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Comparison of production-phase environmental impact metrics derived at the farm- and national-scale for United States agricultural commodities

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Abstract

Agricultural production is critical for human survival and simultaneously contributes to ecosystem degradation. There is a need for transparent, rapid methods for evaluating the environmental impacts of agricultural production at the system-level in order to develop sustainable food supplies. We have developed a method for estimating the greenhouse gas (GHG), land use and reactive nitrogen inputs associated with the agricultural production phase of major crop and livestock commodities produced in the United States (US). Materials flow analysis (MFA) and life cycle assessment (LCA) techniques were applied to national inventory datasets. The net anthropogenic nitrogen inputs (NANI) toolbox served as the primary accounting tool for LCA and MFA. NANI was updated to create links between nitrogen fertilizer and nitrogen fixation associated with feed crops and animal food commodities. Results for the functional units kilogram (kg) of product and kg of protein for 2002 data fall within ranges of published LCA results from farm-scale studies across most metrics. Exceptions include eutrophication potential for milk and GHGs for chicken and eggs, these exceptions arise due to differing methods and boundary assumptions; suggestions for increasing agreement are identified. Land use for livestock commodities are generally higher than reported by other LCA studies due to the inclusion of all land identified as pasture or grazing land in the US in this study and given that most of the estimates from other LCAs were completed in Europe where land is less abundant. The method provides a view of the entire US agricultural system and could be applied to any year using publically available data. Additionally, utilizing a top-down approach reduces data collection and processing time making it possible to develop environmental inventory metrics rapidly for system-level decision-making.

1. Introduction

Modern agriculture in the United States (US) has provided a safe and ample food supply for many decades. Agricultural production has increased despite a steady decrease in the number of people employed in the sector due to mechanization, mono-cropping, fossil fuel and chemical inputs (USDA ERA 2005). Agricultural production results in greenhouse gas (GHG) emissions, nutrient, herbicide and pesticide pollution, decreased biodiversity, water resource

depletion and soil erosion (Cassman *et al* 2005). Without careful planning these impacts are likely to increase due to growing world population and affluence, which has been found to correlate with the consumption of animal-sourced foods, known to be more resource intensive than plant-sourced foods (Nellemann *et al* 2009). This study evaluates three important sustainability metrics for the agricultural system: GHGs emissions, reactive nitrogen (N) inputs/outputs and land use. The interdependence of food, energy and water systems, the ecological impacts

created by these systems and the fundamental need for sustenance make agriculture and food systems particularly relevant for ensuring the ability of future generations to thrive.

Reducing N loading to land and waterways while maintaining food production and energy systems has been identified as one of grandest challenges facing modern humanity by the National Academy of Engineering (2008). Fertilizer application and cultivation of nitrogen-fixing crops are the most significant anthropogenic sources of N input to Earth systems with fossil fuel combustion a distant but important second (Howarth *et al* 2005). The impacts of increasing the input of N species to ecosystems include eutrophication of water bodies, in some cases leading to hypoxic conditions, human health effects, e.g., ground-level ozone formation, and addition of nitrous oxide (N_2O), a GHG, to the atmosphere (Galloway and Cowling 2002, Howarth *et al* 2005, Townsend and Howarth 2010, Sutton *et al* 2011a, 2011b). In the US, two-thirds of the nation's coastal rivers and bays are moderately to severely degraded due to nutrient pollution, (NAS 2000, Bricker *et al* 2007) and globally the prevalence of dead zones is growing (Diaz and Rosenberg 2008).

Since record keeping of GHGs began in 1990 the agricultural sector has consistently contributed an estimated 6 to 7 percent (%), 0.45 gigatone (Gt) carbon dioxide equivalent (CO_2e), of total GHG emissions in the US, according to the methodology used by the US Environmental Protection Agency (EPA 2013). Globally, GHG emissions from agriculture were recently estimated at approximately 4.6 Gt CO_2e using a similar approach (Tubiello *et al* 2013). These estimates are based on older global warming potential (GWP) conversions integrated over a 100 yr time period (EPA 2013). Integrated over 20 yr rather than 100 increases this estimate for GHG emissions from agricultural production to almost 13% of US emissions contributing to climate change in 2002. Note that recent re-estimates of GWP multipliers (Shindell *et al* 2009, IPCC 2013) would increase the magnitude of emissions estimates, as would comparing methane's influence with that of CO_2e at time scales of less than a century, as is encouraged by the IPCC (2013) and used in some recent assessments of emissions from the fossil fuel sector (Howarth 2014, 2015).

Of the 920 million hectares that comprise the US roughly 77 percent (%) of the land surface, including Alaska and Hawaii, is occupied in some way for anthropogenic purposes. Of this land 25% may be described as intensive use, i.e., cropland and developed land, with the remaining 75% classified as pasture and timberland (Costello *et al* 2011). Agricultural lands, i.e., cropland and pastureland, make up approximately 33% of land use globally (Bringezu *et al* 2014). The United Nations project an increase in agricultural land use between 7 and 31% (350–1500 Mha) depending on boundary conditions

and assumptions; this expansion often occurs into forested land (Bringezu *et al* 2014).

Evaluation of the environmental sustainability of agricultural systems generally falls into two categories: small-scale, farm-level life cycle assessment (LCA) studies (Cederberg and Mattson 2000, Núñez *et al* 2005, Williams *et al* 2006, deVries and deBoer 2010) and regional- or global-scale material flow analysis (MFA) (Grote *et al* 2005, MacDonald *et al* 2012). LCA is a widely recognized method for holistically quantifying the environmental impacts of producing a commodity from raw materials acquisition to final disposal. MFA is a systematic assessment of the flows and stocks of materials within a system defined in space and time and obeys the law of the conservation of matter (Brunner and Rechberger 2005). The former, often based on the most representative datasets for the local or regional farms, is unable to capture the variability across the system leaving an analyst unable to determine the representativeness of applying the results to the same product produced in another farm system. The latter is typically reported in terms of flows at the national level with little, if any, connection to specific commodities, limiting the ability to understand the problem from the consumption perspective. An understanding of environmental impacts from a consumption perspective is important for considering demand-side management and policies impacting global trade (Peters 2008, Bringezu *et al* 2014). Neither approach provides metrics for assessing the average impacts associated with major agricultural commodities in the US. Distinct from those two categories, Xue and Landis (2010) assessed the nutrient exports of US food systems using a stochastic, bottom-up approach and linked the reduction of eutrophication potential (EP) with food consumption pattern shifts. Additionally, Pelletier and Leip (2013) used chemical databases and life cycle inventory data from the European Union (E.U.) to estimate N flows associated with food consumption of E.U. citizens. Duchin (2005), Weber and Matthews (2008), Costello *et al* (2011) have used input–output LCA methods to estimate environmental impacts of agricultural sectors; however, the commodity categories in these studies are highly aggregated and do not include nitrogen as an impact metric. Despite these valuable efforts, the systematic environmental assessment of agriculture using a top down approach, particularly in the US, is still lacking. This study combines LCA and MFA through a well-documented nutrient accounting tool to develop environmental assessment metrics often used to characterize the sustainability of agricultural commodities.

