteplace et al. (2001). We assumed SOC equilibrium conditions for all other feed input production systems. We also tested the sensitivity of model outcomes to differences in assumed forage utilization rates under management-intensive grazing in pasture-based beef finishing by alternately applying utilization rates of 30%, 60% and 90% following the range described by Gerrish (2002). A sensitivity analysis was also conducted to estimate GHG emissions for feedlot-finished beef using a methane conversion factor of 3% (IPCC default when feedlot diets contain greater than 90% concentrates) in place of our assumed 5.5%. Finally, we assessed the energy return on investment (EROI) ratios in feedlot and grass-based beef production systems according to: (a) the amount of human-edible food energy produced relative to the total industrial (human-mediated) energy inputs required; (b) the amount of human-edible food energy produced relative to the amount of human-edible food energy

consumed by the cattle; and (c) the amount of gross chemical energy produced relative to the gross energy consumption of cattle in each scenario. Whereas EROI measures typically focus exclusively on returns relative to industrial energy inputs, we believe that these additional EROI measures speak effectively to equally important biotic resource use efficiency considerations (from an anthropocentric perspective for (b) and an ecocentric perspective for (c)) which are often overlooked in discourse regarding resource allocation and depletion issues.

3. Results

3.1. Life cycle inventory results

Since we assumed a similar cow–calf herd provides calves to both feedlot and pasture-based finishing systems, modelling a single ration plan was sufficient (Table 1). This consisted predominately of pasture and hay, with a small amount of wheat fed to cows and heifers. In contrast, inputs and performance in the finishing scenarios varied widely (Table 2) (for detailed inventory data for inputs and emissions associated with the production of feed inputs, see Tables S1 and S2). In particular, the average daily gain (ADG) in the feedlot-only finishing scenario as estimated by the BRANDS model (Dahlke, 2004) was more than twice that of the pasture finishing scenario, with the backgrounding/feedlot finishing scenario falling between these extremes. Feedlot finishing utilized a range of grain and crop co-product inputs.

Similarly, whereas only one set of protein, nitrogen and phosphorus excretion estimates were necessary for the cow/calf phase,

Table 1
Annual ration plan (on an as-fed basis) and outputs for the cow–calf phase.

Feed inputs[a] (tonnes)	Cows (100)	Bulls (3)	Heifers (15)
40% leg pasture[b]	1178.4	36.3	107.2
Mixed grass hay	296.1	11.2	18.1
Wheat	9.3	–	1.5
Outputs			
Calves	75		
Live-weight beef	5445 kg	727 kg	

[a] Minerals and/or supplements not included in analysis.
[b] Direct consumption, but 60% utilization rate assumed hence cultivated area scaled accordingly in actual models.

Table 2
Ration plan (on an as-fed basis) and performance for finishing weaned calves: in feedlots on mixed-ration diets; on a combination of out-of-state wheat pasture and mixed-ration feedlot diets; or on pasture and hay.

Feed input[a] (tonnes)	Feedlot	Backgrounding/feedlot	Pasture
Brome pasture[b]			112.5
40% leg pasture[b]	–		1031.2
Wheat pasture		726.7	
Mixed grass hay			54.3
Alfalfa hay-mid			36.2
Alfalfa hay-mature	25.1	21.2	
Corn silage	60.4		
Corn grain	150.4	109.6	
Corn gluten feed	52.7		
Corn wet distillers		90.2	
Soy meal	5.5		
Weight in (kg)	216	216	216
Weight out (kg)	636	636	505
Days fed	303	450	450
Average daily gain (kg)	1.4	0.9	0.6

[a] Minerals and/or supplements not included in analysis.
[b] Direct consumption, but 60% utilization rate assumed hence cultivated area scaled accordingly in actual models.

these estimates varied considerably between the finishing scenarios (Tables S3 and S4) (for details regarding the composition of feed inputs used for these calculations, see Table S5). On a whole-herd basis, protein and nitrogen intake and excretion were lowest for the feedlot finishing scenario, and highest for the backgrounding/feedlot scenario. In contrast, calculated phosphorus excretion was lowest for grass finishing, and highest for backgrounding/feedlot finishing (Tables S3 and S4).

