United Egg Producers
Leadership by Egg Farmers for Egg Farmers

June 17, 2022

The Honorable Gary Gensler, Chair
U.S. Securities and Exchange Commission
100 F Street, NE
Washington, DC 20549

Submitted via email: rule-comments@sec.gov

SUBJECT: Comments on the Proposed Rule Titled "The Enhancement and Standardization of Climate-Related Disclosures for Investors" (File Number S7–10–22)

Dear Chair Gensler:

The United Egg Producers (UEP) appreciates the opportunity to provide the following comments on the Securities and Exchange Commission's (Commission) proposed rulemaking entitled "The Enhancement and Standardization of Climate-Related Disclosures for Investors," (file number S7-10-22), and published in the Federal Register on Monday, April 11, 2022.

UEP opposes the Commissions' proposed mandatory disclosure requirements as the means to supply the important and worthwhile information that the Commission hopes to generate through this rulemaking. We do this while supporting the Commission's stated objective for the rule (to "provide consistent, comparable, and reliable—and therefore decision-useful—information to investors to enable them to make informed judgments about the impact of climate-related risks …"). Instead of the extensive and mandatory prescriptive disclosures proposed, we urge the Commission to develop and issue additional, principles-based GHG emissions reporting guidance suitable for and targeted towards registered entities subject to SEC authority. Such guidance could draw upon the extensive value of our and agriculture's current and widespread efforts in this arena. Such guidance would best be developed through meaningful collaboration with the egg and other agricultural sectors, its many enthusiastic stakeholders involved in our GHG reporting, as well as the US Department of Agriculture (USDA).

We oppose the proposed mandatory disclosure requirements because we believe it will have a perverse, negative effect on greenhouse gas (GHG) emissions reporting in our sector, and because of multiple practical, policy and liability considerations that are in many ways unique to egg farmers and other agricultural sectors. UEP's farmer members are consistently engaged in increasingly widespread and innovative efforts to supply "consistent, comparable and reliable" greenhouse gas (GHG) emissions information to their customers. Comparable efforts are underway in most other agricultural sectors. By mandating such disclosures by registered entities and the liabilities created for farmers around Scope 3 reporting, the Commission's rule would stifle this innovation.

Egg farmers are rightfully proud of the tremendous strides they took since 1960 to meet the substantial growth in the US's demand for shell eggs and egg products while having very small GHG emissions per unit of protein, per egg, and most notably with **lower total** GHG emissions. Between 1960 and 2010, US egg farmers **increased** total egg production by 30% (from 59.8 billion eggs to 77.8 billion per year) while **reducing** the entire sector's GHG lifecycle footprint by over 60% (from 26.2 million metric tons of CO2 equivalents to 9.8 million metric tons).[1] The US egg sector is now updating that 50-year LCA and recently completed a detailed survey of egg producers' operations. These data are being compiled, aggregated and utilized to generate an updated LCA to be completed later this year. That updated LCA will provide the best, most authoritative, contemporaneous estimates of GHG emissions associated with egg production and for the reasons discussed below, these will be far more accurate than those based solely on operational data from individual operations.

UEP's Specific Concerns with the Proposed Mandatory Disclosure Requirements

I. Imposing Scope 3 emissions reporting requirements, and the associated liabilities that come with that requirement, erroneously presumes that the science, institutions and mechanisms to support estimating a supply chain's Scope 3 emissions are largely well established and settled business practices. This is an incorrect presumption.

UEP, other major egg-industry organizations and a growing number of egg farmers representing a large proportion of the eggs produced in the US are active participants in the multi-stakeholder US Roundtable for Sustainable Poultry and Eggs (US-RSPE). The stakeholders include major food companies that sell eggs, and products with eggs in them, to other customers or the final consumer. Central to the US-RSPE's work is its Sustainability Framework that egg producers will use to voluntarily explore how they best measure and report the sustainability performance of their operations. There have been two iterations of the Framework that have served as pilots for testing and its refinement. The next - version is expected for release later in 2022 and it will remain in a continuous improvement process. The egg sector working with its food company stakeholders will for several years be experimenting with the Framework's best practices, how it should be supported by and integrated with the sector's LCAs, and otherwise working with
climate smart science community to get it correct.

Consider also the responses to USDA's recent notice of funding availability for climate smart agricultural supply chain projects. This May and June USDA received over 600 supply chain proposals that would operate in all fifty states and seeking approximately $18 billion in grants. We are aware of the content of several of these proposals and it is abundantly clear to us that throughout most of agriculture there remains a great deal of variation among them as to how Scope 3 emissions are accounted for, as well as exploration and experimentation as to best practices to help a supply chain account for Scope 3 emissions.

[1] See appended Xin et al., A Comparative Assessment of the Environmental Footprint of the U.S. Egg Industry in 1960 and 2010, Report submitted to the American Egg Board, 2013, page 2, and for further documentation see Nathan Pelletier et al, Comparison of the environmental footprint of the egg industry in the United States in 1960 and 2010, Journal of Poultry Science, 2014.

Our other specific practical, operational and liability concerns with the mandatory disclosure requirement and the reason we oppose it in favor of industry guidance are discussed below.

II. <u>Scope 1, 2 and 3 GHG emissions associated with agricultural commodity production are more accurately and efficiently estimated from the operational values calculated from commodity sector LCAs</u>.

The vast majority of farmers or ranchers are engaged in commodity production, where the commodity is or functions as a raw material for further processing and sale to customers in the supply chain. One unit (a dozen eggs, bushel of grain, gallon of milk, pound, ton, etc.) of a commodity produced in a sector not only is indistinguishable from all the other units produced in that sector, that uniformity is the very point and reason the sector is able to succeed; uniformity in qualities is what makes possible the sufficiently efficient processing of the commodity in the bulk amounts needed to economically create further value-added products with the uniform qualities demanded by customers. The farm or ranch level commodity-producing activities in this system are actively managed to create that uniformity, and producers are commonly penalized (i.e., reductions in prices paid) if certain key qualities are not present at the needed level.

Not surprisingly but not immediately obvious, the supply chain supporting commodity production is itself supported by commodity-producing sectors. The grains and oilseeds that serve as feed for hens are themselves commodities. Furthermore, the types of equipment, technologies and agricultural practices used in raw agricultural commodity production are themselves commodities, or very commodity-like. The successful farmer or rancher tends to be the producer most efficiently and effectively using that standardized equipment, technologies and agricultural practices – those that do not, other things being equal, have higher costs, lower returns and are susceptible to being driven from the sector. Agricultural sector LCAs have conclusively shown that the dominant factors driving the vast majority of the GHG emissions associated with the production of the sector's agricultural commodity involve the commodity production in its own supply chain and these very commodity-like equipment complements and the associated commodity-like technologies and practices. The GHG footprint of an individual farming and ranching operation is therefore highly comparable per unit of raw commodity across all operations in that sector.

There are numerous practical implications of this aspect of commodity production, but two of them are particularly important to estimating the GHG emissions in the production of an agricultural commodity.

1. The GHG footprint of an individual agricultural commodity operation is most accurately and reliably estimated from sector-wide averages derived from sector LCAs. Such sector studies, conducted through rigorous statistical sampling, using quality surveys and subject to quality assurances and controls, will reliably yield better estimates of GHG emissions for a region than would come from aggregating the results of 100's or 1000's of individual farmers' LCA's to supply their customers with Scope 3 information.

2. While commodity-sector wide averages accurately capture the lion's share of an individual operation's emissions, at the margin operations' management may generate some appreciable GHG emissions reductions beyond the sector averages.

Sector-wide LCA's therefore must be the foundation of good and sound GHG emissions estimates under climate smart agricultural policies. Put more plainly, the economics of an agricultural commodity sector require that a commodity form of GHG emissions accounting be used.

It is possible to do this while also recognizing producers who through aggressive or innovative management may attain or point the way towards above average GHG reductions in certain aspects of their commodity operations. The goal of the policy must be to economically and efficiently capture accurate GHG emissions estimates while promoting farm level problem solving and innovation that pushes the envelope of improved GHG performance for the entire sector. As we noted above, though, the science, institutions and mechanisms to help us account for emissions due to ongoing farm level activities remains a work in progress, even while produces may very well be rewarded in the marketplace as we work to refine and improve those institutions and mechanisms. This is one of the reasons we strongly encourage the Commission not to insert itself through mandatory disclosure requirements and the associated liabilities into this ongoing, market-oriented experimentation and development of best business practices.

III. <u>The collection of grains, oilseeds and other products in elevators or processing facilities completely obscures the possible sources of a feed buyer's Scope 3 emissions associated with that feed.</u>

For most of animal agriculture, including the egg farming sector, the grains and oilseeds fed to the animals are produced on multiple farming operations whose identity is unknown to the feed buyer. This is because the feed buyer purchases bulk commodities produced by dozens, hundreds or even thousands of grain and oilseed producers, where those commodities leave the farm and are comingled and aggregated at grain elevators (or feed mills and sometimes further processed) before sale. There can be multiple nodes in that supply chain before an egg producer buys the laying hens' feed. The identity of the farm and the feed grains they produced is not preserved through this supply chain. This generally makes it impossible to associate any unit of raw commodity with any individual farm or any individual farm field. In these instances, there is simply no mechanism today that would allow a food company to estimate their agricultural Scope 3 emissions taking place on the specific individual farm that actually produced the purchased raw agricultural commodities – sector estimates are the only practical solution.

For example, in the case of egg farming, grain sector life cycle analyses must be used by the egg sector to estimate the Scope 3 emissions involving feed grain production. The actual emissions at the actual locations where the grain was produced are estimated from the feed grain LCAs.

Occasionally, food companies purchasing agricultural commodities utilize contracts with specific growers. Some egg farmers purchase their grain from specific grain farmers and do so under contract. In these more limited instances, companies could, at least in theory, work with their contract producers to secure Scope 3 estimates that are specific to the contract growers' operations. In practice, given the strength and accuracy of sector-wide averages based on sector LCAs as discussed above, use of industry averages to estimate a contract grower's emissions will almost always be the best solution. But contract production for raw agricultural commodities is very rare. The vast majority of raw agricultural commodities, particularly in the grains sector, are shipped from the farm to aggregation and processing facilities.

IV. <u>Mandatory disclosure by registered entities of their Scope 3 emissions creates legal liabilities for them and will lead to agricultural suppliers being forced to adopt costly accounting measures to report emissions and certify their accuracy, creating additional legal liabilities for them as well as a cascade of other negative effects, and undermine industries' efforts to use the more accurate and efficient sector LCAs to estimate and report emissions.</u>

The proposed rule requires registered entities to report their Scope 3 emissions if those emissions are material to the performance of the company or if the reporting company has otherwise set Scope 3 targets. We understand that the Commission's and industry's accepted definition of materiality in this instance to be whenever Scope 3 emissions represent 40 percent or more of a registered company's emissions. Our experience tells us that there is not a food company in the US whose Scope 3 emissions derived from the agricultural goods bought in the US do not represent more than 40 percent of the company's total. Additionally, most large food retailers have set or expressed a desire to set Scope 3 targets. As a result, the Commission will, with modest preparation and even less agricultural expertise, be inserting itself overnight directly into the business operations of the vast majority of US farms and ranches, including our members' layer operations. We think this approach is rash and fundamentally unwise.

One of our greatest concerns about the proposal stems from the serious and great liabilities created in the system by mandating Scope 3 emissions disclosures. We recognize that the Commission has proposed a targeted safe harbor for Scope 3 emissions data provided to a registrant, and that the safe harbor is intended to ensure that the "disclosure of Scope 3 emissions by or on behalf of the registrant would be deemed not to be a fraudulent statement unless it is shown that such statement was made or reaffirmed without a reasonable basis or was disclosed other than in good faith." (Federal Register, Vol. 87, No. 69 at 21391).

Unfortunately, we expect this safe harbor would be of little actual value in practice. Despite its presence, a host of new and costly measures will be imposed on farmers by their customers to protect registered entities legal desperate to protect the value of their stock and avoid legal jeopardy.

In our experience, the mettle, strength and degree of protection afforded by a safe harbor or similar measure is most commonly stress-tested in the furnace of our legal system. We believe, and our experience is that most registered food companies will believe, that this safe harbor will be challenged in court by climate activists and shareholders supported through the work of the class action community. These legal fights will be exceedingly expensive, and in our instance an egg farmer and its supplied Scope 3 emissions data, or the data provided by an LCA sponsored by UEP or others, will be brought into that suit. The egg farmer and entities like UEP will not be able to avoid these liabilities and costs either.

That is just the start of the costs to farmers, though. In order to shield themselves from this actual liability as much as possible, a registered food company will seek to shift responsibility (and associated liability) for the Scope 3 information to the farmer or the entity supplying it to the farmer. They will do this by mandating that their suppliers adopt specific reporting protocols and practices, and that on-farm measures intended to reduce emissions be certified through third party audits conducted by the auditor of the registered company's choosing, and that the cost of all this be borne by the farming operator. Despite these costly measures, egg producers will remain embroiled in any legal challenges to these data.

This is not a hypothetical scenario. It has happened already (at great expense) in the egg sector involving other customer-demanded operational changes on egg farms, and our members are understandably concerned about further increasing the likelihood of more challenges, particularly given their razor thin margins. These rules will only cause the costs of farming to go up even more dramatically, resulting in consolidation and increases in average farm size in order to generate the gross revenues needed to support the staff and procedures necessary to meet their registered customer's demands. This is the exact opposite outcome that the President Biden's Administration is attempting to accomplish for farmers through numerous other efforts in the agricultural sector.[2]

Whether it takes a year or several for this scenario to emerge, it will emerge as a result of the Commission's proposal if finalized. For this reason also, we strongly urge the Commission not to adopt these mandatory reporting requirements.

We and our egg farmer members take no comfort from the extreme irony that many of the strongest proponents of the Commission's proposed rule also strongly condemn the concentration of production in agriculture and the growth in the average size of farming operations in general, and egg farms in particular. We do wish, though, that they acknowledged and treated seriously the consequences for the structure of agriculture in light of the manner in which they want GHG emissions to be reported by registered companies.

Additionally, this rule and its adverse impacts on agriculture are proposed at a time when farmers are already facing historic challenges and dramatically higher input prices, record-setting food and energy inflation is rampant, and the U.S. Department of Agriculture estimates "that the demand for food will rise by 70 to 100 percent by 2050."[3] Furthermore, food insecurity is very real today, both domestically and internationally.[4] This rule's requirements will hinder US agriculture's ability to address and weather these challenges.

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[2] https://www.whitehouse.gov/briefing-room/statements-releases/2022/01/03/fact-sheet-the-biden-harris-action-plan-for-a-fairer-more-competitive-and-more-resilient-meat-and-poultry-supply-chain/; https://www.whitehouse.gov/briefing-room/statements-releases/2022/05/11/fact-sheet-president-biden-announces-new-actions-to-address-putins-price-hike-make-food-more-affordable-and-lower-costs-for-farmers/; https://www.usda.gov/media/press-releases/2022/05/26/biden-harris-administration-announces-new-actions-strengthen-food

[3] https://www.usda.gov/topics/food-and-nutrition/food-security

[4] "But with COVID shutdowns, supply chain disruptions, and the war in Ukraine, we now have deep concern about how we will get the food and fertilizer flowing efficiently, ensuring that our producers and those around the world have the necessary tools and continue to keep food on the dinner tables of people across the world. This is a monumental challenge." USDA Secretary Vilsack, in remarks during the "Virtual Roundtable Discussion on "Food Security Issues Arising from Russia's Invasion of Ukraine", June 6, 2022.

Fortunately, there is a better alternative to the Commission's proposed mandatory disclosures. The Commission should develop thoughtful principles-based GHG guidance for use by industries as they report on material risks to their companies, and not mandate such disclosures. The marketplace is already aggressively driving the active exploration and experimentation on how to best report on Scope 1, 2 and 3 GHG emissions. We are truly concerned that the Commission's proposal will unnecessarily impede this work, with a host of truly bad outcomes for our egg farmer members in the process. UEP would welcome the Commission's meaningful participation as a stakeholder in the egg sectors' sustainability and GHG reporting efforts, and collaborating with us, our many other stakeholders and our customers to ensure quality information is being made to the public and investors alike.

Sincerely

Chad Gregory

Chad Gregory
President/CEO

INVITED REVIEW

Comparison of the environmental footprint of the egg industry in the United States in 1960 and 2010[1]

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ABSTRACT The US egg industry has evolved considerably over recent decades by incorporating new technologies and production practices. To date, there has been no comprehensive assessment of the resource demand and environmental effects of these changes. This study quantifies the environmental footprint of egg production supply chains in the United States for 2010 compared with 1960 using life cycle assessment. The analysis considers changes in both foreground (e.g., hen production performance) and background (e.g., efficiencies of energy provision, fertilizer production, production of feed inputs, and transport modes) system variables. The results revealed that feed efficiency, feed composition, and manure management are the 3 primary factors that determine the environmental impacts of US egg production. Further research and improvements in these areas will aid in continual reduction of the environmental footprint of the US egg industry over time. Per kilogram of eggs produced, the environmental footprint for 2010 is 65% lower in acidifying emissions, 71% lower in eutrophying emissions, 71% lower in greenhouse gas emissions, and 31% lower in cumulative energy demand compared with 1960. Table egg production was 30% higher in 2010; however, the total environmental footprint was 54% lower in acidifying emissions, 63% lower in eutrophying emissions, 63% lower in greenhouse gas emissions, and 13% lower in cumulative energy demand compared with 1960. Reductions in the environmental footprint over the 50-yr interval considered can be attributed to the following: 27 to 30% due to improved efficiencies of background systems, which outweighed the declining energy return on energy invested for primary energy sources; 30 to 44% due to changes in feed composition; and 28 to 43% due to improved bird performance.

Key words: egg, pullet, life cycle assessment, environmental footprint, energy return on energy invested

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INTRODUCTION

Agricultural production in the United States has advanced considerably over recent decades by incorporating new technologies to make more efficient use of finite resources such as land, water, and energy (Capper et al., 2009; Capper, 2011; Boyd and Cady, 2012; Hamilton et al., 2013). Egg production has followed a similar trend, achieving productivity levels that would have been difficult to imagine half a century ago. To date, there has been no comprehensive assessment of the resource demand and environmental effects of these changes in production practices and efficiencies.

Life cycle assessment (**LCA**) is an analytical framework for characterizing material and energy flows and emissions along product supply chains and for quantifying how these contribute to a variety of resource use, human health, and environmental impact potentials. The methodology has been standardized by the International Organization for Standardization (**ISO**) in the ISO 14040–14044 standard series (ISO, 2006). The key strength of LCA is that it facilitates identification of opportunities for mitigating key drivers of impacts while being sensitive to burden-shifting, whether between supply chain stages or between different kinds of environmental impacts (for example, greenhouse gas emissions versus ozone-depleting emissions).

In this study, we applied ISO 14044-compliant LCA methods (ISO, 2006) to quantify the changes in the environmental footprint of US egg production between 1960 and 2010. The specific objectives of the study were to

1) develop models of US egg production supply chains for 1960 and 2010 with regard to both

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foreground system variables (such as feed conversion or efficiency, bird body weight, bird mortality rate, hen-day egg production) and background system variables (such as efficiencies of energy provision, fertilizer production, production of feed inputs, transport modes);

2) characterize supply chain environmental performance of the US egg industry for 1960 and 2010 in terms of the following:
   - cumulative energy demand (**CED**, expressed in MJ)—all embodied renewable and nonrenewable energy inputs,
   - acidifying emissions (expressed in SO_2-equivalent units)—all emissions that contribute to ecosystem acidification,
   - eutrophying emissions (expressed in PO_4-equivalent units)—all emissions of N- and P-containing compounds that contribute to eutrophication of fresh water bodies, and
   - greenhouse gas (**GHG**) emissions (expressed in CO_2-equivalent units)—all emissions that contribute to increased atmospheric radiative forcing;
3) determine the magnitude of changes in production performance and environmental impacts associated with technological and management advancements over the 50-yr interval.

The results of the study are intended to provide the US egg industry and other stakeholders with science-based information concerning the impact of advances in egg production on resource utilization efficiencies and environmental performance. The study also offers insight into areas for further mitigation of environmental impacts and conservation of natural resources.