The approach described herein offers a relatively rapid, comprehensive, transparent and reproducible approach for estimating average environmental impacts for the selected metrics associated with the production of major US agricultural commodities. These national-level metrics can be useful for guiding policy relating to GHG, N pollution and land use in

relation to agricultural production and consumption as well as for identifying the most important areas for targeting best management practices (BMPs). The mathematical relationships developed herein extend the capabilities of the net anthropogenic nitrogen inputs (NANI) toolbox (Hong *et al* 2011, 2013) and are available upon request in Matlab format. Creating metrics from national inventories provides a benchmark for assessing results from farm-scale studies and allows for a clear connection between national production and consumption. Using national-level inventories avoids omission of crop and animal food losses that might not be included using farm-level analysis, e.g., crop spoilage during storage. The approach conserves mass within the limits of data generation and collection methods. Commodities are often combined e.g., at grain elevators, and re-distributed for multiple additional processing stages, e.g., milling, breakfast cereal manufacturing, before final consumption making it nearly impossible to track the specific inputs and outputs associated with an individual final product, e.g., breakfast cereal, soups, frozen meals. For these reasons, national average estimates may be more relevant than farm-scale values when assessing the impacts of agricultural production and food at the system level.

2. Methods

This study quantifies N inputs (fertilizer-N, fixation-N) and outputs (manure-N, ammonia ($\text{NH}_3\text{-N}$)), GHGs, and land occupation, hereafter referred to as environmental impacts, associated with the production of major US agricultural commodities for the year 2002. Note that we chose 2002 since the NANI toolbox had originally been presented for that year (Hong *et al* 2011) and it is easiest to present this new methodology for a particular year. However, the methods developed herein can be applied to any study year. The NANI toolbox, described in more detail below, provided the majority of crop and livestock production and nitrogen input and output data. Principles of MFA and LCA were applied to create the necessary links between crops and livestock and to account for ethanol and co-products. Two functional units were selected for comparison with relevant LCA results, per kg of edible product and per kg of protein (Schau and Fet 2008, deVries and deBoer 2010). This study does not account for energy and resources used to produce inputs to farm systems, e.g., fertilizer manufacture, farm equipment nor energy used at the farm or in food processing. The production phase in agricultural products has been found to be the largest single contributor in LCAs of agricultural commodities with regard to GHG emissions (Weber and Matthews 2008, Camargo *et al* 2013), nitrogen inputs (Howarth *et al* 2002, 2012, Pelletier and Leip 2013) and land occupation (Costello *et al* 2011). The output from

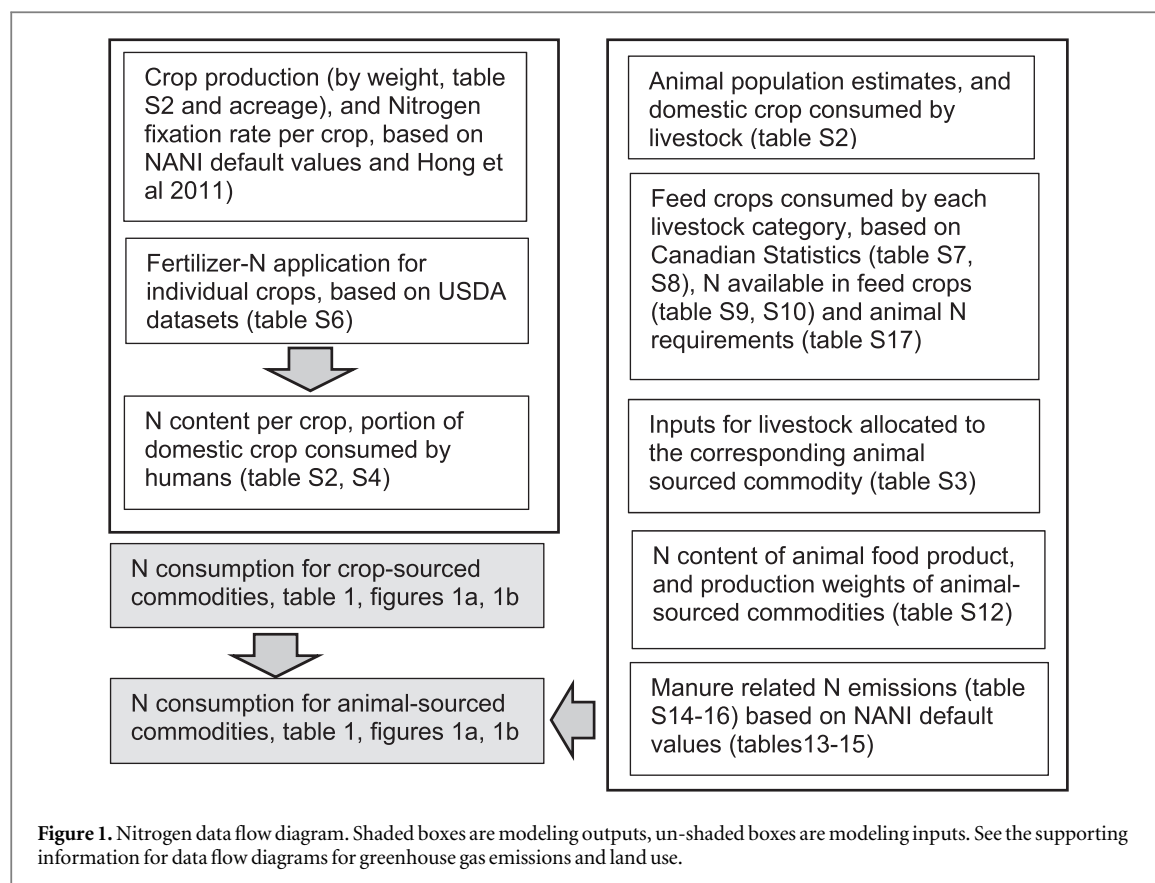
this study could easily be used in conjunction with energy use data at the farm or national scale and upstream impacts could be incorporated for use in a full LCA.