3.2. Life cycle impact assessment results

Maintaining cows is, by an order of magnitude, the most resource and emissions intensive aspect of the cow–calf phase across impact categories (Table 3). Within impact categories, feed production is the dominant contributor to cumulative energy use as well as the ecological footprint of beef production in both the cow/calf and finishing stages (Tables 3 and 4). The size of the ecological footprint is primarily determined by land occupation for crop and pasture production, although the area of global average productive ecosystem required to sequester an amount of carbon dioxide equivalent to the methane and nitrous oxide emissions produced by the cattle and their manure is also substantial (Tables S6 and S7). Nutrient losses from manure, whether directly deposited on pasture by grazing animals or scraped and applied to agricultural fields in the case of feedlot finishing, make the largest contribution to eutrophying emissions for all three scenarios, followed closely by feed production. For greenhouse gas emissions, enteric methane is the leading factor, although both feed production and manure management (primarily nitrous oxide emissions) also make substantial contributions. On a whole-herd basis, impacts are consistently lowest for the feedlot scenario across impact categories. The backgrounding/feedlot herd has the highest cumulative energy demand, greenhouse gas and eutrophying emissions, but a slightly smaller ecological footprint than the grass-finished herd (Tables 3 and 4). Direct, farm-level inputs account for 6% of cumulative energy use for grass finishing, and 13.6% for feedlot finishing.

For all impact categories and scenarios, the cow–calf phase is the greater contributor to resource use and emissions in beef production. Averaged across impact categories, the cow–calf phase is responsible for approximately 63% of impacts per live-weight kg of beef produced in all three of the finishing scenarios (Fig. 1).

Since the BRANDS model (Dahlke, 2004) predicts that feedlot-finished and backgrounding/feedlot-finished animals in the systems we modelled are 132 kg heavier than the grass-finished cattle, impacts per live-weight kg produced follow a different pattern than those observed on a whole-herd basis. Here, impacts are consistently highest across impact categories for grass-finished beef and lowest for feedlot-finished beef. The backgrounding/feedlot-finished beef falls roughly mid-way between these extremes (Fig. 1).

If the rates of 0.12 tonnes C sequestered/ha/year for improved pastures (cow–calf system) and 0.4 tonnes C sequestered/ha/year for pastures recently converted to management-intensive grazing (grass finishing) employed by Phetteplace et al. (2001) are realistic for the Upper Midwestern systems modelled, estimated greenhouse gas emissions per live-weight kg produced would be 1.8 kg less for feedlot-finished or backgrounding/feedlot-finished beef, and 8.2 kg less for beef finished on intensively-grazed improved pastures and hay during the transition phase. Here, rather than the 30% difference in emissions calculated based on assumed equilibrium conditions, grass-finished beef would be 15% less greenhouse gas intensive than feedlot-finished beef (Fig. 2).

We also tested the sensitivity of our models to variation in forage utilization rates in pasture finishing. At a utilization rate of 30%, average impacts were 22% higher than at our assumed utilization rate of 60%. A 90% utilization rate would reduce average im-

Table 3
Annual cradle-to-farm gate life cycle cumulative energy use (MJ), ecological footprint (area of productive ecosystem), and greenhouse gas (CO_2-equiv.) and eutrophying (PO_4-equiv.) emissions associated with a cow–calf herd providing 75 calves for beef production in the Upper Midwestern United states.

	Cumulative energy use (GJ)	GHG emissions (tonnes CO_2-equiv.)	Eutrophying emissions (tonnes PO_4-equiv.)	Ecological footprint (ha)
Bulls (3)	**71.2**	**27.8**	**.179**	**14.0**
Feed production	59.8%	21.5%	27.0%	55.0%
Enteric methane	–	28.9%	–	15.3%
Manure[a]	–	13.6%	41.3%	7.1%
Other[b]	40.2%	36.0%	31.7%	22.6%
Heifers (15)	**106**	**39.7**	**.269**	**25.9**
Feed production	90.7%	34.6%	46.2%	73.2%
Enteric methane	–	44.4%	–	18.2%
Manure	–	19.3%	53.3%	7.9%
Other	9.3%	1.7%	0.5%	0.7%
Cows (100)	**1320**	**531**	**3.74**	**330**
Feed production	95%	33.4%	40%	71.3%
Enteric methane	–	44.1%	–	19.0%
Manure	–	21.6%	59.7%	9.3%
Other	5%	0.9%	0.3%	0.4%
Total	**1500**	**599**	**4.18**	**370**
Feed production	93%	32.9%	40%	70.8%
Enteric methane	–	43.4%	–	18.8%
Manure	–	21.1%	59.5%	9.1%
Other	7%	2.6%	0.5%	1.3%

[a] Predominately nitrous oxide, but also includes manure methane.
[b] Predominately legacy cost of producing bull.