MATERIALS AND METHODS

Goal and Scope

The system boundaries for this analysis included all direct and indirect inputs and emissions arising from the production of raw materials for feed inputs, feed input processing, feed milling, hatcheries, and farm-level material and energy use at pullet and layer facilities for both 1960 and 2010 (Figure 1). The production and maintenance of infrastructure such as machinery and buildings were not included because, in high production-volume contexts, their contributions are typically trivial (Ayer and Tyedmers, 2009). These parallel models were subsequently used to evaluate the environmental footprint of US egg production in terms of CED, GHG, and acidifying and eutrophying emissions for 1960 versus 2010.

Life Cycle Inventory: 2010 Model

Foreground system data refer to information unique to the product system of interest. Foreground system data for feed milling, pullet, and layer facilities were collected via anonymous surveys from participating companies. The data collected represented 57.1 million pullets and 92.5 million laying hens, accounting for 26% of pullet stock and 33% of laying-hen stock in the United States in 2010. In the absence of company-specific information for hatcheries (no participants in the study), data for hatcheries were adopted from an earlier study of US broiler production systems (Pelletier, 2008).

Background system data refer to information regarding processes linked to the foreground system in the supply chain of interest, but shared with other supply chains. In the context of our analysis, this included the provision of energy carriers (i.e., energy sources such as fossil fuels and electricity), inputs to crop production and other feed input production and processing systems, and transportation modes. Background system data for the production and processing of feed ingredients were adapted from recent LCA studies of beef and pork production supply chains in the Upper Midwestern United States (Pelletier et al., 2010a,b) and global salmon aquaculture supply chains (Pelletier et al., 2009). These studies used identical modeling parameters to those of the current analysis and hence the feed input models could be directly adopted. Other background system data, including the provision of energy carriers, fertilizers, pesticides, and transportation models, were derived from the EcoInvent (2010) database and modified to reflect US energy inputs.

Modeling N and P Emissions. Nitrogen and P emissions from pullet and layer facilities were calculated using a nutrient balance model based on feed composition and assuming that hen body mass contains 2.2% N and 0.6% P, and eggs contain 1.7% N and 0.21% P as reported by Koelsch (2007). Nitrogen excretion estimates were used to calculate direct nitrous oxide, ammonia, and nitric oxide emissions from manure management and indirect nitrous oxide emissions from nitrate leaching and ammonia emissions following IPCC (2006) protocols and relevant Tier I and Tier II emission factors at time of deposition, storage, and application. Methane emissions from manure management were calculated following IPCC (2006) Tier I protocols. Phosphorus emissions were calculated at a 2.9% leaching rate at time of application of manure to agricultural lands following Dalgaard et al. (2008).

Coproduct Allocation. Coproduct allocation is required to apportion resource use and emissions between the products of multi-output systems. The mass-adjusted gross chemical energy content of coproducts was used 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 that speaks directly to the ecological efficiency with which the system provides available food energy (whether for direct human consumption or for livestock

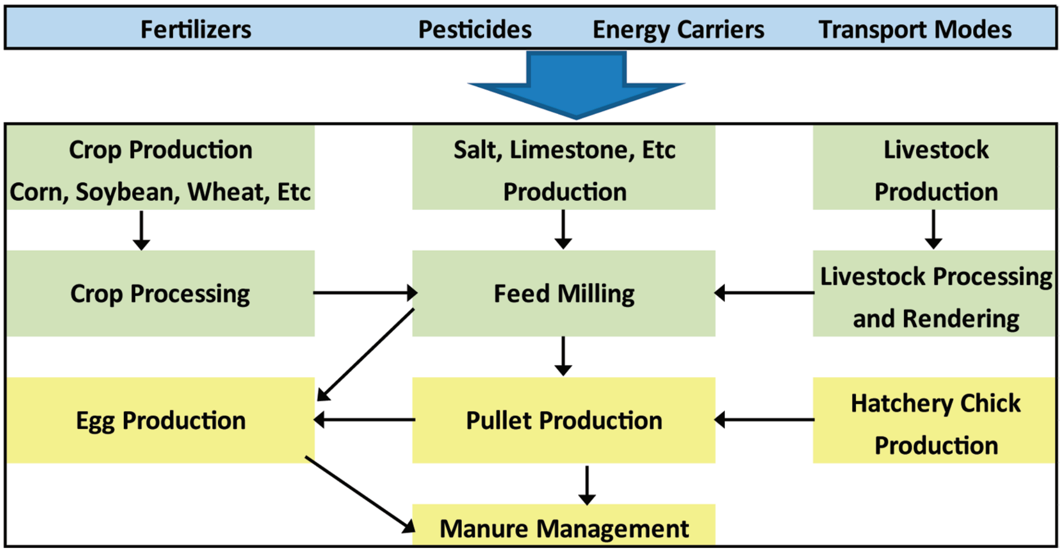

Figure 1. System boundaries for a life cycle assessment of egg production in the United States for 1960 and 2010 (background processes such as fertilizers, pesticides, and transport modes were derived from the EcoInvent (2010) database but were modified to reflect US energy carriers). Color version available in the online PDF.

feed). This allocation strategy is consistent with the ISO 14044 specification that the "inventory is based on material balances between inputs and outputs. Allocation procedures should therefore approximate as much as possible such fundamental input/output relationships and characteristics" (ISO, 2006). A detailed discussion of this allocation rationale was given by Pelletier and Tyedmers (2011). This approach was chosen over economic allocation, which is sometimes used in reported food system LCA, because (1) economic allocation is a last-resort option in the ISO 14044 hierarchy (ISO, 2006) and (2) the use of economic allocation typically produces results that poorly reflect the physical reality of the systems modeled (Pelletier and Tyedmers, 2011). The use of substitution (following a consequential data modeling approach) was similarly deemed inappropriate for our analysis, which was intended to establish baseline models rather than to model market-level consequences of possible changes in production systems.

Life Cycle Inventory: 1960 Model

In developing a model to represent average US egg supply chain characteristics in 1960, industry and academic expert sources were consulted and published literature referenced. This required estimating performance efficiencies for both foreground and background system variables. Where we were not able to identify a robust basis for characterizing specific foreground system variables for 1960 (e.g., energy use in poultry housing systems), we used 2010 data in proxy, but with modifications to accommodate our 1960s background system variables. This likely resulted in an underestimation of differences in the environmental performance of egg production in 1960 versus 2010. Modeling of key variables for 1960 is described below.

1960 Energy Carrier Models. Energy return on energy invested (**EROI**) is a measure of the energy efficiency of energy production. It indicates the amount of energy yield for a given energy carrier (e.g., oil, gas, coal, or electricity) relative to the energy input to its procurement. Several researchers have reported declining EROI values for different energy carriers over time (Gangon et al., 2009; Guilford et al., 2011; Lambert et al., 2012). This is because, as easily accessible, high-quality energy resources are exhausted, an increasing proportion of energy production derives from less-accessible, marginal energy resources that are more energy-intensive to exploit. In short, over time, more energy is required to produce an equivalent unit of energy. From a life cycle perspective, taking into account this changing efficiency and the associated changes in environmental burdens is essential to realistic, time-sensitive modeling.

The EROI values at any given time differ between energy carriers, region of production, and production technology. Moreover, EROI can be described from both production and consumption perspectives. Because energy commodities are widely traded, calculating EROI values for energy carriers used in a given country requires attention to trade patterns and, in the case of electricity, country-specific energy mixes.

For the purpose of the present analysis, EROI values for the United States as well as global EROI values for the production of specific energy carriers were adopted from or calculated based on the work of Gangon et al. (2009), Guilford et al. (2011), and Lambert et al. (2012). In turn, these were used to calculate EROI for primary energy carriers used in the United States in 1960 and 2010 using US Energy Information Administration (**USEIA**; 2012) statistics for US consumption and imports of energy products. The USEIA (2012) statistics for the energy mixes used in US electricity

Table 1. Estimated energy return on energy invested values for energy carriers used in 1960 and 2010 in the United States

Energy carrier	1960	2010	Scaling factor between 2010 and 1960
Coal	75	60	0.8
Oil/gas	47	15	0.3
Nuclear and renewables	15	15	1.0
Electricity	14	14	1.0

production were also employed to calculate 1960 and 2010 EROI values for electricity. On this basis, scaling factors were derived to represent the comparative EROI of energy carriers between 1960 and 2010 (Table 1). These factors were applied to modify the life cycle inventories used for 2010 energy carriers (adapted from the EcoInvent database) to arrive at 1960 energy carrier life cycle inventories that approximate changes in the environmental performance profile of energy carriers used in the United States over this interval. Potential differences in distribution losses for electricity (grid efficiencies) in 1960 compared with 2010 were not considered.

1960 Fertilizer Production Models. The US fertilizer mixes for 1960 were derived from International Fertilizer Industry Association (**IFA**) statistics (IFA, 2012). Ammonia production accounts for 87% of the fertilizer industry's energy consumption (IFA, 2009). Based on data regarding improvements in the efficiency of ammonia plants over time, IFA (2009) shows that efficiencies improved from 58 to 28 MJ of energy required per metric ton of ammonia produced between 1960 and 2010. Effectively, this means that producing ammonia in 1960 required 2.07 times as much direct energy input as in 2010. This ratio was hence applied to scale the energy inputs for average ammonia production for the EcoInvent (2010) life cycle inventory used to represent contemporary ammonia production to arrive at a representative 1960 life cycle inventory.

For all other fertilizer "building blocks," Kongshaug (1998) provides estimates of net energy consumption for "old technology–1970," "average technology–1998," and "best available technology–1998." These estimates largely distinguish between net energy production in the form of steam, which may or may not be productively used. The modified EcoInvent processes for fertilizer production (originally representing average EU production, but modified to reflect US energy inputs) used in the present analysis assume that net energy produced is lost as waste heat. For the purpose of the current analysis, this assumption was similarly adopted; namely, we did not distinguish between sulfuric acid, nitric acid, and phosphoric acid net energy production in 1960 versus 2010 (although we did apply the modified energy carrier inventories in the 1960 fertilizer production models).

1960 Freight Transport Models. United States Department of Energy (**US-DOE**) data were used to calculate differences in the energy efficiency of freight transport by mode in 1960 compared with 2010 (US-DOE, 2012). The energy intensity of US heavy truck freight decreased from 24,960 BTU (26.3 MJ) per vehicle mile in 1970 to 21,463 BTU (22.6 MJ) per vehicle mile in 2010, with an average annual decrease of 0.4%. Making a linear extrapolation to 1960 on this basis, estimated energy intensity of road freight was 25,977 BTU (27.4 MJ) per vehicle mile. A correction factor of 1.21 was therefore applied to the EcoInvent (2010) model used to represent US road freight energy use in 2010 for the 1960 model.

The energy intensity of US rail freight decreased from 691 BTU (0.73 MJ) per ton-mile in 1970 to 289 BTU (0.30 MJ) per ton-mile in 2010, with an average annual decrease of 2.2% (US-DOE, 2012). Making a linear extrapolation to 1960 on this basis, estimated energy intensity of US rail freight was 859 BTU (0.91 MJ) per ton-mile. A correction factor of 2.97 was therefore applied to the EcoInvent (2010) model used to represent US rail freight energy use in 2010 for the 1960 model.

The US-DOE (2012) only provides data for changes in the energy intensity of water freight on taxable waterways from 1997 (266 BTU or 0.28 MJ per ton-mile) to 2010 (217 BTU or 0.23 MJ per ton-mile), with an average annual decrease of 2.20%. Extrapolating back to 1960 suggests an energy intensity of 595 BTU (0.63 MJ) per ton-mile in 1960, which would imply a correction factor of 2.74. This is very similar to the estimated correction factor for rail freight extrapolating from 1970 to 2010 time series data. This estimate was the weakest, however, given that efficiency in 1960 was extrapolated from only 14 yr of data spanning 1997 to 2010.

For comparison, using data from Fearnley's Review (2012) for world seaborne trade from 1969 to 2010 and estimates of marine fuel use from 1950 to 2010 (Eyring et al., 2005), the estimated correction factor for global ocean freight was 1.33. Elsewhere, a study by Lloyd's Register (2008) suggested a 75% improvement in fuel efficiency for shipping between 1976 and 2007. However, for consistency with our calculations for road and rail freight, we adopted the correction factor of 2.74.

1960 Feed Input Models. Smil et al. (1983) reported energy inputs to US corn production for 1959. On this basis, direct energy inputs had declined 61% per unit production compared with reported energy inputs to corn production in 2001 (adopted for 2010) as estimated by the US National Agricultural Statistics Service (NASS, 2004). No similar estimates were available for our 1960s models for soy or wheat; hence, a proportionate decline was assumed in energy inputs relative to

NASS (2004) energy use estimates for soybeans in 2002 and wheat in 1998. Pesticide use for crops was based on statistics for 1964 provided by the USDA (1995). Fertilizer use was also based on statistics for 1964 provided by USDA (2012). Sulfur and lime inputs were assumed to be similar between 1960 and 2010. Crop yield data for 1960 were taken from the USDA Feed Grains Database and USDA Oil Seeds Database.

All animal-derived and other feed inputs were based on the LCA models reported by Pelletier et al. (2009; for fish meal) and Pelletier et al. (2010a,b; for porcine and ruminant materials), which were created using identical modeling protocols to those used for the 2010 model in the current analysis. For ruminant production, the model of Pelletier et al. was used for grass-fed beef production to represent 1960s conditions (versus their model of conventional, feedlot production to represent 2010 conditions). For pig production, the model of Pelletier et al. was used for low-performance niche production to approximate 1960s conditions (versus their model of conventional, commodity production to represent 2010 conditions). In the absence of an alternative model for broiler chicken production (most common source of processing coproducts rendered into poultry by-product meal and fat), it was assumed that the spent hens destined for rendering as modeled in the current analysis were used for the production of poultry by-product meal and fat.

1960 Pullet and Layer Production Models. Bird performance data for pullet and layer production were taken from Winter and Funk (1960), and verified with industry and academia experts. For pullets, this included feed composition, feed consumed per pullet sold, mortality rate (% of initial placement), and age and BW of pullets at the time of moving into the layer houses. For layers, this included feed composition, daily feed consumption, annual egg production per hen, egg weight, feed conversion, mortality rate, and number of pullets added to layer houses per year.

Life Cycle Impact Assessment and Interpretation

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. In this study, CED and GHG acidifying and eutrophying emissions were quantified. Cumulative energy demand (MJ) accounts for conversion efficiencies and the quality of energy inputs (Frischknecht et al., 2007). Quantification of GHG emissions (CO_2-equivalency over a 100-yr time horizon according to IPCC, 2006), acidifying emissions (SO_2-equivalency), and eutrophying emissions (PO_4-equivalency) followed the CML 2 Baseline 2000 method (Guinee et al., 2001).

The environmental impacts were first assessed for each supply chain node considered, then for supply chains in aggregate. Results for the 1960 and 2010 models were subsequently compared to determine differences in environmental performance over time. More detailed contribution analyses were conducted to determine the extent to which the observed differences in environmental performance between 1960 and 2010 were attributable to various factors or model assumptions. The first such analysis evaluated the influence of differences in background system variables only between 1960 and 2010 (i.e., production efficiencies for energy carriers, fertilizers, transport modes, and feed inputs). Here, all 1960 submodels were replaced with 2010 submodels for these parameters. The second analysis used the same feed composition as 2010 in the 1960 model, and also replaced all 1960s background system submodels with 2010 submodels to determine the differences strictly attributable to changes in either feed composition or animal husbandry practices and performance over time.

RESULTS AND DISCUSSION

Life Cycle Inventory Results

The life cycle inventory data used for the 2010 and 1960 models of US egg production supply chains are presented in Tables 2 to 9. Inventory data for production and processing of individual feed ingredients (other than corn, wheat, and soy) are not provided herein but can be found in Pelletier et al. (2009, 2010a,b).

Substantial increases in crop yield over the 50 yr, in many cases, offset the increases in resource inputs, with some inputs higher per unit yield in 1960 or in 2010, depending on the input and crop (Table 2). For feed milling in 2010, the reported proportions and total amounts of different energy carrier inputs per metric ton of feed milled were highly variable (Table 3), as were the distances traveled for the feed inputs sourced (Table 4). For the purpose of our analysis, total consumption-weighted averages were used to arrive at the proportions and feed transport distances modeled.

Reported data were similarly variable for pullet and layer facilities for parameters such as water use, energy use, manure mass, and so on. Again, although the ranges of values are reported in the proceeding tables, production-weighted averages were used to construct the life cycle inventory model.

Both the types and inclusion rate of ingredients in pullet and layer feeds changed between 1960 and 2010 (Tables 5 and 6). Whereas corn and soy products constituted the core bulk ingredients for both periods, wheat was a more important input in 1960 (10% wheat middlings in layer diets vs. 0.8% in 2010). Several ingredients were also used in only one period or the other, for example, green feed (modeled here as alfalfa) and fish meal in 1960 pullet feeds, and bakery material in 2010 pullet and layer feeds. Notable here is the reduced fraction of animal-derived materials (approximately 50% of 1960 levels) in contemporary feeds. The N and

Table 2. Life cycle inventory data per metric ton of corn, soy, and wheat produced in the United States in 1960 and 2010

	2010			1960		
Item	Corn	Soy	Wheat	Corn	Soy	Wheat
Input						
Fertilizer (kg)						
N	16.1	1.12	20.1	16.6	0.74	9.17
P_2O_5	5.55	5.53	6.91	10.8	2.72	7.03
K_2O	5.71	7.75	1.36	8.50	3.35	3.93
Sulfur	0.27	0.13	0.53	0.27	0.13	0.53
Lime	33.5	0.00	0.00	33.4	0.00	0.00
Energy						
Diesel (L)	4.49	10.9	13.2	4.47	17.5	21.3
Gas (L)	1.17	3.49	3.02	12.1	5.62	4.86
Liquid propane gas (L)	7.02	0.00	3.82	2.68	0.00	6.16
Electricity (kWh)	4.33	0.00	11.9	0.00	0.00	19.19
Total pesticides (kg)	0.25	0.46	0.29	0.20	0.21	0.12
Herbicides	0.24	0.45	0.12	0.13	0.09	0.11
Insecticides	0.01	0.01	0.00	0.08	0.11	0.01
Other (fungicides)	0.00	0.00	0.17	0.00	0.00	0.00
Seed (kg)	2.10	23.4	34.5	20.5	45.0	41.8
Output						
Nitrous oxide (kg)	0.46	0.25	0.55	0.49	0.27	0.36
Ammonia (kg)	2.38	2.19	4.13	3.57	3.91	4.46
Nitric oxide (kg)	0.35	0.02	0.43	0.36	0.02	0.20
Carbon dioxide (kg)	17.2	0.17	3.04	14.3	0.03	0.42
Nitrate (kg)	1.44	0.00	0.00	4.49	0.00	0.00
Phosphate (kg)	0.00	0.00	0.03	0.14	0.00	0.00
Yield (t)	1.00	1.00	1.00	1.00	1.00	1.00

P contents of different feed ingredients, as used to estimate nutrient balances, are listed in Table 7.

Perhaps most striking at the inventory level were the differences in resources consumed and other performance parameters for pullet (Table 8) and layer (Table 9) production in 1960 compared with 2010. Feed consumption per pullet raised decreased by 48% over the 50-yr interval, in part explained by a 30% lower BW at the onset of production and in part by a 70% lower mortality rate (Table 8). As a result of reduced mortality, the number of chicks required (per thousand pullets produced) also decreased by a net 8.6% (Table 8). At the same time, estimated losses of N and P decreased by 39 and 60%, respectively. Unfortunately, data for energy inputs to pullet facilities in 1960 could not be found; hence, they were assumed comparable to 2010.

For egg production, lower bird BW (2.04 kg/hen in 1960 vs. 1.54 kg/hen in 2010) was one of the main drivers for the observed 26% lower feed consumption per hen in 2010 (Table 9). Lower daily feed use, combined with a 27% higher hen-day egg production and a 57% lower mortality rate, resulted in 42% less feed consumed per kilogram of egg produced. The number of pullets sourced per metric ton of eggs produced also decreased by 22% (Table 9) due to lower mortality. Nitrogen and P emissions decreased by 47 and 64%, respectively.

Table 3. Energy inputs per metric ton (1,000 kg or 2,200 lb) of pullet/layer feed milled in reporting facilities in the United States in 2010 (representing a total production of 2,679,405 t of feed)[1]

Item	Production-weighted average	Range
Electricity (MJ)	15.8	1.8–52.9
Diesel (MJ)	51.1	0–122.8
Gasoline (MJ)	1.5	0–3.4
Natural gas (MJ)	0	0–0.02

[1]This data set was also used for the 1960 model.