2.1. NANI toolbox: crop production, livestock populations and nitrogen

The structure of the NANI toolbox was used as the starting point for developing the environmental impact estimates described in this work. Prior to this work there were no explicit links between crops and livestock in the NANI toolbox. These links are crucial for estimating life-cycle impacts in the production phase and for tracking the flow of and accumulation of impacts through the production supply chain. Figure 1 describes how data flow is organized in this analysis and table S1 includes all of the equations used in the analysis.

The NANI toolbox includes national crop production (both weight and acreage), animal population estimates, and the majority of data used in the nitrogen portion of this analysis for study year 2002 (Hong *et al* 2011). The links between crops and livestock developed within NANI were also used to determine GHG and land use embodied in the commodities investigated in this analysis. NANI estimates total net nitrogen introduced to a region due to human activities and has been proven to be a good predictor of riverine nitrogen flux across hundreds of watersheds in the US and Europe (Howarth *et al* 1996, 2012, Alexander *et al* 2002, Boyer *et al* 2002, Schaefer and Alber 2007, Schaefer *et al* 2009, Billen *et al* 2011, Hong *et al* 2011, Grizzetti *et al* 2011, Hong *et al* 2012). NANI includes fertilizer-N, fixation-N, atmospheric deposition and net food and feed. Net food and feed is calculated as: (human and animal nitrogen requirements) minus (N in crop and animal products). Atmospheric deposition of NO_y , associated with energy use in the agricultural sector was not included, as NO_y deposition from all sources in the US—including that from energy use in the agricultural sector—is very small compared to nitrogen fertilizer inputs (Hong *et al* 2011). NO_y represents oxidized nitrogen compounds deposited to the land surface and is inclusive of NO_x , N_2O_5 , HNO_3 , plant available N, aerosol nitrates and other organic nitrate. Agricultural commodities included in this analysis are: corn, corn silage, wheat, oats, barley, sorghum, sorghum silage, potatoes, rye, alfalfa, other hay, soybeans, rice, beef, pork, milk, eggs, chicken and turkey. Estimated biomass from cropland and noncropland pasture are also available in the NANI toolbox.

Additional data from the NANI toolbox include: N content for each crop and nitrogen fixation rates per crop (table S4) (Hong *et al* 2013), distribution of crops between humans and livestock (table S2), and N consumption requirements for livestock, manure N and ammonia emission rates for each livestock category



(table S5). The NANI toolbox includes fertilizer input by county based on sales data compiled by the USGS with no individual crop specification (Ruddy *et al* 2006). Therefore, fertilizer associated with individual crops was calculated. National average fertilizer application rates over the most recent five-year periods (USDA 2012a; table S6) were multiplied by harvested acreage data to estimate total fertilizer associated with the production of each crop in the study year. Fertilizer-N use associated with each crop was compared to total fertilizer-N usage data collected by the USDA where possible, table S6 (USDA 2012b).

To establish domestic crop availability the exported quantities were subtracted from total annual production (USDA 2012c). Next, corn utilized for ethanol was subtracted from domestically available corn. The amount of co-product generated during ethanol production and used as animal feed was estimated as described below. Domestic crops were then allocated to humans and livestock using the assumptions included in the NANI toolbox, table S2. A summary of total crop production and animal population values are provided in the supporting information (S.I.), tables S2 and S3. It is critical to estimate the portion of grain, oilseed and forage production allocated to individual livestock types in order to establish the environmental impacts of animal-sourced commodities.

The portion of each individual crop consumed by each livestock category was approximated using Canadian data, table S7 (Statistics Canada 2003). This

Canadian study is the only document found that quantifies feed dispensed to livestock in units of dry weight of whole grains, oilseeds, silage and roughage, rather than livestock nutrient requirements or nutrient content of processed feeds. Rather than use the Canadian estimates directly, the Canadian data were used to generate allocation factors that were applied to US data. First, the ratio of crop N to forage N by livestock type were calculated using the Canadian data, table S8. Next, specific crops and forages consumed by an individual animal to meet N requirements were estimated using the percent contribution of N from each crop to the total N available in all crops, D_c , or forage, D_f , for livestock, see equations (1) and (2) below and table S9 for allocation factors. Ethanol feed co-products were accounted for directly using the percentages consumed by livestock type reported by the Renewable Fuels Association (RFA 2003). The nitrogen content of each domestic feed source was calculated using N content assumptions from NANI, table S4; N content of ethanol co-products is described below. It was assumed that swine and poultry do not consume any forage materials, as is typical in the US (Wheaton and Rea 1993, USDA 2012f). For example, the N requirement for dairy cattle of $130.8 \text{ kg N yr}^{-1} \text{ animal}^{-1}$ was fulfilled with 60% grain and 40% forages based on Canadian data, table S8. The amount of N embodied in ethanol co-products fed to each dairy cattle is estimated to be 10.2 kg N , see details below and in the S.I. The amount of N in these co-products

was subtracted from the total animal N requirement before fulfilling the remaining N requirement with crops and forages as described by equations (1) and (2). An estimated 36.8% (table S9) of the nitrogen available to livestock from grain/oilseed crops is provided by corn, therefore it is estimated that 26.6 kg N from corn crop is fed to each dairy cow per year (table S10)

$$D_c = N_i / \sum(N_i), i = \text{crop}, \quad (1)$$

$$D_f = N_j / \sum(N_j), j = \text{forage}. \quad (2)$$

Feed estimates for livestock are used to quantify corresponding fertilizer-N and fixation-N and ultimately assigned to animal-sourced commodities. The annual total livestock populations in 2002 were determined using a dynamic approach, which takes into account the life cycle of farm animals during the year, table S3 (Kellogg *et al* 2000, Han and Allan 2008, Hong *et al* 2011). Since the functional units are kg and kg protein of edible commodity, crops and environmental impacts associated with the total livestock population were summed according to the animal-sourced commodity supported by the total population, e.g., beef breeding herd crop consumption is allocated to beef, table S3. Production weights of animal-sourced commodities (e.g., beef, eggs) were taken directly from USDA statistics (USDA 2012d), table S12, to avoid introducing another source of uncertainty by using conversion factors to estimate meat, milk and egg production from livestock population data. Protein content of each crop and animal food product included in the analysis was taken from the USDA nutritional data, assumptions are included in tables S4 and S12 (USDA 2012e).