Table 4
Cradle-to-farm gate life cycle cumulative energy use (MJ), ecological footprint (area of productive ecosystem), and greenhouse gas (CO_2-equiv.) and eutrophying (PO_4-equiv.) emissions for 75 head herds of beef cattle finished: in feedlots; a combination of backgrounding followed by feedlot finishing; or grass-based pasture finishing systems in the Upper Midwestern United States.

	Cumulative energy use (GJ)	GHG emissions (tonnes CO_2-equiv.)	Eutrophying emissions (tonnes PO_4-equiv.)	Ecological footprint (ha)
Feedlot	**714**	**262**	**1.85**	**119**
Feed production	85.6%	26.7%	12.6%	56.8%
Enteric methane	–	40.2%	–	23.7%
Manure[a]	0.6%	30.4%	86.9%	17.9%
Other[b]	13.8%	2.7%	0.5%	1.6%
Backgrounding/feedlot	**1110**	**340**	**2.74**	**198**
Backgrounding	577	188	1.42	129
Feed production	45%	20.4%	20.9%	49.4%
Enteric methane	–	20.6%	–	9.5%
Manure	–	12.7%	30.8%	5.8%
Other[c]	6.9%	1.6%	0.1%	0.5%
Feedlot	533	152	1.32	69
Feed production	43.4%	16%	6.5%	21.5%
Enteric methane	–	11.4%	–	5.3%
Manure	<0.2%	16.4%	41.5%	7.5%
Other[c]	4.5%	0.9%	0.2%	0.5%
Pasture	**830**	**325**	**2.67**	**208**
Feed production	93.7%	36.6%	51.3%	73.5%
Enteric methane	–	41.5%	–	17.3%
Manure	–	20.9%	48.4%	8.7%
Other	6.3%	1%	0.3%	0.5%

[a] Includes nitrous oxide and methane emissions, as well as energy-related inputs/emissions associated with manure handling.
[b] Includes on-farm energy use as estimated by Ryan and Tiffany (1998) for Minnesota beef production.
[c] Includes transport of calves to out-of-state pastures in Oklahoma and Kansas, then back to Iowa.

pacts by 7% (Table 5). As a final sensitivity analysis, we tested the influence of our assumed methane conversion factor for feedlot finishing. Compared to our assumed 5.5% conversion factor, at a conversion factor of 3% (the IPCC default where diets contain greater than 90% concentrates), total emissions per live-weight kg of feedlot-finished beef would be only 6% lower than estimated. In light of the general importance of methane to overall emissions in beef production, this surprisingly modest improvement reflects that the feedlot stage contributes only 30% of total emissions.

Human-edible energy return on industrial energy investment in each system closely followed observed patterns in cumulative en-

ergy use between systems, with highest returns (5.2%) for feedlot finishing and lowest returns (4.1%) for pasture finishing. In contrast, because of the fractions of corn and soy (which could be directly consumed by humans) used in feedlot and backgrounding/feedlot finishing rations, returns on human-edible energy investment were an order of magnitude higher for pasture finishing (where the only human-edible material consumed is the small amount of grain used in the cow/calf phase) compared to feedlot-finishing, although still less than 100%. However, due to the higher feed throughput volumes per unit production in pasture finishing, gross chemical energy returns on investment were highest

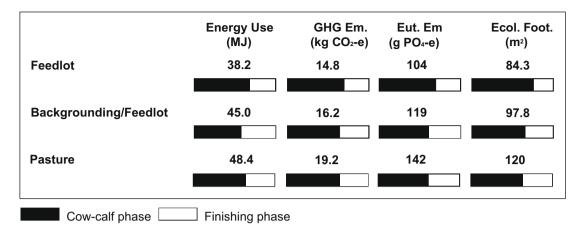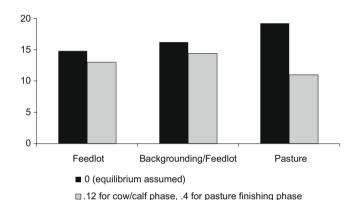|	Energy Use (MJ)	GHG Em. (kg CO_2-e)	Eut. Em (g PO_4-e)	Ecol. Foot. (m²)
Feedlot	38.2	14.8	104	84.3
Backgrounding/Feedlot	45.0	16.2	119	97.8
Pasture	48.4	19.2	142	120

■ Cow-calf phase   □ Finishing phase

Fig. 1. Cradle-to-farm gate life cycle cumulative energy use (MJ), ecological footprint (area of productive ecosystem), and greenhouse gas (CO_2-equiv.) and eutrophying (PO_4-equiv.) emissions per kg of live-weight beef produced in feedlot, backgrounding/feedlot, and pasture-finishing beef production systems in the Upper Midwestern United States.