Interpretation of Life Cycle Impact Assessment Results

Life Cycle Impact Assessment Results for Energy Carriers in 1960 Versus 2010. Energy return on energy invested was substantially higher in 1960 for all primary energy carriers other than coal. As a result, CED and emissions were correspondingly higher in 2010 (Figure 2). The smaller difference for coal in 1960 is explained by the low energy costs of extracting coal relative to the energy costs of transporting coal to markets. Because rail and water freight transport modes were considerably less energy efficient in 1960, these differences effectively offset differences in EROI for coal in 1960 compared with 2010. Eutrophying and GHG emissions for electricity production were also slightly higher in 1960 (Figure 2), largely due to 2 factors. The first factor is the higher fraction of (in particular) coal and other fossil fuels in the 1960 energy mix compared with a greater share of nuclear power generation in 2010. The second factor is the lower efficiency of transforming primary energy carriers into electricity in 1960.

Life Cycle Impact Assessment Results for Fertilizer Inputs in 1960 Versus 2010. Despite the substantial increases in the energy efficiency of ammonia production, declining EROI values for energy production effectively offset these gains. As a result, the compara-

Table 4. Distances traveled for inputs to pullet/layer feed milled in reporting facilities in the United States in 2010 (representing a total production of 2,679,405 t)[1]

Feed input	Distance to processor[2] (km)	Distance to feed mill[3] (km)	Range
Corn		27	24–48
Corn dried distillers grains with solubles	25	116	1–193
Soy meal	100	96	29–133
Bakery material	wheat: 100 to flour mill flour: 1,000 to bakery	258	97–587
Wheat middlings	100	474	241–604
Meat and bone meal	100	151	56–322
Fat	100	272	0–579
Salt	25	370	0–861
Limestone	100	142	0–241
Calcium	100	186	137–225
Phosphate	100	239	0–861
Trace vitamins	100	325	0–563

[1]This data set was also used for the 1960 model.

[2]Assumed average distances.

[3]Production-weighted average.

tive impacts of N fertilizers consumed in the United States in 2010 were very similar to those of 1960. Impacts for P fertilizer were also similar, with the exception of considerably higher eutrophication impacts in 1960, mostly due to the larger fraction of triple super phosphate in the 1960 fertilizer mix. In contrast, all impacts associated with the US potassium fertilizer mix were substantially higher in 1960 compared with 2010 due to the predominance of more energy-intensive K sources in 1960 versus greater reliance on less energy-intensive potassium chloride in 2010 (Figure 3).

Life Cycle Impact Assessment Results for Transport Modes in 1960 Versus 2010. Acidifying, eutrophying, and GHG emissions per metric ton-kilometer of freight transport were considerably higher in 1960 compared with 2010 for both rail and ocean freight. Interestingly, the declining EROI of fossils fuels over this interval offset almost exactly the improved fuel efficiencies enjoyed by contemporary rail and ocean freight, resulting in very similar CED. For road freight, in contrast, CED was much lower in 1960, and all other impacts very similar to those estimated for 2010. This outcome reflects the lower efficiency gains for road freight compared with rail and ocean freight for the 50-yr interval (Figure 4).

Life Cycle Impact Assessment Results for Feed Inputs in 1960 Versus 2010. In general, the production of raw materials was the largest contributor to impacts for feed inputs to pullet and layer systems, although processing-related emissions were notable for some inputs such as corn dried distillers grains with solubles. Milling-related impacts accounted for a very small fraction of emissions per metric ton of feed produced. Production of animal-derived feed inputs was

Table 5. Pullet feed composition for egg production in the United States in 1960 (based on Winter and Funk, 1960) and 2010 (based on the production-weighted average of feed composition data from reporting pullet producers)

Item	1960 (% inclusion)	2010 (% inclusion)	2010 (range)
Corn	78.1	60.0	41.0–70.7
Corn dried distillers grains with solubles	1.0	6.2	0–13.0
Soy meal	10.3	21.0	13.0–27.0
Dehydrated green feed[1]	3.0	0.0	N/A[2]
Fish meal	1.2	0.0	N/A
Bakery material	0.0	1.0	0–13.0
Wheat middlings	0.0	0.9	0–7.0
Meat and bone meal[3]	2.5	1.0	0–5.7
Fat[4]	0.3	0.9	0–1.7
Salt	0.5	0.3	0–0.4
Limestone	1.5	6.2	0–10.5
Dicalcium phosphate	0.6	0.0	N/A
Calcium	0.0	1.3	0–10.0
Phosphate	0.0	0.7	0–1.5
Other[5]	1.0	0.5	0–2.1

[1]Modeled as alfalfa hay based on Pelletier et al. (2010a).

[2]N/A = not applicable.

[3]63% ruminant, 26% porcine, 11% poultry (assumed same as 2010).

[4]50% poultry, 50% vegetable (assumed to be soy oil; assumed same as 2010).

[5]Includes trace vitamins and minerals, modeled as DL-methionine.

Table 6. Layer feed composition for egg production in the United States in 1960 (based on Winter and Funk, 1960) and in 2010 (based on feed composition data from reporting egg producers)

Item	1960 (% inclusion)	2010 (% inclusion)	2010 (range)
Corn	63.9	58.6	40.5–69.2
Corn dried distillers grains with solubles	0	6.1	0–15.1
Soy meal	12	19.3	10.0–25.7
Bakery material		0.9	0–12.4
Wheat middlings	10	0.8	0–9.9
Dehydrated green feed[1]	2.5	0	N/A[2]
Meat and bone meal[3]	5	1.8	0–7.8
Fat[4]	1	0.9	0–4.4
Salt	0.5	0.3	0–1.0
Limestone	3.7	6.8	0–11.6
Dicalcium phosphate	1.3	0	N/A
Calcium	0	2.1	0–9.8
Phosphate	0	0.5	0–1.0
Other[5]	0.1	0.5	0–1.8

[1]Modeled as alfalfa hay.

[2]N/A = not applicable.

[3]81% ruminant, 17% porcine, 2% poultry.

[4]4% ruminant, 2% porcine, 58.5% poultry, 35.5% vegetable (assumed to be soy oil).

[5]Includes trace vitamins and minerals, modeled as DL-methionine.

most impactful across the impact categories. This is unsurprising given the nature of feed conversion, which effectively acts as a multiplier for the impacts of producing the underpinning feed inputs, along with other inputs to animal husbandry, processing, and reduction of processing coproducts into meals and fats. This is particularly true for the production of meat and bone meal and fat from ruminant sources compared with porcine and poultry sources because feed inputs and associated emissions to produce ruminants are considerably higher.

Emission-related impacts for feed inputs produced in 1960 were almost universally higher than those in 2010. This reflected a combination of factors, including improved efficiencies of N fertilizer production, transport modes and, in particular, much-improved yields in 2010. The opposite was true for CED, however, where declining EROI effectively outweighed other efficiency gains (Table 10, Figure 5).

As a result of both the differences in impacts attributable to feed inputs in 1960 compared with 2010 and changes in feed formulation over time (particularly decreased use of animal-derived meals and fats), a similar pattern was observed for pullet and layer feeds. Averaged across emission-related impact categories, impacts for a given quantity of feed produced in 2010 were reduced by 49% relative to 1960 for pullet feeds and reduced by 63% for layer feeds. In contrast, CED was 36 and 2% higher, respectively (Table 11).

Comparing Pullet Production Between 1960 and 2010. Emissions-related impacts of pullet production in both 1960 and 2010 were largely driven by 2 factors—feed inputs and manure management (Figure 6). For CED, direct energy inputs to pullet houses rank second to feed inputs. However, the relative importance of these factors differed between 1960 and 2010. In 1960, feed inputs weighed most heavily across impact categories, particularly for GHG and CED. In 2010, manure management was the most important variable for acidifying and eutrophying emissions, due to decreased emissions associated with the production of feed inputs. The relative importance of direct energy inputs also increased in 2010, again due to the declining relevance of feed inputs as a result of changing feed composition (less animal-derived materials). Averaged across emissions-related impact categories, the environmental impact associated with pullet production was reduced by 56% in 2010 relative to 1960. Cumulative energy demand was also slightly reduced, at 9% (Table 12).

Comparing Egg Production Between 1960 and 2010. The distribution of impacts for egg production was very similar to that of pullet production for both 1960 and 2010. In 2010, manure management replaced feed inputs as the largest source of acidifying and eutrophying emissions (despite substantially lower losses of N and P per quantity of eggs produced), whereas feed remained the dominant (although smaller) contributor to both GHG emissions and CED. These changes re-

Table 7. Proximate composition of feed inputs used for calculating intake, excretion, and losses of N and P

Feed ingredient	% N	% P
Corn	1.224	0.260
Corn dried distillers grains with solubles	4.224	0.710
Soybean meal	6.899	0.620
Bakery by-product	1.728	0.250
Wheat middlings	2.706	0.910
Alfalfa hay (17% CP)	2.720	0.250
Meat and bone meal	8.000	4.000
Fish meal (66% CP)	10.56	3.150
Fat	0	0
Limestone	0	0.020
Phosphate	0	0.4364
Trace vitamins	0	0
Methionine	8.750	0

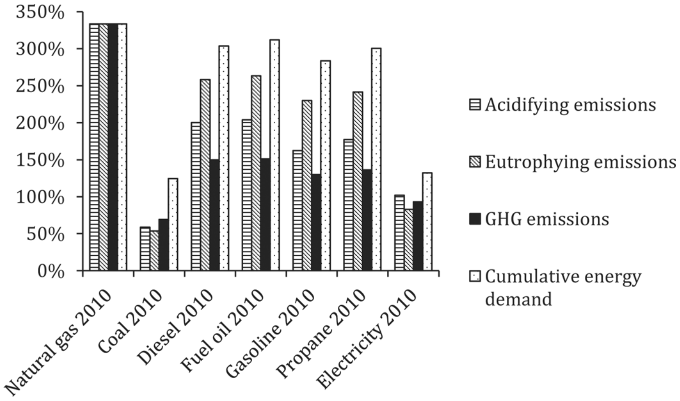

Figure 2. Life cycle impact assessment results for energy carriers used in the United States in 2010 compared with 1960 (all impacts for 2010 presented as a percentage of impacts in 1960). GHG = greenhouse gas.

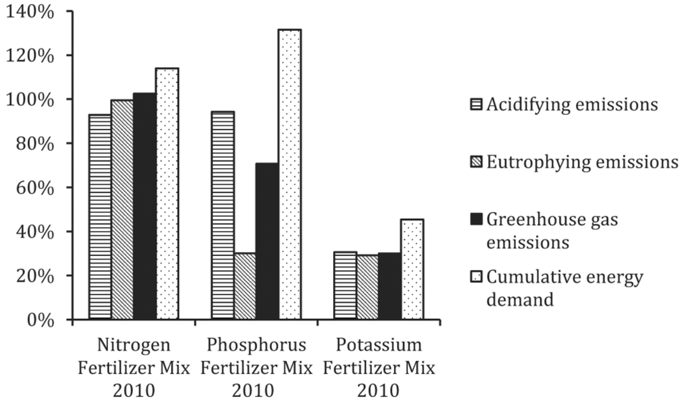

Figure 3. Life cycle impact assessment results for average US fertilizer mixes in 2010 compared with 1960 (all impacts for 2010 presented as a percentage of impacts in 1960).

flected both changing feed composition and improved feed use efficiency. Pullet production contributed approximately 10% to emissions-related impacts in both 1960 and 2010, and slightly more for CED (Figure 7). In general, direct energy inputs were of lesser importance. Overall, emissions-related impacts of egg production in 2010 were reduced by 69% relative to 1960, whereas CED was reduced by 31% (Table 12).

Analysis of Drivers of Observed Differences in Impacts Between 1960 and 2010. Applying 2010 background system submodels in the 1960 egg production model, we estimated that 27 to 30% of the observed differences in acidifying, eutrophying, and GHG emissions were attributable to changes in the efficiencies of background systems such as fertilizer and feed input production, and transport modes. These outweighed the declining EROI for primary energy carriers in these impact categories. For CED, however, applying 2010 energy carriers to the 1960 model resulted in 35% higher impacts in this category (Table 13).

Using both 2010 background system models and feed composition in the 1960 egg production model, we further estimated that changes in feed composition over time accounted for 30% of the observed decline in acidifying emissions for egg production in 2010, 35%

Table 8. Life cycle inventory data for the production of 1,000 pullets in the United States in 1960 (based on Winter and Funk, 1960) and in 2010 (based on the production-weighted average data from reporting pullet producers representing 57,116,182 pullets)

Item	1960 average	2010 average	2010 range	Percent change
Chicks	1,133	1,036	1,021–1,047	−9
Mass/chick (g)	39.8	39.8	39.1–40.0	0
Distance (km)	434	434	32.2–845	0
Feed (kg)	10.2	5.27	4.31–5.75	−48
Distance (km)	19.2	19.2	0–112	0
Water[1] (m^3)	17.9	9.22	7.54–10.1	−48
Energy[2] (MJ)				
Electricity	3,015	3,015	1,425–5,721	0
Diesel	105	105	0–1,084	0
Gasoline	95.8	95.8	0–517	0
Propane	1,654	1,654	0–4,747	0
Natural gas	187	187	0–1,932	0
Fuel oil	2.63	2.63	0–158	0
Output				
Pullets	1,000	1,000	1,000	0
Mass (t)	1.74	1.22	1.16–1.30	−30
Manure[3] (t)	6.46	3.38	0.59–4.59	−48
Distance[4] (km)	10.0	10.0		0
Estimated N loss (kg)	178	108	81.9–122	−39
Estimated P loss (kg)	32.9	13.3	9.09–15.7	−60
BW (kg/bird)	1.7	1.2	1.16–1.30	−30
Mortality rate (%)	11.7	3.5	2.1–4.7	−70

[1]Water use estimated as 1.75 × feed input.

[2]Year 1960 data assumed to be same as 2010.

[3]Manure mass on an as-removed basis, assuming proportionate to the ratio of feed use to manure production in 2010.

[4]Assumed distance of travel from farm to destination of manure application.

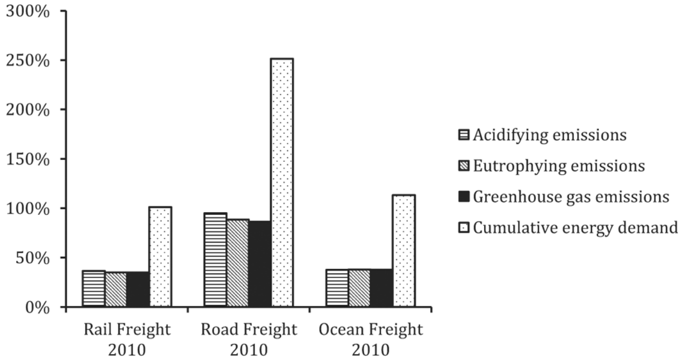

Figure 4. Life cycle impact assessment results per metric ton-kilometer for ocean, rail, and road freight in the United States in 2010 compared with 1960 (all impacts for 2010 presented as a percentage of impacts in 1960).

for eutrophying emissions, and 44% for GHG emissions. The remaining proportion of observed decline was attributable to improved bird performance over the 50-yr interval (e.g., better feed efficiency, lower mortality rate): 43% for acidifying emissions, 35% for eutrophying emissions, and 28% for GHG emissions. Despite declining EROI, CED in 2010 was only 30% that of 1960, due to a combination of changing feed composition and improved bird production practices (Table 13).

Comparison with Other Studies. A limited number of temporal analyses of the environmental impacts of animal production are available. Capper et al. (2009) and Capper (2011) evaluated changes in the environmental performance of beef production in 1977 versus 2007, and dairy production in 1944 versus 2007. Considerable gains in the efficiency of resource utilization (69.9% of animals, 81.4% of feedstuffs, 87.9% of the water, and 67.0% of the land required) per kilogram of beef produced in 2007 compared with 1977, and commensurate decreases (16.3%) in associated GHG emissions, were documented. Similar gains in resource efficiency were estimated for dairy (21% of animals, 23% of feedstuffs, 35% of the water, and only 10% of the land per kg of milk produced in 2007 compared with 1944), whereas GHG emissions were 37% of 1944 levels.

It should be noted that these studies (nor those discussed below) did not take into account changes in the resource efficiencies of background systems, hence are likely quite conservative. Our estimates of the scale of resource efficiencies and emission reductions for egg production between 1960 and 2010 are, nonetheless, of

Table 9. Life cycle inventory data per metric ton of eggs produced in the United States in 1960 (based on Winter and Funk, 1960) and in 2010 (based on the production-weighted average data from reporting egg producers representing 1,542,507.6 t of eggs)

Item	1960 average	2010 average	2010 range	Percent change
Pullets	46	36	21–50	−22
Distance (km)	52.9	52.9	1.61–452	0
Layer feed consumption				
kg/100 layers per d	12.23	9.03	8.1–11.3	−26
kg of feed/kg of eggs	3.44	1.98	1.76–2.32	−42
Distance (km)	12.6	12.6	0–53.1	0
Water (m^3)	6.25	4.26	3.06–6.58	−32
Energy[1] (MJ)				
Electricity	557	557	335–1,030	0
Diesel	69	69	0–318	0
Gasoline	9	9	0–34.0	0
Natural gas	4	4	0–102	0
Liquid propane gas	81	81	0–634	0
Output				
Egg production (t)	1	1	1	0
Eggs/100 layers per d	59.18	75.34	68.8–81.1	27
Eggs/layer per yr	216	275	251–296	27
Mass/egg (g)	60.5	60.0	54–63	−1
Spent hens[2]				
Mass (kg)	64.4	50	32.0–70.0	−22
Distance (km)	100	100	100	0
Manure hauled[3] (kg)	1,980	1,140	510–2,350	−42
Distance[4] (km)	14.4	14.4	0–32.2	0
Estimated N loss (kg)	61.7	32.4	32.4–45.3	−47
Estimated P loss (kg)	16.1	5.78	9.23–9.87	−64
Mortality[5]				
Rate (% per yr)	15.8	6.7	1.2–8.4	−57
Mass (kg)	11.6	5.47	1.10–11.0	−53

[1]Year 1960 data assumed same as 2010.

[2]34.5% to human consumption, 4.5% to pet food, 49.4% to rendering, 6.2% to composting, 5.0% to other.

[3]Manure mass at time of removal. Moisture content varies, depending on residency time and management strategy.

[4]Estimated distance at removed mass.

[5]Includes culls; 60.3% to rendering, 25.2% to composting, 0.5% to burial, 2.1% to landfill, 11.8% to incineration (assuming no energy recovery).

Table 10. Life cycle impact assessment results for acidifying emissions, eutrophying emissions, greenhouse gas (GHG) emissions, and cumulative energy demand (CED) per metric ton of feed inputs at the farm/processor gate in the United States in 1960 and 2010[1]

Feed ingredient	Year	Acidifying emissions (kg of SO_2-e)	Eutrophying emissions (kg of PO_4-e)	GHG emissions (kg of CO_2-e)	CED (MJ)
Corn	1960	7	2	345	1,380
	2010	5	1	301	1,759
CDDGS	1960	10	2	764	4,425
	2010	7	1	719	7,949
Soy meal	1960	7	1	249	1,337
	2010	4	1	227	2,601
Soy oil	1960	15	3	541	2,909
	2010	9	2	493	5,621
Bakery material	1960				
	2010	8	2	551	8,736
Wheat middlings	1960	10	2	430	2,364
	2010	10	2	490	4,222
Alfalfa hay	1960	2	1	101	499
	2010				
Fish meal	1960	6	3	714	4,620
	2010				
Poultry meat and bone meal	1960	191	71	6,472	31,165
	2010	121	45	4,605	42,437
Porcine meat and bone meal	1960	200	74	5,820	20,800
	2010	96	27	4,318	24,221
Ruminant meat and bone meal	1960	565	254	34,100	59,600
	2010	404	185	25,636	74,133
Poultry fat	1960	331	124	11,210	53,980
	2010	209	79	7,975	73,457
Porcine fat	1960	400	149	11,600	41,500
	2010	193	54	8,627	48,306
Ruminant fat	1960	1,136	511	68,468	119,788
	2010	812	371	51,546	148,951
Salt	1960	2	0	300	2,543
	2010	2	0	263	3,936
Limestone	1960	0	0	47	779
	2010	0	0	43	964
Calcium phosphate	1960	39	1	1,094	9,328
	2010	38	1	938	15,188

[1]e = equivalents. CDDGS = corn dried distillers grain with solubles.