The environmental impacts associated with corn-based ethanol production were split between ethanol and feedstocks using a weight-based allocation assumption. The weight-based allocation is the most widely utilized allocation approach in the LCA literature. Allocation factor assumptions are as follows: wet milling: 48% to feed byproducts and 52% to ethanol and dry milling: 49% to feed byproducts and 51% to ethanol (Shapouri *et al* 2002). For example, one kilogram of wet-milled corn requires 0.019 kg of fertilizer-N inputs, 0.009 kg N were allocated to feeds for livestock and 0.01 kg N were allocated to ethanol. In 2002, 3.8% (8.23 million metric tons (Mmt)) of domestically produced corn entered the wet milling process and 5.6% (12.4 Mmt) entered the dry milling process for the production of ethanol (USDA). Wet milling produces 0.22 kg corn gluten feed (3.7% N, dry weight) per kg of corn and 0.04 kg corn gluten meal per kg of corn (10.5% N, dry weight) (O'Brien and Woolverton 2009, Liu 2011). Dry milling produces 0.28 kg dry distillers' grains solubles per kg of corn (4.4% N, dry weight) (Liu 2011). Combined, in 2002 ethanol production yielded 0.26 Mmt of nitrogen in feed co-products. See the S.I. for more details.

Fertiliser-N and fixation-N associated with each animal-sourced commodity were determined using the feed assumptions derived above and national average fertilizer-N and/or fixation-N per kg of produced crop, table S13. Nitrogen contained in manure and ammonia ($\text{NH}_3\text{-N}$) associated with livestock was also calculated using the rates per animal applied in the NANI toolbox, table S5 (also see tables S14–16). Note that in net nitrogen accounting frameworks, like NANI, manure-N and $\text{NH}_3\text{-N}$ are not included in the calculation because these sources of N are already accounted for in fertilizer-N and fixation-N (taken up into grains, oilseeds and biomass) associated with feed crop production and would lead to double-counting (Hong *et al* 2011). However, since inclusion of all four species is typical in LCA studies, due to the study boundary being at the farm-scale, we have included them for comparison purposes. In order to compare results from this study to those reported by other LCA analyses, all four nitrogen species were multiplied by 25%, the approximate amount of net N observed to exit as riverine flux (Howarth *et al* 2012); this estimate is then multiplied by the phosphate-equivalent EP normalization factor for nitrate, 0.1 (Norris 2003), the predominant N species in agricultural runoff (Goolsby and Battaglin 2000). Note that the variation in N runoff as a function of anthropogenic N inputs at the field scale can vary from 3% to 80% (Howarth *et al* 2002). However, when considering more aggregate scales, empirical studies across hundreds of watersheds in Europe and the US indicate that the flux of nitrogen in rivers is on average equal to 25% of nitrogen inputs (Howarth *et al* 2012, Hong *et al* 2013).

2.2. GHG emissions

GHG estimates associated with the production of major agricultural commodities were taken from the USDA (2011) and the EPA (2013), which both utilize the same underlying methodology. GHGs associated with the production of feed crops and forages are allocated to livestock and ultimately animal-sourced commodities, following the methods described above for N. GHG estimates were adjusted to reflect recent re-estimates of GWP multipliers that better account for the interaction of methane with aerosols, 105 and 33 on a mass-to-mass basis for the 20 and 100 yr timeframes, respectively, with an uncertainty of plus or minus 23% (Shindell *et al* 2009). GHG emissions for the 20 yr and 100 yr timeframe are quantified to better demonstrate the significance of methane emissions in the shorter timeframe. Although the GWP of methane in life-cycle analyses has in the past usually been taken for an integrated 100 yr period, this choice is quite arbitrary (IPCC 2013), and shorter time scales may be more appropriate given the role of methane in expected global warming over the next 15–35 yr (Shindell *et al* 2012, Howarth 2014, 2015).

First, GHGs associated with soil management (e.g., nitrogenous fertilizer application) and field burning were assigned to all crop commodities included in the analysis either directly, where GHGs were reported by crop or through allocation. Second, the crops consumed by livestock were used to estimate GHGs associated with animal-sourced commodities. The USDA Inventory reports more detail than the EPA GHG Inventory and specifies N₂O emissions for corn, hay, wheat, soy, sorghum, cotton and provides a single estimate for all other non-major crops. The non-major commodities include oats, barley, potatoes, rye, and rice and differentiate between alfalfa (a nitrogen fixing plant) and other hay. Anthropogenic N inputs are highly correlated with N₂O emissions (IPCC 2006b, Del Grosso *et al* 2008) and thus fertilizer-N was used as a proxy for allocating USDA-estimated N₂O emissions to non-major crops. Cropland and noncropland pasture were assigned GHG values from the pasture and grazing estimate in the USDA GHG Inventory (2011) by allocating to each category using production estimates from NANI. Methane and N₂O emissions associated with enteric fermentation and manure (both managed in confined operations and deposited on pasture- and rangelands) are reported by animal and were directly assigned to each animal-sourced commodity (EPA 2013). For specific details regarding the EPA's GHG emissions methodology, see the EPA GHG Inventory (2013). Similarly, methane emissions due to Field Burning are reported by specific crop. In summary, GHG estimates for animal-sourced commodities include emissions associated with crop production for animal feed as well as enteric emissions and manure management.

2.3. Land use

Land use is a complex issue and can be evaluated in numerous ways. Researchers in the field of sustainability describe land use in terms of occupation and transformation (Lindeijer 2000). The impacts of occupation (e.g., nutrient pollution, GHG emissions) have to do with consequences of activities occurring in the present, while the impacts of transformation (e.g., release of soil carbon, disturbance of migration paths) have usually occurred in the past. Numerous efforts have been taken to develop metrics to describe the impact of land use, some based on biodiversity (Koellner and Scholz 2008, Schmidt 2008), soil organic matter (Canals *et al* 2006), environmental vulnerability indices, and weighting schemes based on degree of conversion from a natural state (Lindeijer 2000, Lenzen and Murray 2001). In this study the metric chosen is occupation of land use in the study year, in units of area, for the production of a particular agricultural commodity. If a researcher is more interested in applying a weighting scheme or some other form of analysis, the quantity of land utilized for a particular purpose provides a useful starting point.