■ 0 (equilibrium assumed)
□ .12 for cow/calf phase, .4 for pasture finishing phase

Fig. 2. Anticipated GHG emissions per live-weight kg of beef produced in feedlot and grass-finishing systems in the Upper Midwestern United States assuming either equilibrium conditions for soil organic carbon in pasture systems or 0.12 kg C sequestration/ha/year for improved pastures (cow–calf phase) and 0.4 tonnes C sequestration/ha/year for intensive grazing following Phetteplace et al. (2001).

for feedlot-finished beef (2%) and lowest for pasture-finished beef (1.6%) (Table 6).

4. Discussion

4.1. Cumulative energy use

Consistent with numerous analyses of animal husbandry systems, we found that feed production was the largest contributor to life cycle energy use in beef production (for example, see Basset-Mens and van der Werf, 2005; Thomassen and de Boer, 2008). However, our results contradict previous suggestions that pasture-finished beef production (48.4 MJ/kg) is necessarily less energy intensive than feedlot-finished beef production (38.2 MJ/kg) (Pimentel and Pimentel, 1996; Koknaroglu et al., 2007). Our

Table 6
Energy return on investment (EROI) ratios for feedlot and grass-based beef production in the Upper Midwestern United States as: (a) human edible caloric energy return on industrial energy investment, (b) human edible caloric energy return on human edible caloric energy investment and (c) gross chemical energy return on gross chemical energy investment.

EROI	Feedlot (%)	Backgrounding/feedlot (%)	Pasture (%)
Industrial energy[a]	5.2	4.4	4.1
Human-edible energy[a]	4.2	5.9	69.1
Gross chemical energy[b]	2.0	1.8	1.6

[a] Assumes 43% yield of boneless meat per live-weight kg produced and an energy density of 4.63 MJ/kg of raw, boneless beef.
[b] Assumes a whole-animal energy density of 4.63 MJ/kg.

finding is somewhat surprising given the widely perceived notion that pastures are largely solar-driven systems, whereas grain-based production is clearly underpinned by fossil energy inputs in the form of fuel for farm machinery, pesticide and fertilizer production and application, and crop processing and transportation. Three important distinctions are necessary in explanation. First, in the temperate climates characteristic of the US Upper Midwest, hay comprises a substantial fraction of the winter diets of grass-fed animals. Hay production and transportation have associated energy costs which may be similar or greater than those of substitutable feed inputs (in the southern US, where pasture is available for a greater proportion of the year, it is possible that energy inputs may be much lower). Second, the managed pastures modelled in our scenarios are quite distinct from unmanaged rangeland in terms of both inputs and forage yields, requiring energy inputs in the form of fertilizer production and application, seeding, and periodic renovation (the latter was not included in our models) to maintain productivity. Third, the large feed throughput volumes and significant trampling rate associated with forage diets in the systems we modelled serve to amplify the areas of managed pastures required. It should be noted that beef produced on unmanaged rangeland may, indeed, be considerably less energy

Table 5
Life cycle impacts per kg of live-weight beef produced under management-intensive pasture finishing regimes in the Upper Midwestern United States as a function of forage utilization rate (bracketed values represent difference relative to assumed 60% utilization rate).

Forage utilization rate (%)	Cumulative energy use (MJ)	GHG emissions (kg CO_2-equiv.)	Eutrophying emissions (g PO_4-equiv.)	Ecological footprint (m²)
30	63.8 (+32%)	21.5 (+12%)	169 (+19%)	150 (+25%)
60	48.4	19.2	142	120
90	43.3 (−10%)	18.4 (−4%)	133 (−6%)	110 (−8%)

intensive than the systems we modelled, although this would also result in tradeoffs in terms of animal performance and associated emissions. Interestingly, the transportation of animals to and from out-of-state pastures in the backgrounding/feedlot scenario contributed negligibly to overall energy use.