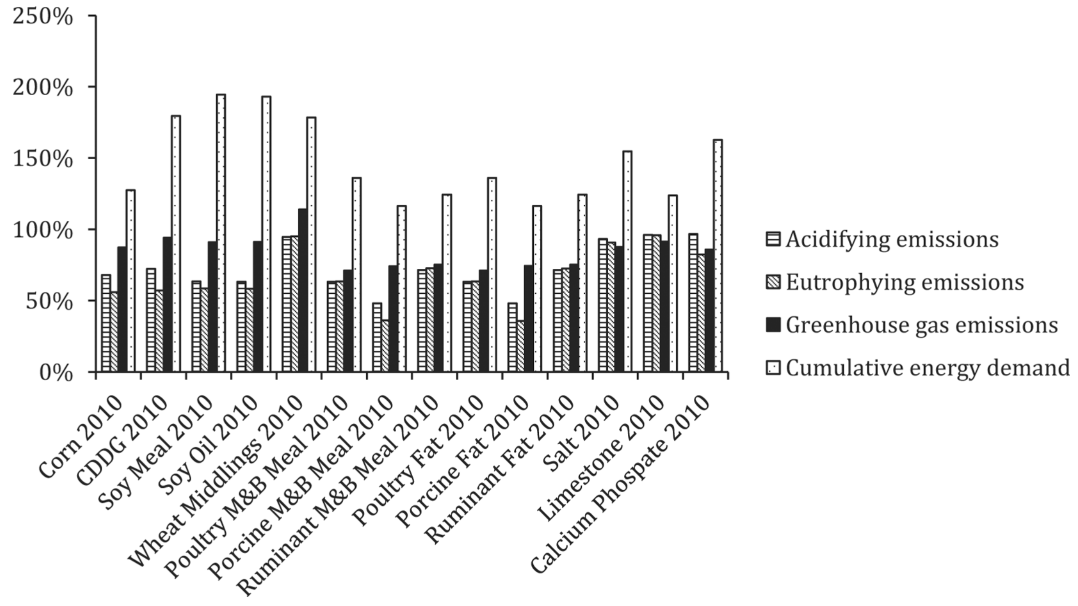

Figure 5. Life cycle impact assessment results for feed inputs to US pullet and layer systems (at the farm or processor gate) in 2010 compared with 1960 (all impacts for 2010 presented as a percentage of impacts in 1960). CDDG = corn dried distillers grains. M+B Meal = meat and bone meal.

Table 11. Life cycle impact assessment results for acidifying emissions, eutrophying emissions, greenhouse gas (GHG) emissions, and cumulative energy demand (CED) per metric ton of pullet and layer feeds produced in the United States in 1960 and 2010[1]

Feed and year	Acidifying emissions (kg of SO_2-e)	Eutrophying emissions (kg of PO_4-e)	GHG emissions (kg of CO_2-e)	CED (MJ)
Pullet feed 1960	18.4	6.8	1,015	3,139
Pullet feed 2010	9.8	2.9	584	4,267
Reduction (%)	47	57	42	−36
Layer feed 1960	34.5	13.8	1,860	4,560
Layer feed 2010	12.5	4.4	782	4,632
Reduction (%)	64	68	58	−1.6

[1]e = equivalents.

a comparable magnitude. In Canada, Vergé et al. (2009) calculated direct GHG emissions from layer facilities along with crops used to produce layer feeds in 1981 compared with 2006. Indirect supply chain emissions were not considered; hence, the study results are not directly comparable with those presented in the current analysis. Still, it is interesting to note that these authors found that the GHG intensity of egg production decreased from 1.9 kg of CO_2 equivalents/dozen eggs in 1981 to 1.76 kg of CO_2 equivalents/dozen eggs in 2006, an approximately 7% reduction over the 25-yr interval. Cederberg et al. (2009) compared GHG emissions from Swedish livestock production in 1990 and 2005 for pork, poultry meat, beef, milk, and eggs. They found that the carbon footprint of pork production decreased from 4 to 3.4 kg of CO_2 equivalents/kg over the 15-yr interval, emissions for poultry meat decreased from 2.5 to 1.9 kg of CO_2 equivalents/kg, and emissions for milk from 1.27 to 1 kg of CO_2 equivalents/kg. Emissions for beef production increased from 18 to 19.8 kg of CO_2 equivalents/kg. Emissions from egg production remained unchanged at 1.4 kg of CO_2 equivalents/kg over this interval. This latter finding was largely attributable to 2 factors: a) the phasing out of animal by-products in feeds as a result of bovine spongiform encephalopathy (**BSE**) concerns, and b) the use of economic allocation in modeling. Here, despite efficiency gains in the sector, the allocation strategy resulted in a study outcome suggesting no net gains in environmental performance.

To date, no other estimates for the life cycle impacts of contemporary national average US egg production are available. Pelletier et al. (2013) previously modeled egg production in Iowa using the same modeling approach as applied in this analysis. In the Iowa study, the authors did not identify the precise source (ruminant, swine, or poultry) of animal-derived meals and fats. However, they estimated that GHG emissions ranged from 2.0 kg of CO_2 equivalents (assuming 100% of the animal-derived products were of poultry origin) to 5.0 kg of CO_2 equivalents per kg of eggs produced (assuming 100% of the animal-derived products were of ruminant origin).

Several studies are available, however, that report environmental performance for egg production supply chains in other countries (Table 14). Although direct comparisons between studies are problematic due to frequent differences in modeling assumptions (e.g., system boundaries for the studies, data sources, allocation strategies, and so on), it is nonetheless interesting to consider the range of reported impacts relative to those of the current study.

In broad strokes, the distribution of impacts along contemporary US egg supply chains seems to be in gen-

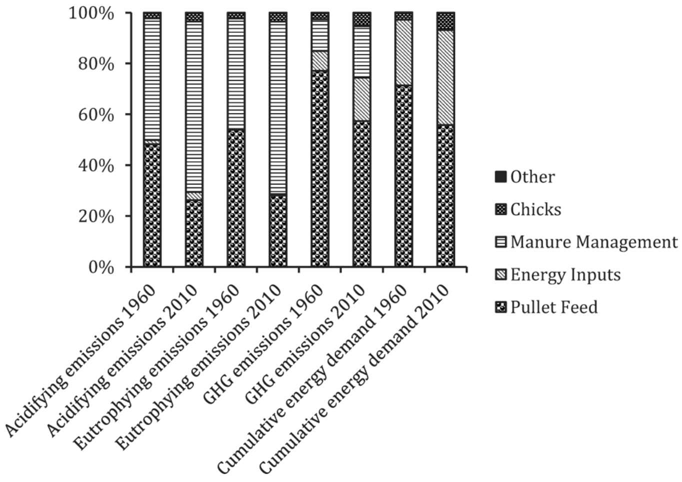

Figure 6. Contribution analysis for the life cycle impact assessment of pullets produced in the United States in 1960 compared with 2010. GHG = greenhouse gas.

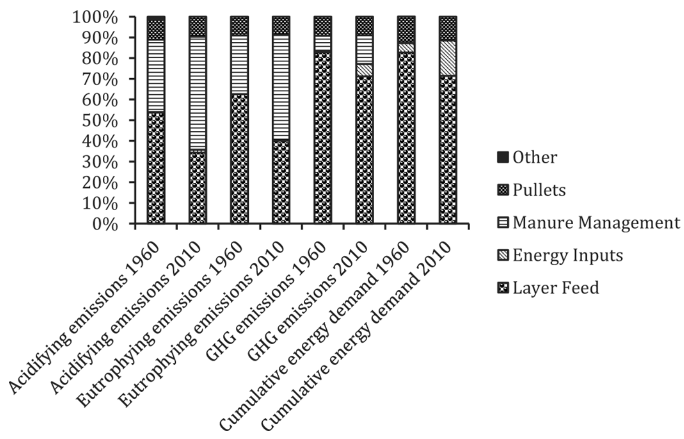

Figure 7. Contribution analysis for the life cycle impact assessment of eggs produced in the United States in 1960 compared with 2010. GHG = greenhouse gas.

Table 12. Life cycle impacts assessment results for acidifying emissions, eutrophying emissions, greenhouse gas (GHG) emissions, and cumulative energy demand (CED) for 1,000 pullets and 1 t of eggs produced in the United States in 1960 and 2010

Item	Acidifying emissions (kg of SO_2-e)		Eutrophying emissions (kg of PO_4-e)		GHG emissions (kg of CO_2-e)		CED (MJ)	
	Pullets	Eggs	Pullets	Eggs	Pullets	Eggs	Pullets	Eggs
Year								
1960	390	200	129	70	13,458	7,230	45	18
2010	196	70	54	20	5,404	2,080	41	12
Reduction (%)	50	65	58	71	60	71	9	31

eral agreement with similar, previously reported LCA research of intensive, cage egg production systems elsewhere (Mollenhorst et al., 2006; Cederberg et al., 2009; Vergé et al., 2009; Wiedemann and McGahan, 2011; Leinonen et al., 2012). In a study examining the social, economic, and ecological dimensions of egg production by housing system in the Netherlands, Mollenhorst et al. (2006) used LCA as a basis for comparing performance in the environmental domain. Conventional cage production was found to perform better according to the environmental LCA variables considered, but the aviary system performed better according to the economic and animal welfare measures employed. In Australia, Wiedemann and McGahan (2011) used a life cycle approach to evaluate GHG emissions, energy, and water use in egg production by housing system. Here, activity data were collected from 4 farms in eastern Australia. Cage systems were found to outperform free-range systems. Estimated impacts overall were low compared with results from most European studies. More recently, Leinonen et al. (2012) used top-down estimates of average UK production conditions in a standard, environmental LCA approach to characterize environmental performance for egg production in cage, barn, free-range, and organic systems. They reported highest impacts for organic production and lowest for cage production, largely due to differences in productivity (i.e., higher feed consumption and number of birds required per unit of egg production in the organic system). Feed production supply chains were the dominant contributor to GHG emissions (64–72%) and CED (54–75%). Similar to our study, energy use in housing systems was the second most important factor for the overall energy intensity of egg production. Manure management contributed most to acidifying and eutrophying emissions.

Where estimated impacts in these other studies (Table 14) are low compared with those of the present analysis, this is typically either because animal by-products are not allowed for use in animal feeds in the countries of concern (e.g., the Swedish study by Cederberg et al. for the 2005 system modeled), or because they were not included in the modeled feeds at all, whether or not they are actually used (e.g., the Australian study by Wiedemann and McGahan). In the latter study, the authors also point toward the low input nature of Australian grain production (compared with European norms) as an important factor influencing their reported outcomes. Considering the study of egg production in Sweden in 1995 compared with 2005 (Cederberg et al., 2009), the reduction in use of animal by-product due to legislative changes in response to BSE concerns over this interval in fact *negatively* affected performance in 2005 due to the use of economic allocation in this study. This is contrary to the results of the current analysis, which showed an improved environmental performance over time by reducing the amount of animal-derived materials used in poultry diets. In light of the resource and emissions intensity of producing livestock (along with the livestock processing coproducts used in animal feeds), the analytical approach of the current study better reflects the actual environmental costs of producing feed inputs for egg production, regardless of the economic value of such materials.

To put the GHG intensity of contemporary US egg production in perspective, the following comparison is provided. Using the same methods, Pelletier et al. (2010a) recently estimated the GHG emissions to be 3 kg of CO_2 equivalents per kg of live weight pork produced in the Midwestern United States. For conventional, feedlot beef production, the estimated GHG emissions were 14.5 kg of CO_2 equivalents per kg of live weight produced (Pelletier et al., 2010b). Similarly, an earlier study of US broiler production (Pelletier, 2008) revealed an estimated GHG emission of 1.7 kg of CO_2 equivalents per kg of live weight produced. Here, we

Table 13. Proportion (in %) of changes in the environmental footprint [acidifying emissions, eutrophying emissions, greenhouse gas (GHG) emissions, and cumulative energy demand (CED)] of egg production in the United States in 2010 compared with 1960 attributable to changes in background systems, feed composition, or bird performance due to improved husbandry and genetics

Footprint change attributable to changes in	Acidifying emissions	Eutrophying emissions	GHG emissions	CED
Background systems (%)	27	30	28	−116
Feed composition (%)	30	35	44	93
Bird performance (%)	43	35	28	123

Table 14. Reported life cycle impacts for acidifying emissions, eutrophying emissions, greenhouse gas (GHG) emissions, and cumulative energy demand (CED) per kilogram of eggs produced in different countries[1]

Study	Acidifying emissions (g of SO_2-e)	Eutrophying emissions (g of PO_4-e)	GHG emissions (kg of CO_2-e)	CED (MJ)
US average (this study)	70	20	2.1	12.3
United Kingdom[2]	53	77	2.9	16.8
The Netherlands[3]	32	25	3.9	—
Sweden[4]	—	—	1.4	—
Canada[5]	—	—	2.5	—
Australia[6]	—	—	1.4	—

[1]e = equivalents.

[2]Leinonen et al. (2012).

[3]Mollenhorst et al. (2006).

[4]Cederberg et al. (2009).

[5]Vergé et al. (2009).

[6]Wiedemann and McGahan (2011).

estimated a GHG intensity of 2.1 kg of CO_2 equivalents per kg of eggs produced in the continental United States in 2010, compared with 7.2 kg of CO_2 equivalents per kg of eggs produced in 1960.

Making a similar comparison on the basis of protein, the GHG intensity, expressed as kilograms of CO_2 equivalent emissions per kilogram of protein produced, is 19.1 for contemporary (2010) US egg protein (raw, from whole eggs), compared with 11.5 for broiler protein, 17.6 for pig protein, and 78.4 for beef protein.

Clearly, the US egg sector has made significant strides in improving resource utilization efficiency and reducing environmental impacts per unit of production since the 1960s. It is equally or more important to consider the extent to which such improvements have affected the total environmental footprint. The total US table egg production in 1960 was 59.8 billion eggs compared with 77.8 billion in 2010 (NASS, 2012), an increase of approximately 30%. Despite the substantial increase in production volume, the total CED in the US egg industry decreased by 13%, whereas total GHG emissions declined by 63%, total acidifying emissions by 54%, and total eutrophying emissions by 63%.

Conclusions and Recommendations

The distribution and magnitude of environmental impacts for US egg production in 2010 and 1960 were analyzed using LCA. The results clearly showed remarkable resource efficiency and environmental performance gains, both per unit production and in aggregate, achieved by the industry over the past 50 yr. The primary influencing factors and their relative contributions to the reductions in environmental footprint were elucidated. Specific insights and key findings are as follows.

From a supply chain management perspective, the key to improving environmental performance in egg production has been and will continue to be efforts to maximize feed efficiency. Feed conversion (feed to egg ratio) for egg production improved from 3.44 in 1960 to 1.98 in 2010, a 42% improvement. Achieving feed efficiencies comparable with the best-performing contemporary facilities (reported feed conversion ranged from 1.76 to 2.32) industry-wide would further reduce aggregate impacts.

Changing feed composition has also played an important role in reducing impacts. This is especially the case with both reduction in the total amount of animal-derived materials used and increased use of porcine and poultry materials in place of ruminant materials. The concept of least-environmental cost feed sourcing is therefore of particular relevance for additional targeted performance improvements for the egg industry. It is recommended that similar biophysical accounting methods to those applied in the current study be used to model potential alternative feed input supply chains to ensure methodological consistency and comparability with the present analysis.

Nitrogen losses from poultry manure are the second largest contributor to acidifying and eutrophying emissions, as well as a nontrivial contributor to GHG emissions for both pullet and layer facilities. Moreover, upstream impacts of N fertilizer production and use are a primary determinant of feed input-related impacts. Feed formulation, breeding, and manure management strategies for optimal N use efficiencies are therefore powerful tools in supply chain environmental management.

The benchmarks reported here, as well as the reported ranges for resource use and production efficiencies in otherwise similar production facilities, provide an excellent reference point for industry-led initiatives to further improve the environmental performance of US egg production.

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A Comparative Assessment of the Environmental Footprint of the U.S. Egg Industry in 1960 and 2010

Final Project Report

Submitted to

American Egg Board
United Egg Producers
Egg Industry Center

by

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EXECUTIVE SUMMARY

Food systems have been identified as a major contributor to environmental change at local, regional and global levels. Continuous progress towards more resource efficient and environmentally friendly food production norms are hence an important societal objective.

The US egg industry has evolved considerably over recent decades by incorporating new technologies and husbandry practices to make more efficient use of finite resources such as land, water and energy. Progress has been made on many fronts, including animal genetics, nutrition, disease prevention, housing equipment and environmental control, and efficiency of feed production and use. Contemporary productivity would have been difficult to imagine 50 years ago. However, to date there has been no comprehensive assessment of the resource demand and environmental effects of these changes in production practices and efficiencies.

Life cycle assessment (LCA) is the most widely used tool for studying environmental performance in food systems from a supply chain perspective. LCA is an ISO (14044) standardized framework for characterizing the material and energy flows and emissions along product supply chains, and quantifying how these contribute to a variety of resource use, human health, and environmental impact potentials. In this study, we used ISO-compliant LCA to quantify the environmental performance of US egg production in 2010 *vs.*1960.

Using industry-supplied activity data that were collected using anonymous surveys, this study first characterized the material and energy inputs and emissions associated with contemporary egg production supply chains in the United States. The system boundaries for this analysis included all cradle-to-facility gate direct and indirect inputs and emissions arising from: the agricultural and industrial production systems from which raw materials for feed inputs are derived; the processing of raw materials; the production of feeds; the production of chicks; and farm-level material and energy use and emissions of pullet and layer facilities. The data collected directly represented 57.1 million pullets and 92.5 million laying hens, or 26% and 33% of the respective stock populations in the United States in 2010. Subsequently, a parallel model of US egg production in 1960 was developed based on published literature sources and in consultation with industry experts for comparison with 2010 production conditions. The environmental footprint indicators used in this study were acidifying emissions (acidification), eutrophying emissions (eutrophication), greenhouse gas (GHG) emissions (global warming potential, GWP), and cumulative energy demand (CED).

In developing the 2010 and 1960 models, the following changes in production performance of pullets and laying hens in the United States were observed over time.

Compared with 1960 pullets, 2010 pullets have:

a) 30% lighter body weight at onset of lay (1.2 vs. 1.7 kg or 2.69 vs. 3.8 lb);
b) 48% less feed use over pullet-rearing period (5.3 vs. 10.2 kg or 11.6 vs. 22.4 lb); and
c) 70% lower mortality over pullet-rearing period (3.5% vs. 11.7%).

Similarly, compared with 1960 laying hens, 2010 laying hens have:

a) 26% less daily feed use (9.03 vs. 12.23 kg/100 hens or 19.9 vs. 26.9 lb/100 hens);
b) 27% higher hen-day egg production (75.3% vs. 59.2%);
c) 42% better feed conversion (1.98 vs. 3.44 kg or lb of feed per kg or lb of egg);
d) 57% lower mortality (6.7% vs. 15.8% per year); and
e) 32% less direct water use per dozen eggs produced (4.5 *vs.* 3.1 L or 1.2 *vs.* 0.8 gal).

Using the models developed for egg production supply chains in 1960 and 2010, the analysis showed the following reductions in the environmental footprint *per kg of eggs* produced in the United States over the 50-year time interval considered:

a) 65% lower acidifying emissions (70 vs. 200 g SO_{2eq});
b) 71% lower eutrophying emissions (20 vs. 70 g PO_{4eq});
c) 71% lower GHG emissions (2.1 vs. 7.2 kg CO_{2eq}); and
d) 31% lower CED (12.3 vs. 17.7 kJ).

The total supply of 77.8 billion eggs produced in the U.S. in 2010 was 30% higher than the 59.8 billion eggs produced in 1960. However, the *total* environmental footprint for 2010, *in million metric tonne of emissions and in million MJ for CED*, is:

a) 54% lower for acidifying emissions (0.329 vs. 0.724 SO_{2eq});
b) 63% lower for eutrophying emissions (0.094 vs. 0.253 PO_{4eq});
c) 63% lower for GHG emissions (9.8 vs. 26.2 CO_{2eq}); and
d) 10% lower for CED (57.9 vs. 64.1).

Further analysis found that using 1960 technologies to produce the amount of egg supply for 2010 would require the following additional resources: raising 27% (78 million) more hens, growing 72% (1.3 million acres or 0.53 million hectares, or 5.2 metric tonne) more corn, and growing 72% (1.8 million acres or 0.73 million hectares, or 1.7 metric tonne) more soybean. Demand for these additional resources would, in turn, translate into greater environmental impacts.