The USDA tracks both planted and harvested acreage each year, and these values are included in the NANI dataset. The difference can be large or small depending on the events of the year, e.g., drought, price fluctuations, etc. Planted acreage for each crop was divided by total production indicated in NANI. The total area classified as pasture or rangelands in the US is 2.92 million square kilometers (km²) including grasslands, pasture and grazed forested land; this value was used to calculate land occupation associated with pastureland (Costello *et al* 2011). Land area associated with each animal food product is the sum of both area occupied by animal operations (USDA 2002) and land area occupied to produce animal feeds.

3. Results

The total nitrogen inputs/outputs, GHGs and land occupation for evaluated agricultural commodities produced for consumption by the US population are presented on table 1. The values for crop commodities represent the portion of domestic production allocated to humans for direct consumption. Values for animal-sourced commodities are inclusive of the crops and forages consumed by livestock. Crops allocated to direct human consumption account for 6.6% of fertilizer-N and fixed-N, ethanol after allocation is estimated to account for 3.3% of fertilizer-N, the remainder of fertilizer-N and fixed-N is allocated to livestock production. Pork and beef production correspond to the highest fertilizer-N inputs, followed by chicken and milk, due to large quantities of feed consumed to produce these commodities. Beef production is the largest driver for nitrogen fixation primarily due to ingestion of alfalfa hay (60%) followed by soybean cultivation (30%). Pork is the next largest driver for this category and is 100% due to soybean cultivation, as is the case with chicken, eggs and turkey. Nitrogen fixation associated with milk production is due to soybean cultivation (62%) and alfalfa (31%). Note that litter for housing and bedding purposes were not included in this study, but may be relevant (Williams *et al* 2006).

GHG emissions are much larger for animal-sourced commodities due to manure management, enteric fermentation and large quantities of feed consumed. Comparison across products using the 20 yr GWP values highlights the significance of CH₄ emissions in the nearer term. A recent Australian study found that agriculture, and ruminant livestock in particular, become a more significant contributor and, also offer more potential for mitigation efforts, when analyzed using 20 yr GHG multipliers (Wedderburn-Bishop *et al* 2015). It should be noted that CH₄ emissions might be larger than estimated here and in most GHG emissions, perhaps by a factor of 2, according to a recent study that used spatial monitoring data for atmospheric methane across the US in comparison to

Table 1. Summary of select environmental metrics for major agricultural commodities associated with human consumption in the US in 2002^a.

Nitrogen inputs/outputs (Gg) ^b					Greenhouse gases (Tg CO ₂ e)												Land occupation (Mha)			
Commodity	Fertilizer	Fixation	Manure	Ammonia	Soil management		Enteric fermentation		Manure management				Total		Crops	Hay	Pasture	Livestock operations		
					100 yr		20 yr		100 yr	20 yr	100 yr		20 yr						100 yr	20 yr
					N ₂ O	CH ₄	N ₂ O	CH ₄	CH ₄		N ₂ O	CH ₄	N ₂ O	CH ₄						
Corn	120	—	—	—	1.4	—	1.4	—	—	—	—	—	—	1.4	1.4	1.0	—	—	—	
Wheat	330	—	—	—	2.8	—	2.7	—	—	—	—	—	—	2.8	2.7	6.1	—	—	—	
Oats	2.9	—	—	—	0.01	—	0.01	—	—	—	—	—	—	0.01	0.01	0.12	—	—	—	
Barley	3.0	—	—	—	0.01	—	0.01	—	—	—	—	—	—	0.01	0.01	0.05	—	—	—	
Potatoes	120	—	—	—	0.6	—	0.5	—	—	—	—	—	—	0.6	0.5	0.50	—	—	—	
Rye	0.9	—	—	—	0.004	—	0.004	—	—	—	—	—	—	0.004	0.004	0.09	—	—	—	
Soybeans	1.7	60.4	—	—	0.3	—	0.3	—	—	—	—	—	—	0.3	0.3	0.37	—	—	—	
Rice	140	—	—	—	0.7	6.4	0.7	20.3	—	—	—	—	—	7.1	21.0	0.78	—	—	—	
Beef	1030	1920	3500	680	53.3	2.5	51.7	8.0	170	530	7.7	4.4	7.5	14.0	235	610	9.8	15.3	119	10.5
Milk	580	590	990	240	14.6	0.4	14.2	1.3	50	154	5.7	33	5.5	105	104	280	7.0	2.6	20.4	11.1
Pork	1040	880	410	210	16.3	—	15.8	—	2.2	7.0	1.6	29	1.6	93	49	120	15.3	0.0	0.0	3.4
Eggs	180	150	230	100	2.8	—	2.7	—	—	—	0.02	0.7	0.0	2.3	3.5	5.0	2.6	0.0	0.0	0.4
Chicken	770	660	1020	440	12.1	—	11.7	—	—	—	0.1	3.3	0.1	10.4	15.5	22.1	11.3	0.0	0.0	1.9
Turkey	160	140	190	80	2.6	—	2.5	—	—	—	0.006	0.3	0.0	0.8	2.8	3.3	2.4	0.0	0.0	0.2
Ethanol	200	0	0	0	2.3	—	2.2	—	—	—	—	—	—	—	2.3	2.2	1.5	—	—	—

Notes:

^aThese values represent impacts as a function of human consumption of these commodities, not the total impacts associated with production of each commodity. The greenhouse gas, nitrogen inputs/outputs, and land occupation values for animal-sourced commodities are inclusive of the crops and forages consumed by livestock.

^b All values are in units of nitrogen, i.e., Gg of nitrogen in manure excreted.

EPA inventory estimates (Miller *et al* 2013). Beef production is the leading source of GHG emissions across agricultural commodities with enteric fermentation and feed production dominating. Cattle for beef production result in the largest quantities of manure produced (table S15), but relatively small N₂O emissions reflecting higher percentages of nitrogen in swine and poultry manures and higher instances of confinement in swine and poultry operations (Kellogg *et al* 2000). Since dairy cows, swine and poultry are housed for the majority of their lives, and methane emissions are larger in managed manure systems than in pasture systems (IPCC 2006a), the contribution of GHG from manure is relatively large.