We also recognize that because we were unable to accurately characterize direct feedlot or pasture-level energy use, our use of average energy consumption for Minnesota beef production systems as a proxy for system-specific estimates is a weakness in our analysis. Preliminary work examining the economic costs of energy use in forage-only versus conventional cow herds in Iowa suggests slightly higher energy costs for forage-only systems (pers. comm., Denise Schwab, Iowa County Extension). However, given the trivial contribution made by these direct energy inputs in our analysis (only 6% of cumulative energy use for grass finishing and 13.6% for feedlot finishing) relative to the energy use associated with feed provision (Koknaroglu et al., 2007) in all systems modelled, we are confident that this deficiency does not significantly influence our results.

4.2. Greenhouse gas emissions

Our results suggest that pasture-finished beef (19.2 kg CO_2-e/kg) from managed grazing systems as currently practiced in the US Upper Midwest is more greenhouse gas intensive than feedlot-finished beef (14.8 kg CO_2-e/kg) when viewed on an equal live-weight production basis. This conclusion is consistent with previous research, which has shown that higher quality diets and increased growth rates reduce ruminant methane and manure nitrous oxide emissions, both of which are key contributors to life cycle emissions (Holter and Young, 1992; Lovett et al., 2005; Benchaar et al., 2001). For example, Casey and Holden (2006a) found that high concentrate diets and the associated reduction in finishing time reduced greenhouse gas emissions in Irish beef production. Hyslop (2008) made similar observations from simulations of UK beef production, and Phetteplace et al. (2001) suggested that shorter finishing times achieved by moving calves directly to the feedlot reduced emissions in US beef production. These reductions serve to mitigate the greenhouse gas emissions associated with producing, processing, and transporting feedlot ration inputs. However, greenhouse gas-reduction benefits have also been attributed to intensive grazing systems. Phetteplace et al. (2001) found that a transition to intensive grazing during the cow–calf phase as opposed to less management-intensive grazing can reduce emissions, and DeRamus et al. (2003) reported that best management practices in grazing systems could reduce enteric methane emissions by as much as 22% compared to continuous grazing. Casey and Holden (2006b) suggest that extensifying beef production through organic practices may result in lower GHG emissions in Irish suckler-beef herds.

Certainly, substantial reductions in greenhouse gas emissions may be possible in both feedlot and pasture-based production systems through genetic selection, forage selection and management, methane inhibition and animal management (Wittenberg, 2008). For example, several studies point to the potential of improving the greenhouse gas performance of forage diets through the inclusion of specific legumes (see McCaughey et al., 1999; Waghorn et al., 2002). Although beyond the scope of the current analysis, it would be interesting to assess the comparative greenhouse gas emissions of the feedlot and pasture-finished beef productions systems modelled here where diets are tailored expressly for reduced methane emissions. As 1 kg of methane is equivalent to 55.2 MJ of lost feed energy, mitigation strategies can have the additional benefits of improved feed utilization (Wittenberg, 2008), and decreased energy use and emissions for feed production.

The majority of models of greenhouse gas balances in agricultural production assume that established systems achieve equilibrium conditions in soil organic carbon (SOC) flux (for example, see Watson et al., 2002; Freibauer et al., 2004). In other words, although grass-based systems may certainly store larger soil carbon stocks than cultivated systems, no net differences in annual flux should be anticipated between well established systems. Following expert consultation (pers. comm., Keith Paustian, Colorado State University, and Cindy Cambardella, National Soil Tilth Laboratory), we adopted this assumption in our modelling endeavours. Several authors, however, have suggested that some pasture lands may, in fact, sequester carbon on an on-going basis. For example, Soussana et al. (2007) challenge the concept of carbon sink saturation in European grasslands. Less controversial is the idea that changes in management strategies may also change SOC dynamics (Conant et al., 2003). Our sensitivity analysis, which employed the soil organic carbon sequestration rates used by Phetteplace et al. (2001) for US pastures undergoing improvement or a transition to management-intensive grazing, suggested a substantial net reduction in the total GHG balance in grazing systems, with pastured beef producing 15% less net GHG emissions compared to feedlot-finished beef. This sensitivity analysis did not, however, account for decreasing SOC sequestration rates over time as the systems approach new equilibrium conditions.