The analysis also identified areas for future improvement in the industry's environmental footprint. Feed efficiency, least-environmental cost feed sourcing, and manure management are the three primary factors that determine the environmental impacts of US egg production. Efforts focused on further research and improvements in these areas will therefore aid in continual reduction of the environmental footprint of the U.S. egg industry over time.

KEYWORDS: Eggs, life cycle assessment (LCA), pullets, production performance, environmental footprint, energy return on energy invested (EROI)

1. INTRODUCTION

Food systems have been identified as a major contributor to environmental change at local, regional and global levels (FAO 2006; Garnett 2008; Pelletier and Tyedmers 2011a). For example, it is estimated that food systems contribute 30% to anthropogenic greenhouse (GHG) emissions in the European Union (Tukker et al. 2006). Due to enhanced biological nitrogen fixation in agriculture and the production and use of nitrogen fertilizers, food production is also the primary source of reactive nitrogen mobilization, accounting for approximately 80% of anthropogenic fixation (Socolow 1999, Galloway *et al.* 2004, 2008). Moreover, the food sector is a key driver of biotic resource appropriation (Vitousek *et al.* 1986; Imhoff *et al.* 2004; Haberl *et al.* 2007) and consumes significant amounts of energy (Pimentel and Pimentel 1996; Pimentel *et al.* 2005). Given that total food production volumes are anticipated to almost double by 2050 (FAO 2006) to meet the demand of a growing and increasingly affluent population, how to meet these demands without severely compromising ecological integrity across scales constitutes to be a defining challenge for contemporary society (Pelletier *et al.* 2008; Pelletier and Tyedmers 2011a).

The development of technologically advanced food production, processing and distribution systems over the past 50 years has engendered both substantial productivity gains and environmental consequences, despite continuing increases in resource utilization efficiency. In recent decades, considerable research effort has therefore been invested in elucidating the material and energy dependencies and environmental impacts associated with diverse food production systems, including livestock systems.

Odum's pioneering work on the energetics of global food systems (Odum 1967) spawned a wealth of research regarding energy use in food production, much of which was led by American researcher David Pimentel (as summarized in Pimentel and Pimentel 1996). More recent work on food system energetics includes analyses of beef (Heitschmidt *et al.* 1996), conventional and organic dairy (Refsgaard *et al.* 1998), bread (Gronroos *et al.* 2006) and poultry production systems (Castellini *et al.* 2006).

Ecological footprint analyses have similarly been used as an indicator of biophysical sustainability in food systems, and have variously been applied to tomato, dairy and wine production (Wada 1993; Thomassen and de Boer 2005; Niccolucci *et al.* 2008), farms and cropland (van der Werf *et al.* 2007; Cuadra and Bjorklund 2008; Liu *et al.* 2008), several aquaculture products (Larsen *et al.* 1994; Kautsky *et al.* 1997;), and to quantify the resource appropriation associated with different dietary patterns (White 2000; Gerbens-Leenes and Nonhebel 2002).

Life cycle assessment (LCA) has been the most widely used tool for studying environmental performance in food systems from a supply chain perspective. LCA is an ISO (14044) standardized framework for characterizing the material and energy flows and emissions along product supply chains, and quantifying how these contribute to a variety of resource use, human health, and environmental impact potentials.

Most published LCA studies have treated single product systems or made comparisons between production technologies. Published studies have variously investigated oil seed crops (Schmidt 2007; Pelletier *et al.* 2008; Dalgaard *et al.* 2008), dairy systems (Cederberg and Mattsson 2000; Hogass-Eide 2002; Casey and Holden 2005; Olesen *et al.* 2006; Thomassen and De Boer 2008; Arsenault *et al.* 2009); beef production (Nunez *et al.* 2005;

Ogino *et al.* 2004, 2007; Casey and Holden 2006; Pelletier et al. 2010a), pork production (Nunez *et al.* 2005; Eriksson *et al.* 2005; Basset-Mens and van der Werf 2005; Pelletier et al. 2010b) and poultry production (Mollenhorst *et al.* 2006; Ellingsen and Aanondsen 2006; Williams *et al.* 2006; Pelletier 2008). Several studies of fisheries and aquaculture production systems have also been reported (Zeigler *et al.* 2003; Papatryphon *et al.* 2004; Hospido and Tyedmers 2005; Thrane 2006; Ellingsen and Aanondsen 2006; Mungkung *et al.* 2006; Pelletier and Tyedmers 2007; Gronroos *et al.* 2006; Ayer and Tyedmers 2009; Pelletier *et al.* 2009; Pelletier and Tyedmers 2010). What is clear from all of these studies is that the impacts of food production vary widely both within and between production technologies, as well as along different dimensions of environmental performance. Also clear is that mitigation strategies must be attentive to trade-offs across environmental domains and supply chain activities.

Agricultural production in the United States has advanced considerably over recent decades by incorporating new technologies to make more efficient use of finite resources such as land, water and energy. Egg production has followed the same trend, achieving productivity levels that would have been difficult to imagine half a century ago. However, to date there has been no comprehensive assessment of the environmental effects of these changes in production practices and efficiencies. In this study, we applied ISO-compliant LCA methods to quantify the changes in environmental performance in the US egg industry between 1960 and 2010 as a result of these changes in production efficiencies.

The specific objectives of the study were to:

1) Develop models of US egg production supply chains in 1960 and 2010 with regard to both foreground system variables (e.g. feed conversion efficiency, bird body weight, bird mortality rate, hen-day egg production, etc.) and background system variables (e.g. efficiencies of energy provision, fertilizer production, production of feed inputs, transport modes, etc.);
2) Characterize supply chain environmental performance for the US egg industry in 1960 and 2010 in terms of energy use, acidifying, eutrophying, and GHG emissions; and
3) Quantify the production performance gains and reduction in environmental impacts associated with technological and husbandry advancements over this 50-year interval.

The results of the study are intended to provide the US egg industry and other stakeholders with science-based information concerning the impact of technological advancements in egg production on resource efficiencies and environmental performance. The study will also offer insight as to key leverage points for further mitigation of environmental impacts and conservation of natural resources.

2. METHODS

2.1 GOAL AND SCOPE

An industry-wide, anonymous survey was conducted to acquire the necessary data for characterizing production performance and modeling the environmental footprint of the contemporary US egg supply chain. The collected data represented 57.1 million pullets and 92.5 million laying hens, accounting for 26% and 33% of the pullet and laying-hen populations, respectively. The system boundaries for this analysis included all cradle-to-facility gate direct and indirect inputs and emissions arising from: the agricultural and industrial production systems from which raw materials for feed inputs are derived; the processing of raw materials; the production of feeds; the production of chicks; and farm-level material and energy use at pullet and layer facilities (Figure 1). In the absence of company-specific information for hatcheries, data were adopted from an earlier study of US broiler production systems (Pelletier 2008). This analysis did not include emissions associated with the production and maintenance of infrastructure such as machinery and buildings (these typically make trivial contributions to supply chain emissions in high production volume contexts, since they must be amortized against total production over their anticipated lifespan – for example, see Ayer and Tyedmers 2009).

We then developed a parallel model of US egg production in 1960 based on published literature sources and in consultation with industry experts. These parallel models were subsequently used to quantify and evaluate the environmental performance of each supply chain node in terms of cumulative energy demand (CED), greenhouse gas (GHG), acidifying and eutrophying emissions in 1960 versus 2010.

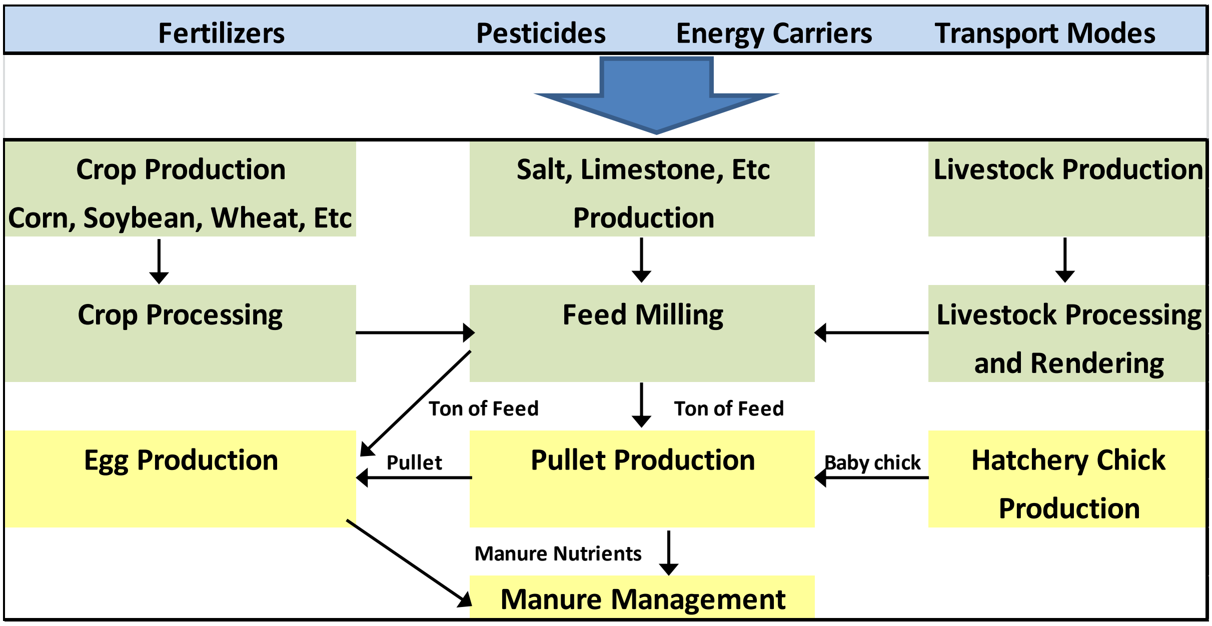

Figure 1. System boundaries for a life cycle assessment of egg production in the United States in 1960 and 2010 (background processes such as fertilizers, pesticides, and transport modes were derived from the EcoInvent (2010) database but were modified to reflect US energy carriers).

2.2 LIFE CYCLE INVENTORY

The life cycle inventory phase of LCA requires compiling inventory data representing the material and energy inputs and outputs at each stage of the supply chain of interest. Data for each supply chain node are expressed in terms of a relevant unit of analysis.

2.2.1 The 2010 Model

Foreground system data refer to information unique to the product system of interest. Foreground system data for feed milling, pullet and layer facilities were collected via anonymous surveys from participating companies. As previously stated, the data collected represented 57.1 million pullets and 92.5 million laying hens – accounting for 26% of pullet stock and 33% of laying-hen stock in the United States in 2010.

Background system data refer to information regarding processes linked to the foreground system in the supply chain of interest, but shared with other supply chains. In the context of our analysis, this includes the provision of energy carriers, inputs to crop production and other feed input production and processing systems, and transportation modes.

Background system data for the production and processing of feed ingredients were adapted from recent LCA studies by Pelletier et al. (2010 a,b) of beef and pork production supply chains in the Upper Midwestern United States, and global salmon aquaculture supply chains (Pelletier et al. 2009) (details below). These studies used identical modeling parameters to those of the current analysis and hence the feed input models could be directly adopted. Other background system data, including the provision of energy carriers, fertilizers, pesticides, and transportation models, were derived from the EcoInvent (2010) database and modified to reflect US energy inputs.

2.2.1.1 Agricultural Feed Ingredient Models

Inventory data for wheat, soy and corn-based feed inputs were derived from US National Agricultural Statistics Service (NASS) publications, Iowa State University extension publications and peer-reviewed literature. Yields were based on 5-year averages for 2005-2010 calculated from NASS (2012) data. Fertilizer and pesticide mixes and application rates correspond to average US consumption for each crop as reported by NASS (2012) and International Fertilizer Association (IFA 2012). Energy inputs to cropping systems were also based on US averages (NASS 2004). Field-level ammonia, nitrous oxide, nitric oxide, nitrate and carbon dioxide (from urea fertilizers) emissions were calculated following IPCC (2006) Tier 1 protocols using relevant default emission factors. A 2.9% surplus phosphate emission rate was assumed following Dalgaard *et al.* (2008). All fertilizers and pesticides were assumed to be transported 1000 km (625 miles) by truck, and all seed inputs 100 km (62.5 miles) by truck. Processing of wheat, soy and corn was based on inventory data reported by Pelletier *et al.* (2009; 2010 a,b). Data for the production of ruminant and porcine meat and bone meal and fat followed those in Pelletier *et al.* (2010 a,b) and Pelletier *et al.* (2009).

2.2.1.2 Modeling N and P Emissions

Nitrogen and phosphorus emission rates were calculated using a nutrient balance model based on feed composition and assuming that 2.2% of hen body mass is nitrogen and 0.6% is phosphorus, whereas eggs are assumed to contain 1.7% nitrogen and 0.21% phosphorus following Koelsh (2007). Nitrogen excretion estimates were subsequently used to calculate direct nitrous oxide, ammonia and nitric oxide emissions from manure management and indirect nitrous oxide emissions from nitrate leaching and ammonia emissions following IPCC (2006) protocols and relevant Tier I and Tier II emission factors at time of deposition, storage and application. Methane emissions from manure management were calculated following IPCC (2006) Tier I protocols. Phosphorus emissions were calculated at a 2.9% leaching rate at time of application of manure to agricultural lands following Dalgaard *et al.* (2008).

2.2.1.3 Co-product Allocation

Co-product allocation is required to apportion resource use and emissions between the products of multi-output systems. Since the purpose of the present analysis is to describe the cause-effect biophysical flows and associated environmental impacts of a food production system, it was deemed appropriate to base allocation decisions on an inherent biophysical characteristic of co-products which is also relevant to the function provided by the product system. To this end, the gross chemical energy content of co-product streams was used 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 ecological efficiency with which the system provides available food energy. For a detailed discussion of this rationale, see Pelletier and Tyedmers (2011b). This approach was chosen over economic allocation, which is sometimes used in reported food system LCAs, because (1) economic allocation is a last-resort option in the ISO 14044 hierarchy and (2) the use of economic allocation typically produces results that poorly reflect the physical reality of the systems that are modeled. The use of substitution (following a consequential data modeling approach) was similarly deemed inappropriate for our analysis, which intends to establish a baseline rather than to model market-level consequences of possible changes in production systems.

2.2.2 The 1960 Model

In developing a model to represent average US egg supply chain characteristics in 1960, we consulted a variety of expert sources and published literature. This required estimating performance efficiencies for both foreground (e.g., egg production rate, feed conversion, bird mortalities) and background production system variables (e.g., provision of energy carriers, production of inputs to cropping systems, production of feed inputs, transportation modes, etc.). Where we were not able to identify a robust basis for characterizing specific foreground system variables for 1960 (e.g., energy use in poultry housing systems) we used 2010 data in proxy, but with modifications to accommodate our 1960's background system variables. This almost certainly resulted in an underestimation of differences in the environmental performance of egg production in 1960 versus 2010. Modeling of key variables for 1960 is described below.

2.2.2.1 1960 Energy Carriers

Energy return on energy invested (EROI) is a measure of the energy efficiency of energy production. Specifically, it is a dimensionless indicator of the amount of energy that is required to produce and bring to market an equivalent unit of a given energy carrier (for example, oil, gas, or electricity). Several researchers have reported declining EROI values for different energy carriers over time. This is because, as easily accessible, high-quality energy resources are exhausted, an increasing proportion of energy production derives from less-accessible, marginal energy resources that are more energy-intensive to exploit. In short, over time, more energy is required to produce an equivalent unit of energy. From a life cycle perspective, taking into account this changing efficiency and the associated changes in environmental burdens is essential to realistic, time-sensitive modeling.

EROI values at any given time differ between energy carriers, region of production, and production technology. Moreover, EROI can be described from both production and consumption perspectives. Since energy commodities are widely traded, calculating EROI values for energy carriers consumed in a given jurisdiction requires attention to trade patterns and, in the case of electricity, regionally-specific energy mixes.

For the purpose of the present analysis, EROI values for the United States as well as global EROI values for the production of specific energy carriers were adopted from or calculated based on the work of Lambert et al. (2012), Gangon and Hall (2009), Balogh et al. (in prep) and Guildford et al. (2011). In turn, these were used to calculate EROIs for primary energy carriers consumed in the United States in 1960 and 2010 based on USEIA (2012) statistics for US consumption and imports of energy products. USEIA (2012) statistics for the energy mixes used in US electricity production were also used to calculate 1960 and 2010 EROI values for electricity consumption (Table 1). On this basis, scaling factors were derived to represent the comparative EROI of energy carriers between 1960 and 2010. These factors were applied to modify the life cycle inventories used for 2010 energy carriers (adapted from the EcoInvent database) in order to arrive at 1960 energy carrier life cycle inventories which approximate changes in the environmental performance profile of energy carriers used in the United States over this interval. Potential differences in distribution losses for electricity (grid efficiencies) in 1960 compared to 2010 were not considered.

Table 1. Estimated EROI values for energy consumed in 1960 and 2010 in the United States.

Energy Carrier	1960	2010	Scaling Factor between 2010 and 1960
Coal	75	60	0.8
Oil/Gas	47	15	0.3
Nuclear and Renewables	15	15	1.0
Electricity	14	14	1.0

2.2.2.2 1960 Fertilizer Production

US fertilizer mixes for 1960 were derived from IFA statistics (IFA 2012). Ammonia production accounts for 87% of the fertilizer industry's energy consumption (IFA 2009). Based on data regarding improvements in the efficiency of ammonia plants over time, IFA (2009) shows that efficiencies improved from 58 to 28 MJ of energy required per tonne of ammonia produced between 1960 and 2010. Effectively, this means that producing ammonia in 1960 required 2.07 times as much direct energy input as in 2010. This ratio was hence applied in order to scale the energy inputs for average contemporary ammonia production for the EcoInvent life cycle inventory used to represent contemporary ammonia production in order to arrive at a representative 1960 life cycle inventory.

For all other fertilizer "building blocks," Kongshaug (1998) provides estimates of net energy consumption for "old technology - 1970", "average technology - 1998" and "best available technology – 1998." These estimates largely distinguish between net energy production in the form of steam, which may or may not be productively utilized. The modified EcoInvent processes for fertilizer production (originally representing average EU production, but modified to reflect US energy inputs) used in the present analysis assume that net energy produced is lost as waste heat. For the purpose of our analysis, we similarly adopted this assumption; hence we do not distinguish between sulphuric acid, nitric acid and phosphoric acid net energy production in 1960 versus 2010 (although we do apply the modified energy carrier inventories in the 1960 fertilizer production models).

2.2.2.3 1960 Freight Transport

United States Department of Energy (USDE) data were used to calculate differences in the energy efficiency of freight transport by mode in 1960 compared to 2010 (USDE 2012). The energy intensity of US heavy truck freight decreased from 24,960 BTU per vehicle mile in 1970 to 21,463 BTU per vehicle mile in 2010, with an average annual decrease of 0.4%. Making a linear extrapolation to 1960 on this basis, estimated energy intensity of road freight was 25,977 BTU per vehicle mile. A correction factor of 1.21 was therefore applied to the EcoInvent models used to represent US road freight energy use in 2010 for the 1960 model.

The energy intensity of US rail freight decreased from 691 BTU per ton-mile in 1970 to 289 BTU per ton-mile in 2010, with an average annual decrease of 2.2%. Making a linear extrapolation to 1960 on this basis, estimated energy intensity of US rail freight was 859 BTU per ton-mile. A correction factor of 2.97 was therefore applied to the EcoInvent model used to represent US road freight energy use in 2010 for the 1960 model.

USDE (2012) only provides data for changes in the energy intensity of water freight on taxable waterways from 1997 (266 BTU per ton-mile) to 2010 (217 BTU per ton-mile), with an average annual decrease of 2.20%. Extrapolating back to 1960 suggests an energy intensity of 595 BTU per ton-mile in 1960, which would imply a correction factor of 2.74. This is very similar to the estimated correction factor for rail freight extrapolating from 1970-2010 time series data. This estimate is the weakest given that efficiency in 1960 is extrapolated from only 14 years of data spanning 1997-2010.

For comparison, using data from Fernley's Review for world seaborne trade from 1969-2010 (http://www.marisec.org/shippingfacts/worldtrade/volume-world-trade-sea.php) and estimates of marine fuel use from 1950-2010 (Eyring et al. 2005), the estimated correction factor for global ocean freight is 1.33. Elsewhere, based on a review of Llyod's Register data, it has been suggested that the energy efficiency of ocean container freight has increased 35% between 1985 and 2008, suggesting an annual increase of 1.52% per year (http://www.worldshipping.org/benefits-of-liner-shipping/low-environmental-impact). However, for consistency with our calculations for road and rail freight, we adopted the correction factor of 2.74.