Land area occupied for the production of each commodity follows similar trends with beef, milk and pork dominating. Land occupation is broken into four categories: crop, hay, pasture and area occupied by livestock operations. The majority of land occupied for the production of animal-sourced commodities is occupied for crops, hay and pasture. Total cropland utilized for the production of animal-sourced commodities is about six times that of cropland used to directly feed humans. Pastureland estimates reflect the assumption that all of the land classified as grazing lands by the USDA are occupied (see methods), data are lacking regarding the extent of grazing occurring on these lands, particularly at the national level. Additional research is needed to assess the extent of use on these lands.

Results for per kg product or per kg protein (figures 2(a)–(h)) again demonstrate that GHG, N inputs/outputs and land occupation associated with animal-sourced commodities are larger than for crop commodities for human consumption, with the exception of GHG emissions for rice compared to poultry products (figures 2(c)–(f)). Results based on kilocalorie normalization are included in the S.I. This is consistent with findings from similar studies and reflects the impact of using resource-intensive, cultivated feeds for the production of animal-sourced foods and direct GHG emissions associated with ruminants and livestock manure management. Per unit product GHG emissions are largest for beef and pork despite total GHGs associated with milk production being larger than that of pork production. The gap between pork and milk closes when viewed on a per unit protein basis, reflecting the high water content of milk. Enteric emissions of methane constitute the largest contribution of GHG emissions for beef and milk. GHG emissions from manure management are large for milk and pork due to high rates of confinement. Observing the 20 yr GWP values, beef is still the largest GWP contributor.

Land area occupied per kg of product are higher for animal-sourced commodities per kg and per kg protein, with the exception of rye having a higher land use per kg of protein than turkey; this is because the area harvested for rye was about 20% of the area

planted in 2002. Roughly 40% of the area planted with oats was harvested, for all other crops 75% or more of the planted acreage was harvested. Note that rye, oats, and barley represent a very small fraction, <1%, 2%, and 2%, respectively, of the acreage cultivated for grains and soybeans in the US and thus do not impact results significantly.

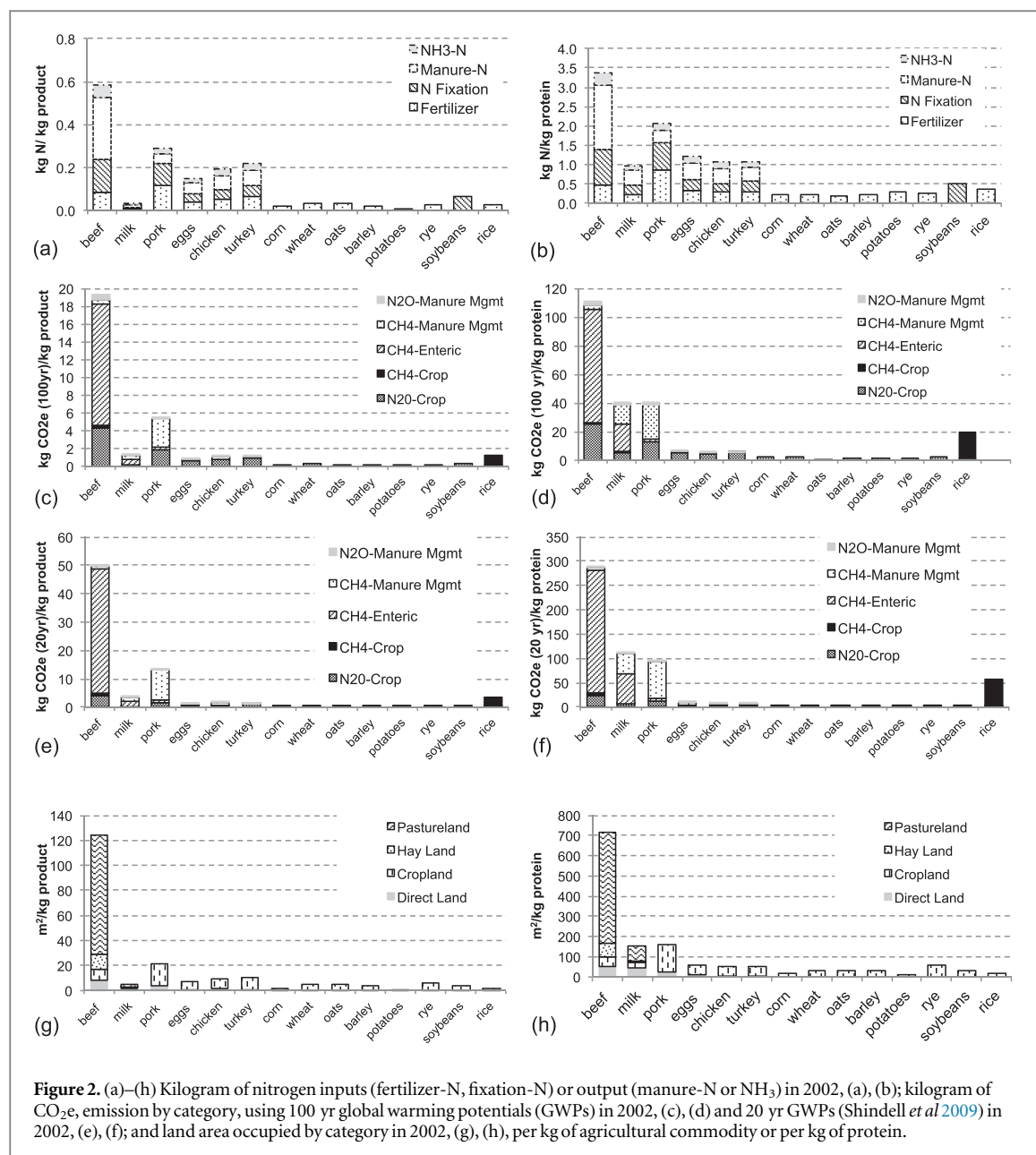
Animal-sourced commodities result in greater N inputs over all four N categories than that of crop commodities per kg of product and per kg of protein delivered with beef associated with the largest sources of N inputs. However, pork production causes more fertilizer-N inputs per kg of protein than beef. Manure and ammonia are not included when calculating NANI because this N is already accounted for in the feed consumed by the livestock (the N in feed is accounted for by fertilizer-N and fixation-N), thus including them would result in double-counting. In order to more directly compare these results to LCA results, N contained in manure and ammonia per unit of animal-sourced commodity are shown on figures 2(a) and (b), as these sources are often, but not always, included in farm-scale analyses.