4.3. Eutrophying emissions

Again, the higher estimated eutrophication potential for grass versus feedlot-finished beef production (142 versus 104 g PO_4-e/kg) in the systems we modelled was a direct product of the large feed throughput for forage diets, compounded by a significant trampling rate, the associated scaling effect for the area of managed pasture required, and the greater amount of manure produced relative to live-weight production. Even though losses from the feedlot system were accounted for both at time of excretion and when manure was applied to agricultural land, these were ultimately less than total emissions in the grass-finishing system. We recognize, however, that leaching rates are context-sensitive. While our application of standard leaching factors across production systems provides a reasonable first order estimation of eutrophication potential, we recommend further research in this area. Also of note is that, in contrast to nitrogen excretion, phosphorus excretion was lowest in pasture-based production, which may have important implications for comparative eutrophication potentials of alternative beef production strategies in the US Upper Midwest. Haan et al. (2006) found that percent surface cover was the most important determinant of P losses, with well-managed pasture lands not increasing surface water P-levels relative to ungrazed grassland. Similarly, Boody et al. (2005) reported that replacing 7–14% of cultivated lands with grasslands in two Minnesota watersheds resulted in a 71–75% decrease in phosphorus loading.

4.4. Ecological footprint

Direct land occupation contributes the largest share of the ecological footprint of both feedlot (for grain and co-product production) and grass-finished (for pasture and hay) beef production. Although the footprint method weights pasture and cropland differently (because pasture provides a greater range of ecosystem services than does cultivated cropland) (Frischknecht et al., 2003), the large areas required ultimately contribute to a larger ecological footprint for grass-finished (120 m^2/kg) versus feedlot-finished (84.3 m^2/kg) beef. The footprint tool also points to the large indirect ecosystem requirements of beef production to

assimilate an amount of carbon dioxide equivalent to the greenhouse gas emissions (including methane and nitrous oxide) produced.

4.5. Whole-system perspective

While certain parallels may be drawn to other major animal husbandry sectors (for example, the importance of feed production in life cycle energy use), the patterns of resource use and emissions in beef production are in many ways unique. Foremost among these is the tremendous importance of the cow–calf phase. Similar to Phetteplace et al. (2001), we found that the cow–calf phase is the dominant contributor to most impact categories regardless of finishing strategy. This is largely attributable to the low fecundity of cattle compared to other species such as pigs and chickens. Since a cow will produce at most one calf per year, a mature cow is maintained (along with bulls and heifers) for every marketed animal (Williams et al., 2006). This more than doubles the resource requirements and emissions per live-weight kg of beef produced. Casey and Holden (2006a) and Cederberg and Stadig (2003) both recommend combined dairy and beef systems as a means of reducing the impacts of calf production, since dairy cows produce both milk and calves whereas beef cow/calf herds are maintained for calf production only.

Also important is the relative feed use efficiency of ruminants compared to monogastric species. Ruminants are able to subsist on relatively high volumes of low-quality forage due to their unique digestive system, which relies on symbiotic methanogenic bacteria to break down cellulose. However, this throughput volume not only magnifies the resource and emissions burdens of feed production, but also results in considerable methane generation, manure nitrous oxide, and eutrophying emissions.

Taken together, these factors contribute to the relatively low resource returns on material/energy investment observed in the beef production systems we modelled. For edible energy return on industrial energy investment, these range from 4.1% (grass finishing) to 5.2% (feedlot finishing). The energy intensity of beef relative to other food commodities (Flachowsky, 2002; Carlsson-Kanyama et al., 2003; Carlsson-Kanyama, 2004) thus places the sector in a disadvantaged position. For example, the edible energy return on industrial energy investment for US broiler poultry is roughly 16% (calculated from Pelletier, 2008), and close to 27% for high-performing US pork production systems (Pelletier et al., submitted for publication). For nationally-sourced vegetable protein sources, Carlsson-Kanyama et al. (2003) reports a range in energy use three to 10 times lower than for beef products.

Also of interest are the edible food energy returns relative to the amount of feedstuffs consumed by cattle which could have otherwise been directly consumed by humans. Despite the large fraction of human-inedible co-products consumed by feedlot cattle, the small volumes of grain consumed by the cow–calf herd and the corn and soy consumed in the feedlot result in a return of only 4.2%, with slightly higher returns for backgrounded cattle. Since the pastured beef cattle we modelled are not consuming any human-edible products during finishing, the returns are an order of magnitude higher than in the feedlot system. From an anthropocentric perspective, this underscores the benefits of producing pastured beef on land not suitable for agricultural crops. Even the pastured system modelled, however, resulted in a net deficit of human-edible food energy due to the low-level grain consumption during the cow–calf phase.