2.2.2.4 1960 Feed Input Models

Smil et al. (1983) report energy inputs to US corn production for 1959. On this basis, we calculate that direct energy inputs have declined 61% per unit production compared to reported energy inputs to corn production in 2001 (which we adopt for 2010) as estimated by NASS (2004). No similar estimates are available for our 1960s models for soy and wheat, hence we assume a proportionate decline in energy inputs relative to NASS (2004) energy use estimates for soybeans in 2002 and wheat in 1998. Pesticide use for crops is based on statistics for 1964 provided by USDA (1995). Fertilizer use is also based on statistics for 1964 provided by USDA (2012). We assume sulphur and lime inputs are similar between 1960 and 2010. Crop yield data for 1960 is taken from the USDA Feed Grains Database and USDA Oil Seeds Database.

All animal-derived and other feed inputs are based on the LCA models reported by Pelletier et al. (2009) (for fish meal) and Pelletier et al. (2010 a,b) (for porcine and ruminant materials), which were created using identical modeling protocols to those used for the 2010 model in the current analysis. For ruminant production, we used Pelletier et al.'s model for grass-fed beef production to represent 1960's conditions (versus their model of conventional, feedlot production to represent 2010 conditions). For pig production, we used Pelletier et al.'s model for low-performance niche production to approximate 1960's conditions (versus their model of conventional, commodity production to represent 2010 conditions). In the absence of an alternative model for broiler poultry production, we assumed that the spent layers destined for rendering as modeled in the current analysis are used for the production of poultry by-product meal and fat.

2.2.2.5 1960 Pullet and Layer Production

Animal husbandry performance data for pullet and layer production were taken from Winter and Funk (1960), and verified with industry experts. For pullets, this included feed composition, feed consumed per pullet sold, mortality rate (% of initial placement), and age and body weight of pullets at the time of moving into the layer houses. For layers, this included feed composition, feed consumption per day, egg production/layer/year, egg weight, feed conversion ratio, mortality rate, and number of pullets added to layer houses per year.

2.3 IMPACT ASSESSMENT AND INTERPRETATION

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 quantified cumulative energy use, GHG, acidifying and eutrophying emissions.

Energy use (MJ) was quantified following the Cumulative Energy Demand (CED) method (Frischnect *et al.* 2003), which accounts for conversion efficiencies and the quality of energy inputs. Global warming (CO_2-equivalency over a 100-year time horizon according to IPCC 2006), acidification (SO_2-equivalency), and eutrophication (PO_4-equivalency) potentials were quantified according to the CML 2 Baseline 2000 method (Guinee et al. 2001). These assessment methods follow the problem-oriented mid-point approach, meaning that results are expressed in terms of their potential environmental impacts (as measured in resources used or emissions to the environment) rather than actual damage levels.

We first calculated impacts per relevant unit of analysis for each supply chain node considered, and for supply chains in aggregate. Results for the 1960 and 2010 models were subsequently compared in order to determine differences in production efficiencies and environmental performance over time.

We also conducted more detailed contribution analyses in order to determine to what extent observed differences in environmental performance between egg production in 1960 and 2010 were attributable to different factors or model assumptions. The first such analysis evaluated the influence of differences in background system variables between 1960 and 2010 only (i.e., production efficiencies for energy carriers, fertilizers, transport modes, and feed inputs). Here, we replaced all 1960 sub-models with 2010 models for these parameters. The second analysis used the same feed composition as 2010 in the 1960 model, and also replaced all 1960's background system sub-models with 2010 models in order to determine the differences strictly attributable to changes in either feed composition or animal husbandry practices and performance over time.

3. RESULTS AND DISCUSSION

3.1 LIFE CYCLE INVENTORY RESULTS

Tables 2-9 report the life cycle inventory data employed for the 2010 and 1960 models of US egg production supply chains. Inventory data for production and processing of individual feed ingredients (other than corn, wheat and soy) are not provided herein but can be found in the studies by Pelletier et al. (2009, 2010a,b).

In spite of suggestions in the literature that inputs to US cropping systems (fertilizers and pesticides) have increased substantially over time, when compared per tonne of crop yield as opposed to per hectare cultivated it is apparent that this generalization is overly simplistic (Table 2). Substantial increases in yield over the 50-year interval considered, in many cases, offset per hectare increases in inputs, with some inputs higher in 1960 or in 2010, depending on the input and crop.

For feed milling, the reported proportions and total amounts of different energy carrier inputs per tonne of feed milled is highly variable (Table 3), as are the distances travelled for the feed inputs sourced (Table 4). For the purpose of our analysis, we applied total consumption-weighted averages to arrive at the proportions and feed transport distances we modeled.

Reported data were similarly variable for pullet and layer facilities for parameters such as water use, energy use, manure mass, etc. Again, although we also include the ranges of reported values in the proceeding tables, production-weighted averages were used to construct the life cycle inventory model.

Table 2. Life cycle inventory data per tonne of corn, soy and wheat produced in 1960 and 2010. For other feed input life cycle inventory data, see Pelletier et al. (2009, 2010a,b).

	2010			1960		
INPUTS	**Corn**	**Soy**	**Wheat**	**Corn**	**Soy**	**Wheat**
Fertilizer (kg)						
N	16.1	1.12	20.1	16.6	0.74	9.17
P_2O_5	5.55	5.53	6.91	10.8	2.72	7.03
K_2O	5.71	7.75	1.36	8.50	3.35	3.93
Sulphur	0.27	0.13	0.53	0.27	0.13	0.53
Lime	33.5	0.00	0.00	33.4	0.00	0.00
Energy						
Diesel (l)	4.49	10.9	13.2	4.47	17.5	21.3
Gas (l)	1.17	3.49	3.02	12.1	5.62	4.86
LPG (l)	7.02	0.00	3.82	2.68	0.00	6.16
Elect. (kWh)	4.33	0.00	11.9	0.00	0.00	19.19
Total Pesticides (kg)	0.25	0.46	0.29	0.20	0.21	0.12
Herbicides	0.24	0.45	0.12	0.13	0.09	0.11
Insecticides	0.01	0.01	0.00	0.08	0.11	0.01
Other (fungicides)	0.00	0.00	0.17	0.00	0.00	0.00
Seed (kg)	2.10	23.4	34.5	20.5	45.0	41.8
OUTPUTS						
Nitrous Oxide (kg)	0.46	0.25	0.55	0.49	0.27	0.36
Ammonia (kg)	2.38	2.19	4.13	3.57	3.91	4.46
Nitric Oxide (kg)	0.35	0.02	0.43	0.36	0.02	0.20
Carbon Dioxide (kg)	17.2	0.17	3.04	14.3	0.03	0.42
Nitrate (kg)	1.44	0.00	0.00	4.49	0.00	0.00
Phosphate (kg)	0.00	0.00	0.03	0.14	0.00	0.00
Yield (tonne)	1.00	1.00	1.00	1.00	1.00	1.00

Table 3. Energy inputs per tonne (1000 kg or 2200 lb) of pullet/layer feed milled in reporting facilities in the United States in 2010 (representing a total production of 2,679,405 tonne of feed). This dataset was also used for the 1960 model.

	Production-weighted Average	**Range**
Electricity (MJ)	15.8	1.8-52.9
Diesel (MJ)	51.1	0-122.8
Gasoline (MJ)	1.5	0-3.4
Natural Gas (MJ)	0	0-0.02

Table 4. Distances travelled for inputs to pullet/layer feed milled in reporting facilities in the United States in 2010 (representing a total production of 2,679,405 tonnes). This dataset was also used for the 1960 model.

Feed Input	Distance to Processor[1] (km)	Distance to Feed Mill[2] (km)	Range
Corn		27	24-48
Corn Dried Distillers Grains with Solubles (CDDGS)	25	116	1-193
Soy Meal	100	96	29-133
Bakery Material	wheat: 100 to flour mill, flour: 1000 to bakery	258	97-587
Wheat Middlings	100	474	241-604
Meat and Bone Meal	100	151	56-322
Fat	100	272	0-579
Salt	25	370	0-861
Limestone	100	142	0-241
Calcium	100	186	137-225
Phosphate	100	239	0-861
Trace Vitamins	100	325	0-563

(1) Assumed average distances
(2) Production-weighted average

Both the types and inclusion rate (%) of ingredients in pullet and layer feeds have changed between 1960 and 2010 (Tables 5 and 6). While corn and soy products constitute the core bulk ingredients for both periods considered, wheat was a more important input in 1960 than in contemporary egg production. Several ingredients also figure in only one period or the other – for example, green feed and fish meal in 1960 pullet feeds, and bakery material in 2010 pullet and layer feeds. Notable here is the reduced fraction of animal-derived materials (roughly 50% of 1960 levels) in contemporary feeds. The nitrogen and phosphorus percentages in different feed ingredients, as used to estimate the nutrient balance, are listed in Table 7.

Table 5. Pullet feed composition for egg production in the US in 1960 (based on Winter and Funk 1960) and 2010 (based on the production-weighted average of feed composition data from the reporting pullet producers).

	1960 % inclusion	2010 % inclusion	2010 range
Corn	78.1	60.0	41.0-70.7
Corn Dried Distillers Grains with Solubles (CDDGS)	1.0	6.2	0-13.0
Soy Meal	10.3	21.0	13.0-27.0
Dehydrated Green Feed[1]	3.0	0.0	N/A
Fish meal	1.2	0.0	N/A
Bakery Material	0.0	1.0	0-13.0
Wheat Middlings	0.0	0.9	0-7.0
Meat and Bone Meal[2]	2.5	1.0	0-5.7
Fat[3]	0.3	0.9	0-1.7
Salt	0.5	0.3	0-0.4
Limestone	1.5	6.2	0-10.5
Dicalcium Phosphate	0.6	0.0	N/A
Calcium	0.0	1.3	0-10.0
Phosphate	0.0	0.7	0-1.5
Other[4]	1.0	0.5	0-2.1

(1) Modeled as alfalfa hay based on Pelletier et al. (2010a)
(2) 63% ruminant, 26% porcine, 11% poultry (assumed same as 2010)
(3) 50% poultry, 50% vegetable (assumed to be soy oil) (assumed same as 2010)
(4) Includes trace vitamins and minerals, modeled as DL-methionine

Table 6. Layer feed composition for egg production in the US in 1960 (based on Winter and Funk 1960) and in 2010 (based on feed composition data from the reporting egg producers).

	1960 % inclusion	2010 % inclusion	2010 range
Corn	63.9	58.6	40.5-69.2
Corn Dried Distillers Grains with Solubles (CDDGS)	0	6.1	0-15.1
Soy Meal	12	19.3	10.0-25.7
Bakery Material		0.9	0-12.4
Wheat Middlings	10	0.8	0-9.9
Dehydrated Green Feed[1]	2.5	0	N/A
Meat and Bone Meal[2]	5	1.8	0-7.8
Fat[3]	1	0.9	0-4.4
Salt	0.5	0.3	0-1.0
Limestone	3.7	6.8	0-11.6
Dicalcium Phosphate	1.3	0	N/A
Calcium	0	2.1	0-9.8
Phosphate	0	0.5	0-1.0
Other[4]	0.1	0.5	0-1.8

(1) Modelled as alfalfa hay
(2) 81% ruminant, 17% porcine, 2% poultry
(3) 4% ruminant, 2% porcine, 58.5% poultry, 35.5% vegetable (assumed to be soy oil)
(4) Includes trace vitamins and minerals, modeled as DL-methionine

Table 7. Proximate composition of feed inputs used for calculating intake, excretion and losses of N and P.

Feed Ingredients	% N	% P
Corn	1.224	0.260
Corn Dried Distillers Grains with Solubles (CDDGS)	4.224	0.710
Soybean Meal	6.899	0.620
Bakery By-product	1.728	0.250
Wheat Middlings	2.706	0.910
Alfalfa Hay (17% CP)	2.720	0.250
Meat and Bone Meal	8.000	4.000
Fish Meal (66% CP)	10.56	3.150
Fat	0	0
Limestone	0	0.020
Phosphate	0	0.4364
Trace Vitamins	0	0
Methionine	8.750	0

Perhaps most striking at the inventory level are the differences in resources consumed and other performance parameters for pullet (Table 8) and layer (Table 9) production in 1960 compared to 2010. Feed consumption per pullet decreased by 48% over the 50-year interval considered, in part explained by a 30% lower body weight at the onset of production (which requires less feed) and in part by a 70% lower mortality rate, therefore less feed input into pullets that then died (Table 8). As a result of reduced mortality, the number of chicks required (per thousand pullets produced) has also decreased by a net 8.6% (Table 8). At the same time, estimated losses of N and P have decreased by 39% and 60%, respectively. Unfortunately we were unable to find data for energy inputs to pullet facilities in 1960; hence we simply assumed comparable energy use to 2010.

Table 8. Life cycle inventory data for the production of 1000 pullets in the United States in 1960 (based on Winter and Funk 1960) and in 2010 (based on the production-weighted average data from the reporting pullet producers representing 57,116,182 pullets).

	1960 average	2010 average	2010 range	Percent Change
Chicks	1133	1036	1021-1047	-9%
Mass/chick (g)	39.8	39.8	39.1-40.0	0%
Distance (km)	434	434	32.2-845	0%
Feed (kg)	10.2	5.27	4.31-5.75	-48%
Distance (km)	19.2	19.2	0-112	0%
Water[1] (m^3)	17.9	9.22	7.54-10.1	-48%
Energy[2] (MJ)				
Electricity	3015	3015	1425-5721	0%
Diesel	105	105	0-1084	0%
Gasoline	95.8	95.8	0-517	0%
Propane	1654	1654	0-4747	0%
Natural Gas	187	187	0-1932	0%
Fuel Oil	2.63	2.63	0-158	0%
OUTPUT				
Pullets	1000	1000	1000	0%
Mass (tonne)	1.74	1.22	1.16-1.30	-30%
Manure[3] (tonne)	6.46	3.38	0.59-4.59	-48%
Distance[4] (km)	10.0	10.0		0%
Estimated N loss (kg)	178	108	81.9-122	-39%
Estimated P loss (kg)	32.9	13.3	9.09-15.7	-60%
Body weight (kg/bird)	1.7	1.2	1.16-1.30	-30%
Mortality rate (%)	11.7	3.5	2.1-4.7	-70%

(1) Water use estimated as 1.75 x feed input.
(2) Year 1960 data assumed to be same as 2010
(3) Manure mass on an as-removed basis, assuming proportionate to the ratio of feed use to manure production in 2010
(4) Assumed distance of travel from farm to destination of manure application

Table 9. Life cycle inventory data per tonne of eggs produced in the United States in 1960 (based on Winter and Funk 1960) and in 2010 (based on the production-weighted average data from the reporting egg producers representing 1,542,507.6 tonnes of eggs).

	1960 average	**2010 average**	**2010 range**	**Percent Change**
Pullets	46	36	21-50	-22%
Distance (km)	52.9	52.9	1.61-452	0%
Layer Feed Consumption				
kg/100 layers/day	12.23	9.03	8.1-11.3	-26%
kg of feed/kg of eggs	3.44	1.98	1.76-2.32	-42%
Distance (km)	12.6	12.6	0-53.1	0%
Water (m^3)	6.25	4.26	3.06-6.58	-32%
Energy[1] (MJ)				
Electricity	557	557	335-1030	0%
Diesel	69	69	0-318	0%
Gasoline	9	9	0-34.0	0%
Natural Gas	4	4	0-102	0%
LPG	81	81	0-634	0%
OUTPUT				
Egg Production	1	1	1	0%
Eggs/100 layers/day)	59.18	75.34	68.8-81.1	27%
Eggs/layer/year	216	275	251-296	27%
Mass/egg (g)	60.5	60.0	54-63	-1%
Spent Hens[2]				
Mass (kg)	64.4	50	32.0-70.0	-22%
Distance (km)	100	100	100	0%
Manure hauled[3] (kg)	1980	1140	510-2350	-42%
Distance[4] (km)	14.4	14.4	0-32.2	0%
Estimated N loss (kg)	61.7	32.4	32.4-45.3	-47%
Estimated P loss (kg)	16.1	5.78	9.23-9.87	-64%
Mortalities[5]				
Rate (% per year)	15.8	6.7	1.2-8.4	-57%
Mass (kg)	11.6	5.47	1.10-11.0	-53%

(1) Year 1960 data assumed same as 2010
(2) 34.5% to human consumption, 4.5% to pet food, 49.4% to rendering, 6.2% to composting, 5.0% to “other”.
(3) Manure mass at time of removal. Moisture content varies, depending on residency time and management strategy.
(4) Estimated distance at removed mass.
(5) Includes culls. 60.3% to rendering, 25.2% to composting, 0.5% to burial, 2.1% to landfill, 11.8% to incineration (assuming no energy recovery).

For egg production, the lower bird body weight (4.5 lb/layer in 1960 vs. 3.4 lb/layer in 2010) is one of the main drivers for the 26% lower feed consumption per hen in 2010 (Table 9). The lower daily feed use, combined with a 27% higher hen-day egg production and a 57% lower mortality rate, results in a 42% less feed consumed per kg of egg produced, i.e., improvement in feed conversion. The number of pullets sourced per tonne of eggs produced has decreased by 22% (Table 9) because of the lower mortality. Nitrogen and phosphorus emissions have decreased by 47% and 64%, respectively.

3.2 LIFE CYCLE IMPACT ASSESSMENT AND INTERPRETATION OF RESULTS

3.2.1 Life Cycle Impact Assessment Results for Energy Carriers in 1960 versus 2010

Energy return on energy invested (EROI) was substantially higher (i.e. 35%-65%) in 1960 for all primary energy carriers other than coal (Figure 2). The low EROI for coal in 1960 is explained by the low energy costs of extracting coal relative to the energy costs of transporting coal to markets. Since rail and water freight transport modes were considerably less energy efficient in 1960, the end result is a lower overall EROI for coal in 1960 compared to 2010. Emissions for electricity production are also slightly higher in 1960 compared to 2010, largely due to two factors. First is the higher fraction of (in particular) coal and other fossil fuels in the 1960 energy mix compared to a greater share of nuclear power generation in 2010. Second is the lower efficiencies of transforming primary energy carriers into electricity in 1960.

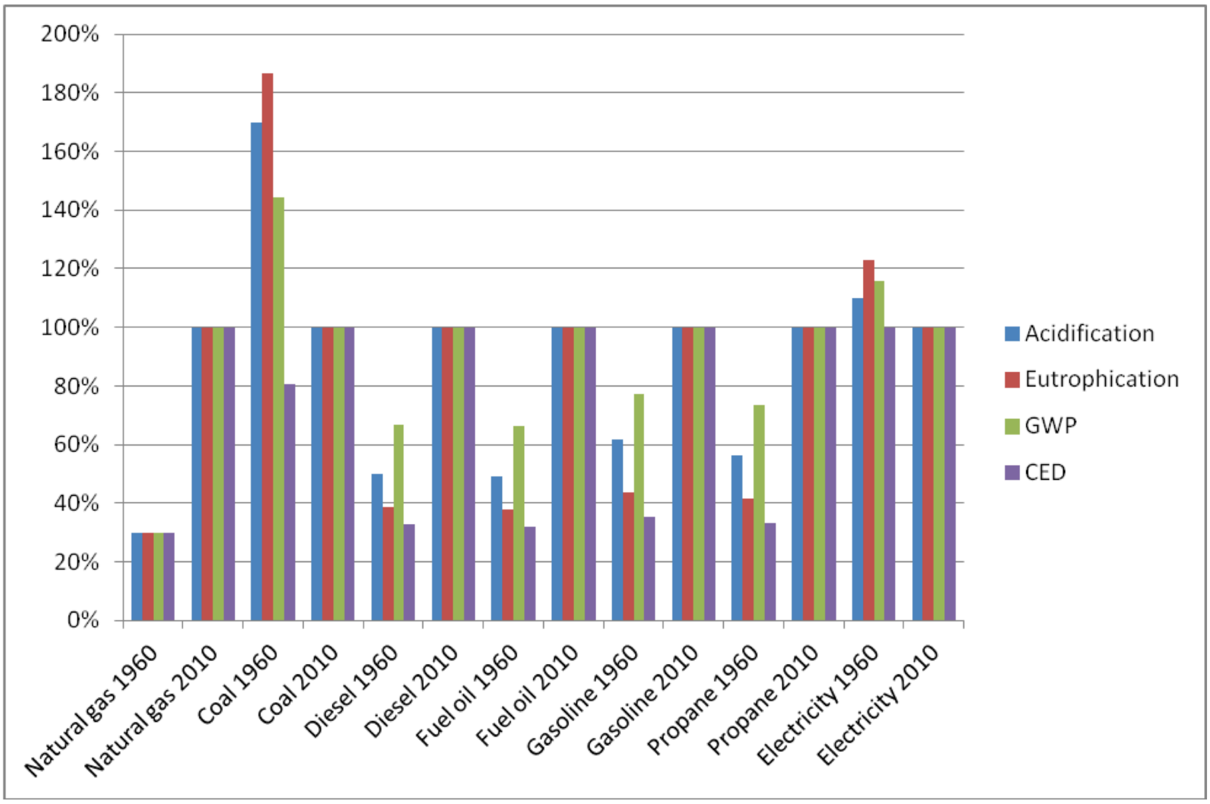

Figure 2. Life cycle impact assessment results for energy carriers in 1960 compared to 2010 (all impacts for 1960 presented as a percentage of impacts in 2010).