All manure-N produced due to livestock in the US was accounted for in defining the N and GHG impact metrics. The EPA GHG Inventory accounts for the myriad of management options utilized by livestock producers in the US and the corresponding variation in GHG emissions. The N metric does not explicitly account for the variation in manure management; rather it reflects how much manure-N is generated per animal or commodity. Since not all manure is managed the same actual environmental impacts may vary significantly across individual farms and should be investigated in an individual farm-scale LCA.

4. Discussion

4.1. Reliability of data and partial sensitivity analysis for some major assumptions

In any LCA study, and particularly for agricultural systems operating on massive scales within a highly complex and dynamic environment, there are numerous sources of uncertainty and variability. All of the underlying data used in this analysis are generated by government agencies and are transparent for future studies. One should be cognizant that the methods and models employed are in a constant state of improvement, e.g., the EPA GHG Inventory or annual USDA survey data. The uncertainty generated through assumptions and methods employed within this study include allocation of crops to humans and livestock, N excretion rates from livestock, N consumption requirements of livestock, protein content of animal-sourced commodities and the mix of crops that make up livestock diets and nutrient content of commodities. Where possible, we have cross-checked estimates against other data sources; additional detail



is provided in the S.I. For example, total manure estimates were compared to reported estimates from 1997 (Kellogg *et al* 2000), table S15, beef cattle, dairy cattle and swine estimates using the NANI assumptions compare well with -2% , -9% and 24% differences compared to the Kellogg estimates; however, the manure estimates for poultry indicate a percent difference of -61% suggesting that poultry manure estimates may be underrepresented in this analysis. The approach for estimating fertilizer-N use per crop was compared to an independent dataset, as described above. As noted throughout this paper, the law of mass conservation is observed with respect to crop and livestock production and the corresponding environmental impacts for the study year. All of the environmental impact metrics are derived from national datasets and peer-reviewed data.

A comprehensive sensitivity analysis is well beyond what can be presented in this single paper. Here, we have presented a partial sensitivity analysis for the animal N requirement value, which drives the allocation of crops to livestock and ultimately environmental impact metrics for animal-sourced commodities, figure 1. The manure and ammonia emission per animal estimates used in this study were also varied using the range of values reported in Boyer *et al* (2002), David *et al* (2010) and van Horn (2000), tables S14 through S16, and per unit estimates for environmental metrics were recalculated, the range of values are reflected in table 2.

Total feed for livestock estimated using the approach developed herein compares well to the quantity of crops allocated to livestock in the NANI toolbox, tables S11. The total estimated crops required

Table 2. Comparison of results from this study to similar published life cycle analysis results.

Eutrophication potential ^a kg PO ₄ ³⁻ e /kg product				
Commodity	Fertilizer-N + Fixation-N	Fertilizer-N, fixation-N, manure and NH ₃	Xue and Landis (2010)	deVries and deBoer (2010)
Beef	0.006–0.009	0.014–0.018	0.02–0.14	0.075–0.33
Milk/dairy ^b	3.2E-04–5.7E-04	6.1E-04–1.2E-03	0.011–0.029	0.003–0.12
Pork	0.001–0.006	0.003–0.01	—	0.003–0.20
Eggs	0.002–0.003	0.002–0.006	—	0.025–0.064
Chicken	2.5E-04–0.003	0.001–0.006	0.02	0.004–0.08
kg CO2e/kg product ^c				
Commodity	Feed, manure, enteric	deVries and deBoer (2010)		
Beef	18.9–21.1	17–32		
Milk/dairy ^b	1.3–1.4	0.9–1.4		
Pork	4.2–5.7	4.3–12		
Eggs	0.8–1.1	3.9–4.9		
Chicken	0.3–1.3	2.6–6.9		
Land use m ² /kg product				
Commodity	Occupied by farm operation	Feed	Total	deVries and deBoer (2010)
Beef ^d	8.6	105.5–156.5	114.1–165.1	27–49
Milk/dairy ^{b,d}	1.4	2.9–5.6	4.4–7.1	1.1–2.0
Pork	3.8	4.5–19	8.3–22.8	8.9–12.1
Eggs	1.0	5.8–9.2	6.7–10.2	4.5–6.2
Chicken	1.3	0.8–10.6	2.1–11.9	8.1–9.9

Notes:

^a EP was calculated using the sum of nitrogen inputs (fertilizer-N and fixation-N), the assumption that 25% of N input to land is exported to riverine systems (Hong *et al* 2012) and the 0.1 conversion factor for nitrate (Norris 2003) the predominate species in runoff.

^b This study and deVries and deBoer (2010) evaluated milk, Xue and Landis (2010) evaluated numerous dairy products.

^c The older Global Warming Potential value for methane of 21 from the USDA (2011) and 24 from the EPA (2013) were preserved in order to be most compatible with the deVries and deBoer (2010) reported results.

^d All lands classified by the USDA as pasture and grazing lands are assigned to livestock, while the comparison studies include only area associated with consumption of specific amounts of vegetation. Data from this study are from 2002. Data from other studies were published over the period of 1997–2009.

to meet US livestock population needs are about 7% lower than the estimated quantity of crops allocated to livestock for grains and oilseeds and 27% lower for alfalfa and other hay. Since no waste or spoilage factor was applied to crops, it is likely that differences could be accounted for in this way; MacDonald *et al* (2012) estimated 4% and the NANI toolbox assumes 10% crop waste (Hong *et al* 2011, 2013). The larger discrepancies for hay are assumed to reflect the possibility that these materials may be made available for dietary and bedding reasons that do not meet nutritional requirements. When the highest animal N requirement value, representing a higher bound for feed intake, from the literature was used to estimate feed consumption per animal the crop consumption was estimated to be ~25% greater than the estimated quantity of crops allocated to livestock and in most cases is greater than all domestic crop production, thus it can be assumed that the high estimates for animal N requirements are unlikely to apply across all livestock categories, table S11. Likewise, the application of the lowest animal N requirements leaves ~57% of crops allocated to livestock unutilized. This is unlikely to

reflect reality, for example, 74% of domestic corn production in 2002 was directly used for feed purposes (USDA 2013) with large quantities of feed byproducts resulting from milled grains.