In a related vein, it has been argued elsewhere that ruminant production provides an efficient means of converting otherwise inedible biotic resources (forage and crop processing co-products) into a human-edible food source (Garnett, 2009). From an ecological perspective, however, efficiency returns are even less. Since the chemical energy content of biological materials represents a crude

but reasonable proxy for the limited net primary productivity underpinning almost every trophic web, gross energy return on investment provides a reasonable first-order approximation of the ecological efficiency with which our food systems supply food energy relative to the demands they place on ecological communities (Pelletier and Tyedmers, 2007). Our analysis suggests low returns (2.0% and 1.6% for feedlot and pastured beef respectively) in all systems modelled.

We should also acknowledge some important limitations to our study. First, the very act of defining "representative" systems masks the variability which exists within and between management strategies, with important implications for apparent environmental performance. For example, following consultation with producers and beef researchers, our models of pasture-based beef finishing in the US Upper Midwest assumed a pasture utilization rate of 60%. While this may, indeed, be a reasonable average for the systems modelled, context-specific utilization rates may range from 30% to 90%, with the highest utilization rates achieved in well-managed temperate pastures where stock are rotated through paddocks daily (Gerrish, 2002). At the high end of this range, our sensitivity analysis suggested that modestly improved resource efficiencies and lower net emissions per unit production would be anticipated. At the low end, impacts per unit production are significantly higher. We also have not considered the potential benefits of organic pasture and feed input production strategies. Research in Ireland suggested that organic beef production may lower emissions and improve resource use efficiencies (Casey and Holden, 2006b), although earlier work in Germany using an LCA-like analysis found no net benefits per unit production (Flessa et al., 2002). We further recognize that pasture-based beef finishing systems elsewhere in the US which have selected for superior genetics and which have longer grazing seasons may have considerably better environmental performance than the Iowa systems we modelled.

Also of critical importance is our use of IPCC Tier 1 default emission factors for modelling field-level emissions related to fertilizer and manure application on pastures and cropland. These are reasonably well-suited to modelling at the macroscale, but mask considerable variability at the microscale. We recommend further research of alternative beef production strategies that applies process-based models such as DAYCENT (Parton et al., 2001) and DNDC (Li, 2000) at a systems-levels in order to develop more nuanced insights of the variability characteristic of context-specific management regimes, taking into account soil and climatic factors.

Finally, given the importance of ruminant methane to overall greenhouse gas emissions in beef production, the choice of methane conversion factor can strongly influence modelling outcomes. For example, although the rations for the feedlot-finishing system contained a high level of concentrates, the total was less than the 90% stipulated by IPCC for use of a 3% methane conversion factor. We hence applied a factor of 5.5%. However, our sensitivity analysis indicated that estimated emissions would be only 6% lower using the 3% factor because the finishing stage contributes only 30% to total emissions for feedlot beef.

5. Conclusions

Life cycle assessment is increasingly used to describe the macroscale environmental dimensions of products and services. By making visible the resource flows and emissions characteristic of specific technologies, it provides a starting point for evaluating certain aspects of the relative environmental performance of competing products and services.

According to the metrics employed in this analysis, it would appear that feedlot-finished beef products are less resource and emissions-intensive relative to management-intensive pastured beef

production in the US Upper Midwest production systems we modelled along four important dimensions of environmental performance. We recognize, however, that in some cases there may be substantial reductions in net greenhouse gas emissions for pasture systems under conditions of positive soil organic carbon sequestration potential (i.e. following changes in land use, but with declining sequestration rates over time). Furthermore, optimally-managed pasture systems would perform better than our modelled "average" system.

We would also stress that none of the systems analyzed can be described as ecologically efficient relative to most other food production strategies. Certainly, our measures of resource returns on investment provide strong indications to the contrary. Moreover, our work does not provide insights into the social and economic dimensions of these activities. For example, we do not consider costs and benefits related to variables like job creation or quality of life, nor do we address a spectrum of proximate ecological considerations, including biodiversity impacts, or concerns such as animal welfare. Our results should therefore not be taken as stand-alone metrics of the sustainability of feedlot versus pasture-finished beef production in the US Upper Midwest. Rather, they are intended to contribute to our necessarily evolving and increasingly nuanced understanding of beef production and food system sustainability issues generally, and offer insights into how the beef production systems considered here might best pursue improved environmental performance.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.agsy.2010.03.009.

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