3.2.2 Life Cycle Impact Assessment Results for Fertilizer Inputs in 1960 versus 2010

Despite the substantial increases in the energy efficiency of ammonia production, declining EROI values for energy production effectively offset these gains. As a result, the comparative impacts of nitrogen fertilizers consumed in the US in 1960 are very similar for 2010. Impacts for phosphorus fertilizer are also similar, with the exception of considerably higher eutrophication impacts in 1960, largely due to the larger fraction of triple super phosphate in the 1960 fertilizer mix. In contrast, all impacts associated with the US potassium fertilizer mix were substantially higher in 1960 compared to 2010 due to the predominance of more energy-intensive NPK mixes in 1960 versus greater reliance on less energy-intensive potassium chloride in 2010 (Figure 3).

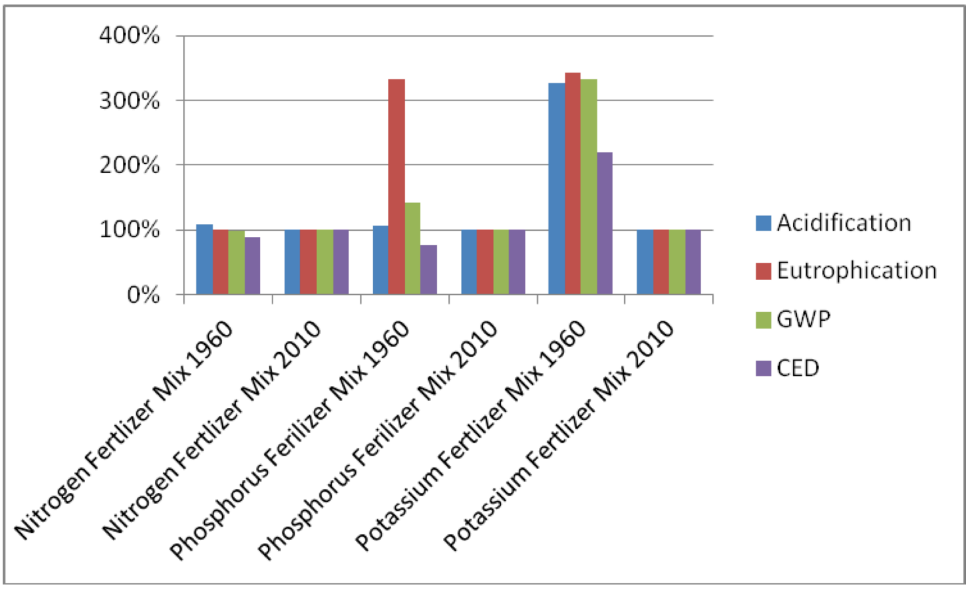

Figure 3. Life cycle impact assessment results for average US fertilizer mixes in 1960 versus 2010 (all 1960 results presented as a percentage of 2010 results for each fertilizer mix).

3.2.3 Life Cycle Impact Assessment Results for Transport Modes in 1960 versus 2010

Acidifying, eutrophying, and GHG emissions per tonne-km of freight transport were considerably higher (>250%) in 1960 compared to 2010 for both rail and ocean freight. Interestingly, the declining EROI of fossils fuels over this interval offset almost exactly the improved fuel efficiencies enjoyed by contemporary fleets, resulting in very similar cumulative energy demand. For road freight, in contrast, CED was much lower in 1960, and all other impacts very similar to those estimated for 2010. This outcome reflects the lower efficiency gains for road freight compared to rail and ocean freight for the 50-year interval considered (Figure 4).

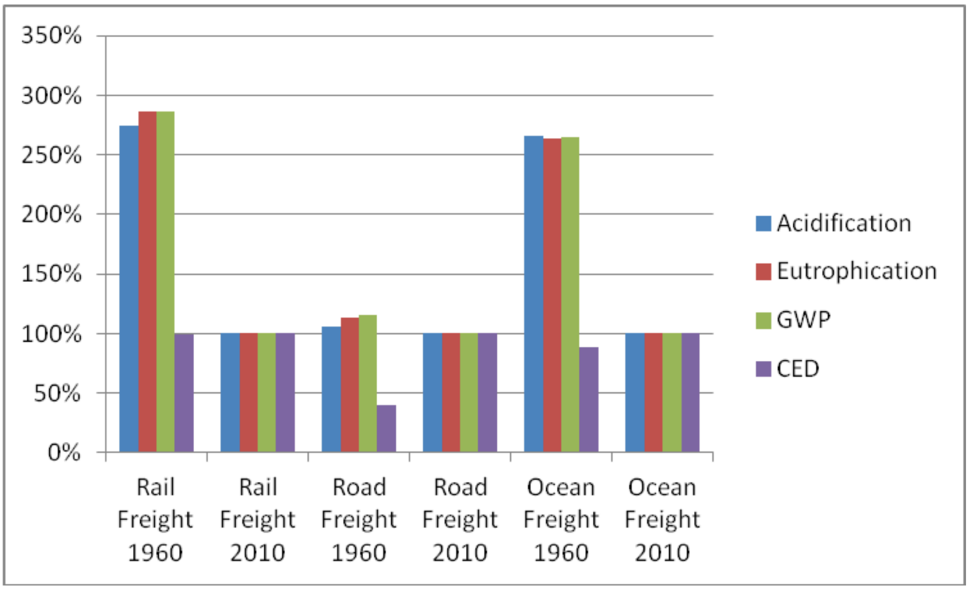

Figure 4. Life cycle impact assessment results per tonne-km for ocean, rail and road freight in 1960 vs. 2010 (all results for 1960 are presented as a percentage of impacts in 2010).

3.2.4 Life Cycle Impact Assessment Results for Feed Inputs in 1960 versus 2010

In general, the production of raw materials is the largest contributor to cradle-to-mill gate impacts for feed inputs to pullet and layer systems, although processing-related emissions are notable for some inputs such as corn dried distillers grains with solubles (CDDGS). Milling-related impacts account for a very small fraction of emissions per tonne of feed produced. Production of animal-derived feed inputs is most impactful across impact categories. This is unsurprising given the inefficiencies inherent to biological feed conversion, which effectively act as a multiplier for the impacts of producing the underpinning feed inputs, along with other inputs to animal husbandry, processing, and reduction of processing co-products into meals and fats. This is particularly true for the production of meat and bone meal and fat from ruminant sources compared to porcine and poultry sources, because feed inputs and associated emissions to produce ruminants are considerably higher.

Emission-related impacts for feed inputs produced in 1960 are almost universally higher than those in 2010. This reflects a combination of factors, including improved efficiencies of nitrogen fertilizer production, transport modes and, in particular, much-improved yields in 2010. The opposite is true for CED, however, where declining EROI effectively outweighs other efficiency gains (Table 10, Figure 5).

Table 10. Life cycle impact assessment results per tonne of feed inputs at the farm or processor gate in 1960 and 2010 (Acidification in kg SO_2-e, Eutrophication in kg PO_4-e, Global Warming Potential in kg CO_2-e, and Cumulative Energy Demand in MJ).

Feed Ingredients	Year	Acidification	Eutrophication	GWP	CED
Corn	1960	7	2	345	1380
	2010	5	1	301	1759
CDDG	1960	10	2	764	4425
	2010	7	1	719	7949
Soy Meal	1960	7	1	249	1337
	2010	4	1	227	2601
Soy Oil	1960	15	3	541	2909
	2010	9	2	493	5621
Bakery Material	1960				
	2010	8	2	551	8736
Wheat Middlings	1960	10	2	430	2364
	2010	10	2	490	4222
Alfalfa Hay	1960	2	1	101	499
	2010				
Fish Meal	1960	6	3	714	4620
	2010				
Poultry M&B	1960	191	71	6472	31165
Meal	2010	121	45	4605	42437
Porcine M&B	1960	200	74	5820	20800
Meal	2010	96	27	4318	24221
Ruminant M&B	1960	565	254	34100	59600
Meal	2010	404	185	25636	74133
Poultry Fat	1960	331	124	11210	53980
	2010	209	79	7975	73457
Porcine Fat	1960	400	149	11600	41500
	2010	193	54	8627	48306
Ruminant Fat	1960	1136	511	68468	119788
	2010	812	371	51546	148951
Salt	1960	2	0	300	2543
	2010	2	0	263	3936
Limestone	1960	0	0	47	779
	2010	0	0	43	964
Calcium	1960	39	1	1094	9328
Phosphate	2010	38	1	938	15188

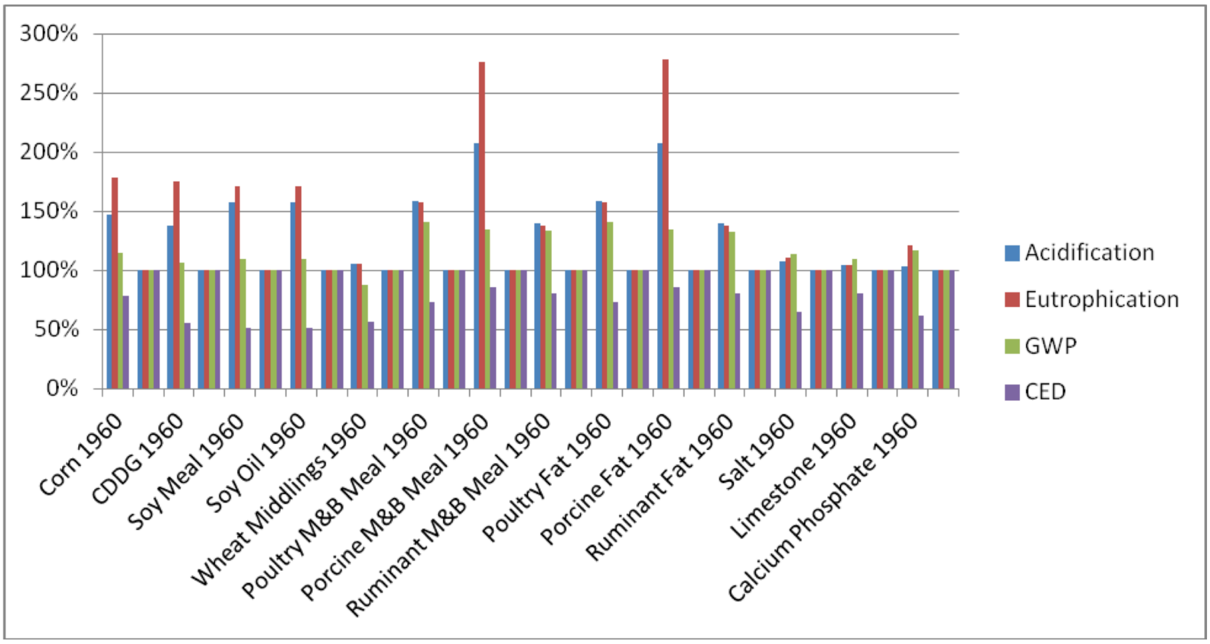

Figure 5. Life cycle impact assessment results for feed inputs to US pullet and layer systems (at the farm or processor gate) in 1960 compared to 2010 (all results for 1960 are presented as a % of impacts in 2010 for each feed input).

As a result of both, the differences in impacts attributable to feed inputs in 1960 compared to 2010, as well as changes in feed formulation over time (in particular, decreased use of animal-derived meals and oils), a similar pattern is observed for pullet and layer feeds. Averaged across emission-related impact categories, impacts for feeds in 2010 are 51% of those in 1960 for pullet feeds, and 37% for layer feeds per tonne of feed produced. In contrast, CED is 36% and 2% higher, respectively (Table 11, Figure 6).

Table 11. Life cycle impact assessment results per tonne of pullet and layer feeds produced in 1960 and 2010 (Acidification in kg SO_2-e, Eutrophication in kg PO_4-e, Global Warming Potential in kg CO_2-e, and Cumulative Energy Demand in MJ).

Feed & Year	Acidification	Eutrophication	GWP	CED
Pullet Feed 1960	18.4	6.8	1015	3139
Pullet Feed 2010	9.8	2.9	584	4267
Layer Feed 1960	34.5	13.8	1860	4560
Layer Feed 2010	12.5	4.4	782	4632

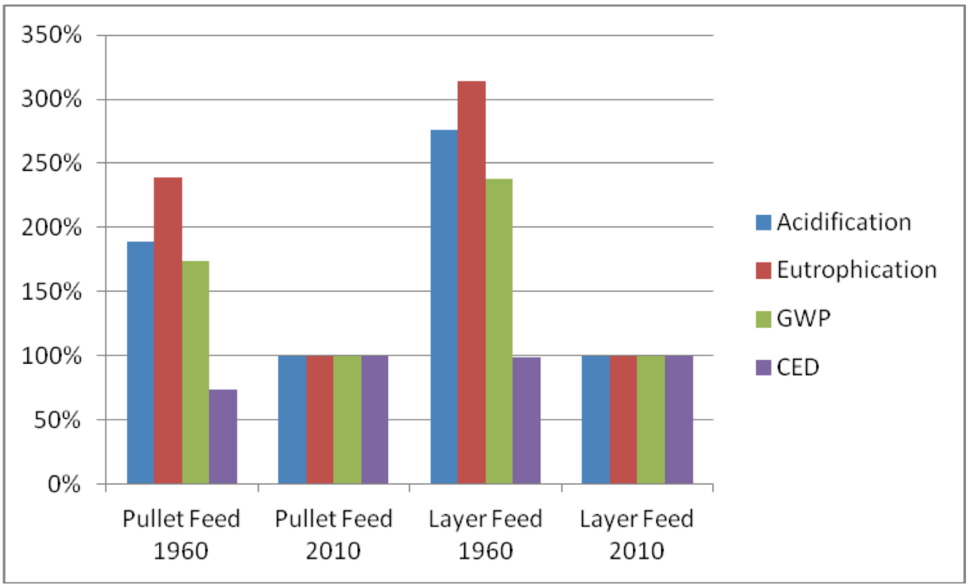

Figure 6. Life cycle impact assessment results for pullet and layer feeds in 1960 compared to 2010 (all impacts for feeds produced in 1960 are expressed as a % of impacts for feeds in 2010).

3.3 COMPARING PULLET PRODUCTION IN 1960 AND 2010

Emissions-related impacts of pullet production in both 1960 and 2010 are largely driven by two factors – feed inputs and manure management (Figures 7 and 8). For CED, direct energy inputs to pullet houses figure along side feed inputs as a major contributor. However, the relative importance of these factors differ between 1960 and 2010. In 1960, feed inputs weighed most heavily across impact categories – in particular for GWP and CED. In 2010, manure management is the most important variable for acidifying and eutrophying emissions, due to decreased emissions associated with the production of feed inputs. The importance of direct energy inputs has also increased in 2010, again due to the declining relevance of feed inputs as a result of changing feed composition (less animal-derived materials, which are particularly energy-intensive to produce).

Averaged across emissions-related impact categories, pullet production in 2010 has 44% of the impacts estimated for 1960. Cumulative energy demand is also slightly lower, at 91% (Table 12).

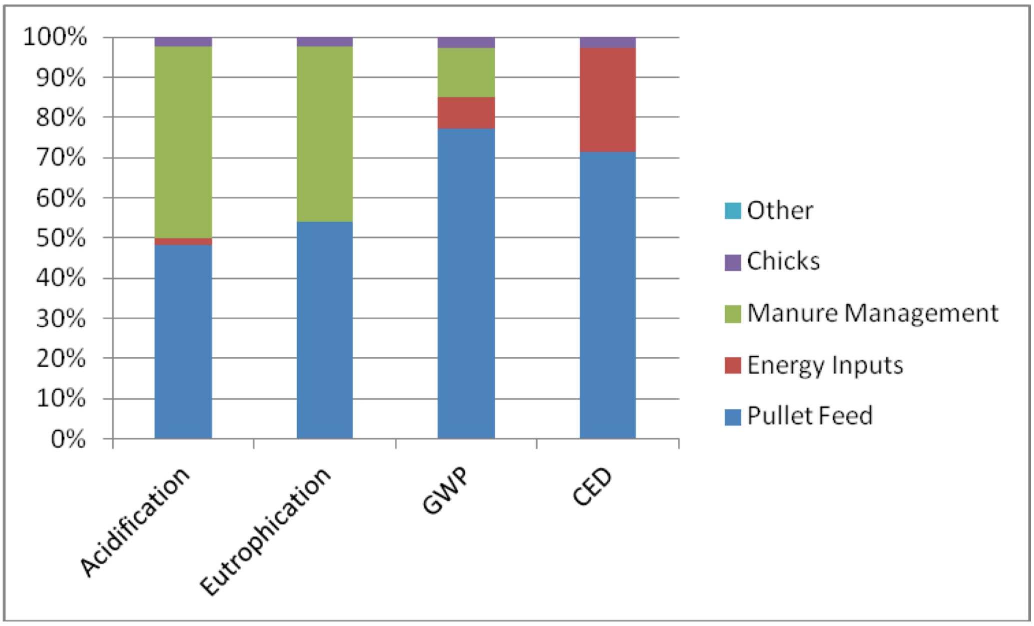

Figure 7. Contribution analysis for the life cycle impact assessment of pullets produced in the United States in 1960.

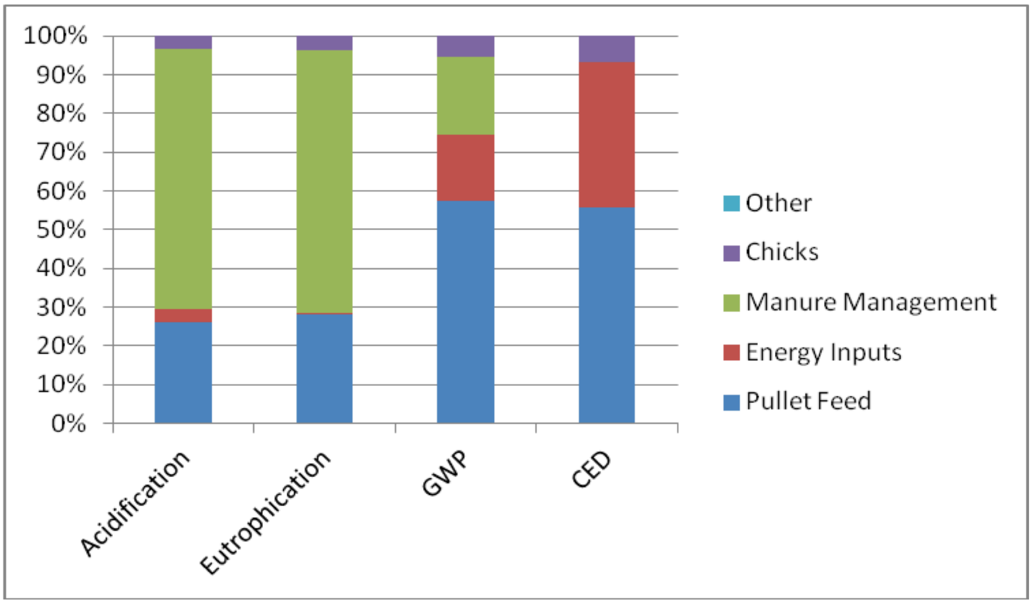

Figure 8. Contribution analysis for the life cycle impact assessment of pullets produced in the United States in 2010.

Table 12. Life cycle impacts assessment results for 1000 pullets produced in 1960 and 2010 in the United States.