4.2. Comparison with LCA studies

Table 2 compares the GHG, EP and land use values found in this study with similar values from a recent review of agricultural and food LCAs (deVries and deBoer 2010) and Xue and Landis (2010, EP only). Xue and Landis (2010) estimated EP values using inventory data from SimaPro (Pré Consulting) and GREET (Burnham *et al* 2006) and relevant literature; a range of estimates were determined by sampling distribution functions generated from all available values for each stage of production. Additional information regarding the criteria for inclusion of studies in de Vries and de Boer as well as details regarding the calculation of EPs can be found in the S.I.

The approach and data sources for quantifying EP vary across LCA studies highlighting the need for transparency and the value in deriving system-wide averages that enable one to put the farm-scale results

into a broader context. The range of EP values observed in this study is narrower than the findings presented by deVries and deBoer (2010) and Xue and Landis (2010). The wider range of EPs reported in the LCA studies evaluated as compared to this study result from the different approaches for estimating nitrogen releases to riverine systems, variation in eutrophying species included in the calculation of EP, and the variation in the specific feeds and locations modeled, see below and the S.I. for more details.

In many LCA studies, a nutrient balance is calculated according to the physical boundary of the farm (farm-balance) and may or may not also include nutrient inputs associated with purchases from off-farm activities (e.g., fertilizer-N applied to produce feed). Inputs to the farm-balance typically include fertilizer-N inputs and fixation-N associated with cultivation on the farm, nutrients in purchased feed concentrates, nutrient content of imported livestock, and atmospheric deposition of N species to the farm (Thomassen *et al* 2008). Total nutrient inputs are then normalized using an EP normalization factor (Norris 2003). Outputs include nutrient content of animal food products, (e.g., carcass, milk, eggs), exported roughage and feed produced on-farm and manure (Thomassen *et al* 2008). Other studies use physical models that estimate nutrient fluxes given farm characteristics (Cederberg and Matteson 2000, Cederberg and Flysjö 2004, Basset-Mens and van der Werf 2005, Williams *et al* 2006). Another commonly used approach applies empirically derived multipliers to the net nutrient farm-balance to estimate N leakage (Mollenhorst *et al* 2006, Thomassen *et al* 2008), this is conceptually similar to the NANI approach at a different scale.

In addition, not all of the studies included the same eutrophying species. Phosphorous species (not accounted for in this study) contributed between zero and 63% of total EP (table S18) in studies included in deVries and deBoer (2010) and Xue and Landis (2010). Due to the transparent nature of the data and results included herein, addition of P could be completed and incorporated in the future. Furthermore, Xue *et al* grouped milk, butter, cheese into a single food category, whereas dairy estimates in this study only include milk, with the nutritional value observed at the farm-gate. This resulted in a higher environmental impact per unit of food in the Xue dairy category. In addition, as noted previously, this study estimated lower manure for poultry values than a national manure estimate (Kellogg *et al* 2000), this could be significant.

GHG estimates from this study were within the range estimated by de Vries and de Boer for beef, milk and pork, and less than the range reported for eggs and chicken (2010). Additional insight into the difference observed in the chicken and egg categories is needed; however, the LCA studies used for comparison did not provide sufficient detail for each stage included to

identify the key areas where the difference originates. The high values in the de Vries and de Boer are larger than the high value found in this study, most notably for chicken; this is not particularly surprising since this study did not account for upstream and on-farm energy consumption or energy used in food processing. However, it is important to note that numerous studies have found that the largest contributor to life cycle GHG emissions from cradle-to-farm gate for agricultural commodities originate from the production stage (Weber and Matthews 2008, Camargo *et al* 2013). Camargo *et al* found that on-farm energy use contributed approximately 14% of life cycle GHG emissions and fertilizer manufacturing another 16% while N₂O emissions from soils averaged 44% across all crops (2013). Note that the older 100 yr GWP values for methane of 21 from the USDA (2011) and 24 from the EPA (2013) were preserved in this comparison in order to be most compatible with the de Vries reported results. Land use values are within the same order of magnitude though higher than those in the de Vries and de Boer study. The larger values for beef and milk compared to other LCA results reflect the assumptions in this study about pastureland. The relative abundance of land in the US as compared to other study areas (Europe and New Zealand) may account for some of this discrepancy. Very few of the LCA studies provided sufficient detail to convert values to the original GHG species, nutrient inputs or particular land use types. For further discussion regarding the differences in studies used for EP comparison see the S.I.

Farm-scale analyses are useful for comparing specific management options and deepening our understanding of the full range of variability within the overall system. With enough of these studies and corresponding location and time-dependent information it would be possible to develop stochastic methods to fully characterize that variability and explore correlations between a particular environmental impact and site-specific conditions e.g., ammonia volatilization and air temperature, or nitrate leaching and soil characteristics to further understand important factors associated with actual impact incurred with spatial specificity. However, developing site-specific datasets at the farm-scale is time- and resource-intensive and could delay decision-making efforts to increase the environmental sustainability of agricultural systems in the US. Since US commodities are highly aggregated it is nearly impossible to identify the specific location and environmental impacts of any individual food item, which highlights the need for system-wide impact metrics. Novel cultivation practices such as organic farming constitute a small portion of the market share for commodities included in this study (USDA 2014) and thus inclusion of these practices would not be expected to change the findings. Thus system-wide estimates for each commodity are manageable and useful for evaluating the majority of foods

available in the US and complimentary to LCA research conducted at the farm-scale.

The top down approach presented in this study may serve as the scientific foundation for policy aiding in the areas of diet (consumption), agricultural subsidies, and targeting the agricultural production stages in most need of BMPs. Compared with farm-scale analyses, our top-down approach provides an explicit link between the environmental impacts (GHG emissions, nitrogen releases, and land use) of agricultural systems and national consumption patterns of food/crop groups. The top down approach is effective for identifying the environmental hotspots among various food groups and production stages for future improvement. Due to the aggregated nature, the top down approach may not be appropriate for suggesting improvements of BMPs for individual crops or particular farms.

The dynamic nature of agricultural and food systems make these systems good candidates for developing methods to incorporate space and time with respect to ecological impacts. As research into agricultural and food systems progresses, life cycle impact assessment results should be reported in terms of basic physical units with as much spatial and temporal specificity as possible and all assumptions should be explicitly stated. This would begin to create the opportunity for developing methods to more accurately reflect real damages associated with the production and subsequent consumption of particular goods. The top-down approach outlined herein provides a relatively rapid method for estimating national-scale metrics for evaluating the environmental performance of the US agricultural system. Further, a framework in which stochastic methods could be integrated has been provided, which could be used to link top-down and bottom-up LCA approaches.

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