Year	Acidifying Emissions (kg SO_2-e)	Eutrophying Emissions (kg PO_4-e)	GHG Emissions (kg CO_2-e)	CED (MJ)
1960	390	129	13458	45
2010	196	54	5404	41
Reduction, %	50	58	60	9

3.4 COMPARING EGG PRODUCTION IN 1960 AND 2010

The distribution of impacts for egg production is very similar to that of pullet production for both 1960 and 2010 – in particular with respect to the changing importance of feed inputs and manure management. In 2010, manure management replaces feed inputs as the largest source of acidifying and eutrophying emissions (despite substantially lower losses of N and P per kg of eggs produced), whereas feed remains the dominant (although smaller) contributor to both GWP and CED. These changes are reflective of both changing feed composition and improved feed conversion efficiencies. Poultry production contributes roughly 10% to emissions-related impacts in both 1960 and 2010, and slightly more for CED (Figures 9-10). In general, direct energy inputs are of lesser importance. Overall, emissions-related impacts of egg production in 2010 are estimated to be 31% of those of 1960, while CED is 69% (Table 13).

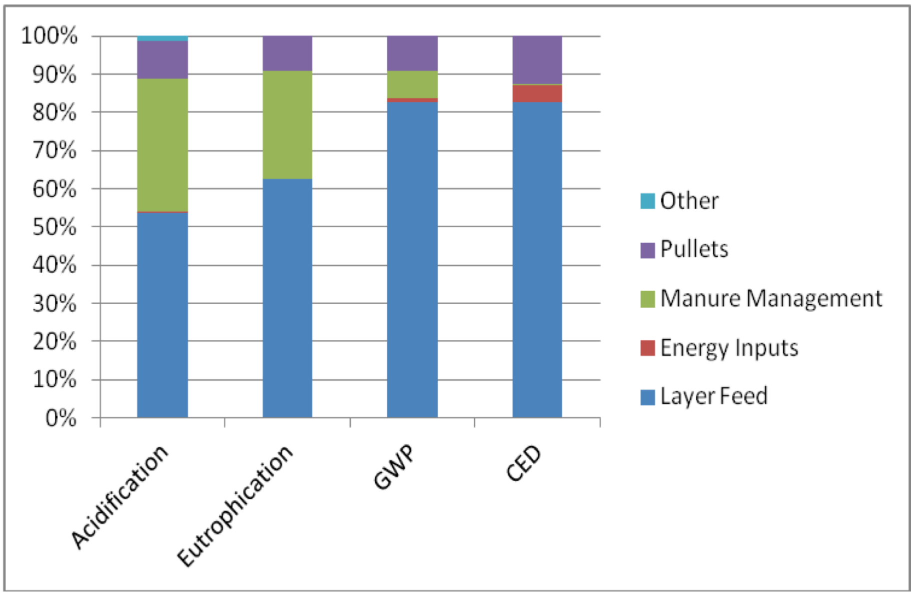

Figure 9. Contribution analysis for the life cycle impact assessment of eggs produced in the United States in 1960.

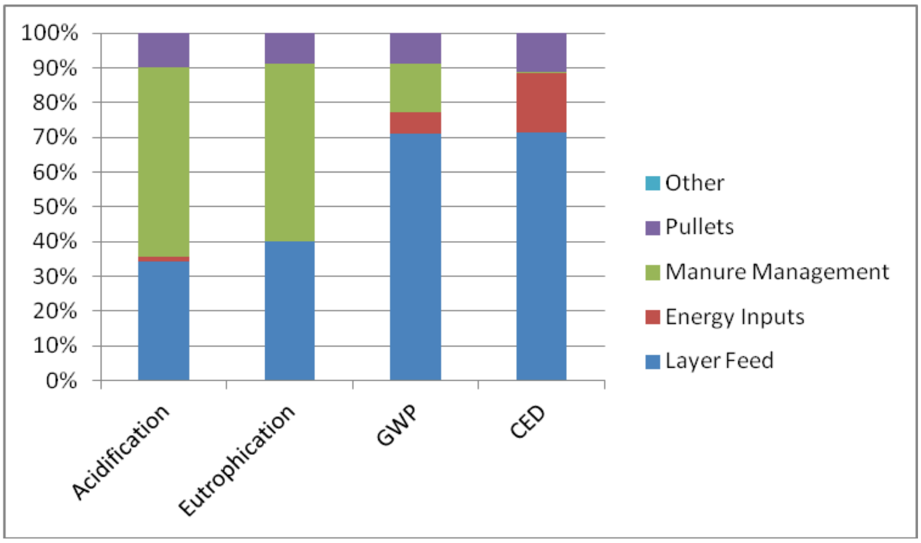

Figure 10. Contribution analysis for the life cycle impact assessment of eggs produced in the United States in 2010.

Table 13. Life cycle impacts assessment results for one kg of eggs produced in 1960 and 2010 in the United States, and % reduction in impacts over the 50-year interval considered (Acidifying emissions in g SO_2-e, Eutrophying emissions in g PO_4-e, GHG emissions in g CO_2-e, and Cumulative Energy Demand in MJ).

Year	**Acidifying Emissions**	**Eutrophying Emissions**	**GHG Emissions**	**CED**
1960	200	70	7230	17.7
2010	70	20	2080	12.3
Reduction, %	65	71	71	31

3.5 ANALYSIS ON DRIVERS OF OBSERVED DIFFERENCES IN IMPACTS BETWEEN 1960 AND 2010

Averaged across impact categories, impacts for egg production in 2010 were 60% lower than that of 1960 (Table 14, Figure 12). By applying 2010 background system sub-models in the 1960 egg production model, we estimate that 27-30% of the observed differences in acidification, eutrophication, and GWP are attributable to changes in the efficiencies of background systems such as fertilizer and feed input production, and transport modes. These outweighed the declining energy return on energy invested (EROI) ratio for primary energy carriers in these impact categories. For CED, however, applying 2010 energy carriers to the 1960 model resulted in 35% higher impacts in this category (Table 14, Figure 12).

Using both 2010 background system models and feed composition in the 1960 egg production model, we further estimate that changes in feed composition over time accounted for 30% of the observed decline in acidification potential for egg production in 1960 versus 2010, 34% for eutrophication potential, and 44% for GWP. On balance, we hence estimate that changes in animal performance due to improved husbandry over the 50-year interval (e.g., improved feed conversion, lower mortality rates, etc.) were responsible for 43% of the observed decline in acidification potential, 35% for eutrophication potential, and 28% for GWP for egg production in 2010 compared to 1960. Despite declining EROI, CED in 2010 was only 30% that of 1960, due to a combination of changing feed composition and improved animal husbandry practices (Table 14, Figure 12).

Table 14. Proportion (in %) of changes in environmental footprint of egg production in 2010 compared to 1960 attributable to changes in background systems, changes in feed composition, or changes in animal performance due to improved husbandry and genetics.

	Acidifying Emissions	Eutrophying Emissions	GHG Emissions	CED
% change due to changes in background systems	27	30	28	-116
% change due to changes in feed composition	30	34	44	93
% change due to changes in animal performance	43	35	28	123

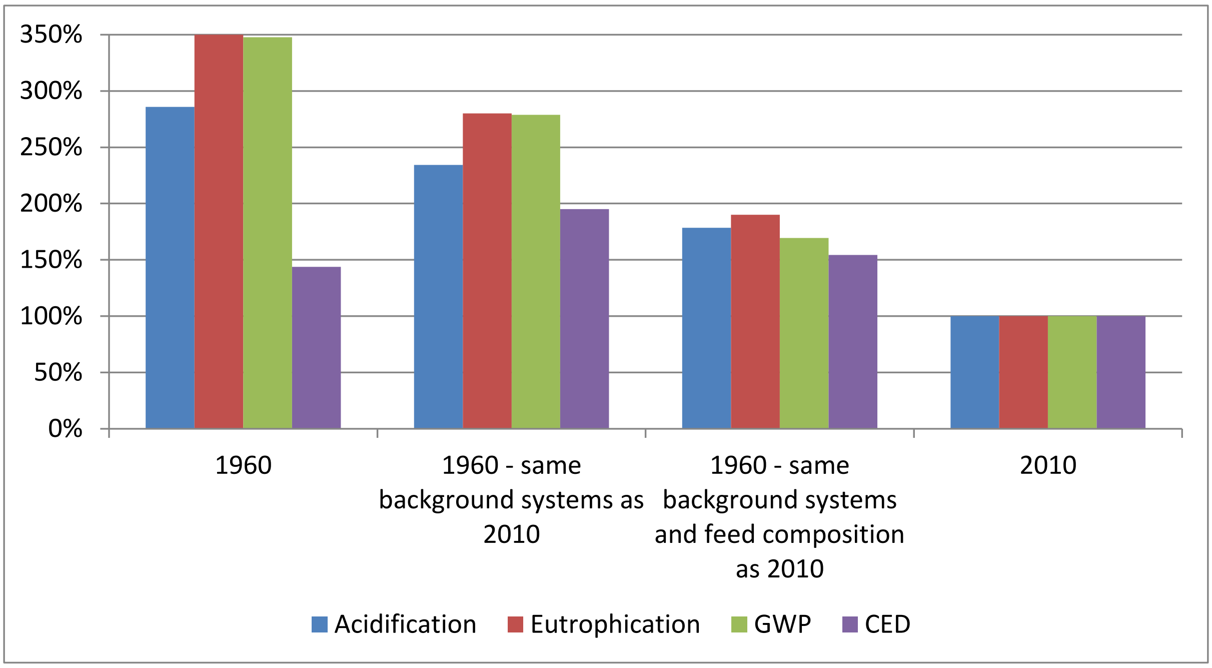

Figure 11. Scenario analyses to determine the relative contributions of assumed differences in background systems, feed composition, and animal husbandry performance to the estimated impacts for US egg production in 1960 compared to 2010.

3.6 COMPARISON WITH OTHER STUDIES

A limited number of time series analyses of the environmental impacts of animal husbandry are available. Capper et al. (2009) and Capper (2011) evaluated changes in the environmental performance of beef production in 1977 versus 2007, and dairy production in 1944 versus 2007. Considerable gains in the resources use efficiency (69.9% of animals, 81.4% of feedstuffs, 87.9% of the water, and 67.0% of the land required) to produce 1 billion kg of beef in 2007 compared to 1977, and commensurate decreases (16.3%) in associated GHG emissions, were documented. Similar gains in resource efficiency were estimated for dairy (21% of animals, 23% of feedstuffs, 35% of the water, and only 10% of the land required to produce 1 billion kg of milk in 2007 compared to 1944), while GHG emissions were 37% of 1944 levels. It should be noted that these studies (nor those discussed below) did not take into account changes in the resource efficiencies of background systems, hence are likely quite conservative. Our estimates of the scale of resource efficiencies and emissions reductions for egg production between 1960 and 2010 are, nonetheless, of a comparable magnitude. In Canada, Verge et al. (2009) calculated direct GHG emissions from layer facilities along with crops used to produce layer feeds in 1981 compared to 2006. Indirect supply chain emissions were not considered, hence the study results are not comparable with those presented in the current analysis. Nonetheless, it is interesting to note that these authors found that the GHG intensity of egg production decreased from 1.9 kg of CO_2-e/dozen eggs in 1981 to 1.76 kg of CO_2-e/dozen eggs in 2006, or approximately 7% reduction over this 25-year interval. Cederberg et al. (2009) compared the GHG emissions from Swedish livestock production in 1990 and 2005 for pork, poultry meat, beef, milk and eggs. They found that the carbon footprint of pork production decreased from 4 to 3.4 kg CO_2-e/kg over this 15-year interval, whereas emissions for poultry meat decreased from 2.5 to 1.9 kg CO_2-e/kg, milk from 1.27 to 1 kg CO_2-e/kg, and emissions from beef increased from 18 to 19.8 kg CO_2-e/kg. Emissions from egg production remained unchanged at 1.4 kg CO_2-e/kg over this interval. This latter finding is in large part attributable to two factors. First is the phasing out of animal by-products in feeds as a result of the BSE issues and second is the use of economic allocation. Here, despite efficiency gains in the sector, the allocation strategy resulted in a study outcome suggesting no net gains in environmental performance.

To date, no other estimates for the life cycle impacts of contemporary US egg production are available. Pelletier et al. (in press) previously modeled egg production in Iowa using the same modeling approach as applied in this analysis. In the Iowa study the authors didn't identify the sources (ruminant, swine or poultry) of by-products and they estimated the GHG emissions ranged from 2.0 kg of CO_2-e per kg of eggs (assuming 100% of the animal by-product being of poultry origin) to 5.0 kg of CO_2-e per kg of eggs (assuming 100% of the animal by-products being of ruminant origin).

Several studies are available, however, that report environmental performance for egg production supply chains in other countries. Although direct comparisons between studies are problematic due to frequent differences in modeling parameters (e.g, system boundaries for the studies, data sources, allocation rules, etc.), it is nonetheless interesting to consider the range of reported impacts relative to those of the current study.

In broad strokes, the distribution of impacts along contemporary US egg supply chains seems to be in general agreement with similar, previously reported LCA research of intensive, cage egg production systems elsewhere (Mollenhorst et al. 2006; Cederberg et al. 2009; Verge et al. 2009; Wiedemann and McGahan 2011; Leinonen et al. 2012). In a study examining the social, economic, and ecological dimensions of egg production by housing system in the Netherlands, Mollenhorst et al. (2006) used LCA as a basis for comparing performance in the environmental domain. Conventional cage production was found to perform better according to the environmental LCA variables considered, but the aviary system performed better according to the economic and animal welfare measures employed. In Australia, Wiedemann and McGahan (2011) used a life cycle approach to evaluate GHG emissions, energy and water use in egg production by housing system. Here, activity data were collected from four farms in eastern Australia. Cage systems were found to outperform free-range systems. Estimated impacts overall were low compared to results from most European studies. More recently, Leinonen et al. (2012) used top-down estimates of average UK production conditions in a standard, environmental LCA approach to characterize environmental performance for egg production in cage, barn, free-range, and organic systems. They reported highest impacts for organic production and lowest for cage production, largely due to differences in productivity (i.e., higher feed consumption and number of birds required per unit of egg production in the organic system). Feed production supply chains were the dominant contributor to GHG emissions and energy use (54-75% of the primary energy use and 64-72% of the GWP). Similar to our study, energy use in housing systems was the second most important factor for the overall energy intensity of egg production. Manure management contributed most to acidifying and eutrophying emissions.

Table 15 provides a summary of results from these studies. Where estimated impacts in these other studies are low compared to those of the present analysis, this is typically either because animal by-products are not allowed for use in animal feeds in the countries of concern (for example, the Swedish study by Cederberg et al. 2009 for the 2005 system modeled), or because they were not included in the modeled feeds at all, whether or not they are actually used (for example, the Australian study by Wiedemann and McGahan). In the latter study, the authors also point towards the low input nature of Australian grain production (compared to European norms) as an important factor influencing their reported outcomes. Considering the study of egg production in Sweden in 1995 compared to 2005 (Cederberg et al. 2009), the reduction in use of animal by-product due to legislative changes in response to the BSE concerns over the interval considered actually *negatively* impacted performance in 2005 due to the use of economic allocation in this study. This is contrary to the results of the current analysis, which shows an improved environmental performance over time by reducing the amount of animal by-products used in pullet and layer feeds. In light of the resource and emissions intensity of producing livestock (along with the livestock processing co-products used in animal feeds), we suggest that our analytical approach better reflects the actual environmental costs of producing feed inputs for egg production, regardless of the economic value of such materials.

To put the GHG intensity of contemporary US egg production in perspective, we provide the following comparison: using the same methods, Pelletier et al. (2010a) recently estimated the GHG emissions per kg of pork production in this region at 3 kg CO2-e per kg live weight produced. For conventional, feedlot beef production, estimated emissions were 14.5 kg CO2-e per kg live weight produced (Pelletier et al. 2010b). Adapting the inventory data and methods of an earlier study of US broiler production (Pelletier 2008) for methodological consistency with these analyses provides an estimate of 1.7 kg CO2-e per kg live weight produced. Here, we estimated a GHG intensity of 2.1 kg CO2-e per kg of eggs produced in the continental United States, compared to 7.2 kg CO2-e per kg of eggs produced in 1960.

Making a similar comparison on the basis of protein, the GHG intensity of US egg protein production (raw, from whole eggs) is 19.1 CO2-e/kg of protein compared to 11.5 kg CO2-e/kg of broiler protein, 17.6 kg CO2-e/ kg for pig protein, and 78.4 kg CO2-e/kg of beef protein.

Table 15. Reported life cycle impacts per kg of eggs produced in different countries.

Study	**Energy Use (MJ)**	**GHG Emissions (kg CO_2-e)**	**Acidifying Emissions (g SO_2-e)**	**Eutrophying Emissions (g PO_4-e)**
US average (this study)	12.3	2.1	70	**20**
UK[1]	16.8	2.9	53	77
Netherlands[2]		3.9	32	25
Sweden[3]	-	1.4	-	-
Canada[4]	-	2.5	-	-
Australia[5]	-	1.4	-	-

(1) Leinonen et al. (2012)
(2) Mollenhorst et al. (2006)
(3) Cederberg et al. (2009)
(4) Verge et al. 2009
(5) Wiedemann and McGahan (2011)

Clearly, the US egg sector has made significant strides in improving resource use efficiency and reducing environmental impacts per unit production since the 1960's. It is also interesting, however, to consider the extent to which such improvements mitigate impacts when considered in terms of changes in the scale of production. The total US table egg production in 1960 was 59.8 billion eggs compared to 77.8 billion in 2010 (USDA NASS) – an increase of roughly 30%. Effectively, this means that, despite the substantial increase in production volumes, absolute cumulative energy demand in the US egg industry nonetheless decreased almost 10%, while GHG emissions declined by 63%, acidifying emissions by 54%, and eutrophying emissions by 63%.

4. CONCLUSIONS AND RECOMMENDATIONS

Our analysis of the distribution and magnitude of life cycle impacts for egg production in the United States in 1960 compared to 2010 provides a clear indication of the scale of environmental performance gains, both per unit production and in aggregate, achieved by the industry over the past 50 years, as well as insights into the primary contributing factors. Several key insights emerge.

From a supply chain management perspective, the key leverage point for environmental performance improvements in egg production has been and will continue to be efforts to maximize feed use efficiencies, because feed production accounts for the largest share of impacts in egg production both in 1960 and at present. The feed conversion ratio for egg production improved from 3.44 kg/kg in 1960 to 1.98 kg/kg – a gain of 42%. Nonetheless, achieving feed use efficiencies comparable to the best performing contemporary facilities (the range reported by survey respondents was 1.76-2.32 kg/kg) industry-wide would do much to further reduce aggregate impacts.

Changing feed composition has also played an important role in reducing impacts – in particular, both reduction in the total amount of animal-derived materials used as well as increased use of porcine and poultry materials in place of ruminant materials. The concept of least-environmental cost feed sourcing is therefore of particular relevance for additional targeted performance improvements for this industry. It is recommended that similar biophysical accounting methods to those applied in the current study be used to model potential alternative feed input supply chains to ensure methodological consistency and comparability with the present analysis.

Managing feed supply chains for environmental performance must also take into account nitrogen use efficiencies. N losses from poultry manure are the second largest contributor to acidifying and eutrophying emissions, as well as a non-trivial contributor to GHG emissions in both pullet and layer facilities. Moreover, upstream impacts of N fertilizer production and use are a primary determinant of feed input-related impacts. Feed formulation, breeding, and selecting manure management strategies for optimal N use efficiencies are therefore powerful tools in supply chain environmental management. Here, we modeled N losses using standard IPCC protocols. Given the margin of error associated with manure N sampling, we recommend using this IPCC-based modeling approach. This will also maximize inter- and intra-company and product comparability. However, we also suggest continued efforts to improve and standardize company-level manure-N sampling accuracy, in order to allow for differentiation between facilities and production strategies looking forward.

Overall, our analysis provides compelling evidence that considerable strides in resource use efficiency and animal husbandry performance in the US egg sector over the past 50 years have much reduced both the relative and absolute impacts of US egg production. Also apparent, however, is that there remains substantial scope for continued improvement. Moreover, in light of continued declines in EROI for energy carriers consumed in egg supply chains, continuous improvement will likely be necessary simply to maintain the current status quo environmental footprint of the US egg sector. The benchmarks reported here, as well as the reported ranges for resource use and production efficiencies in what are, ostensibly, otherwise similar production facilities, provide an excellent reference point for industry-led initiatives for further improving the environmental performance of US egg production.

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