Mapping the Influence of Food Waste in Food Packaging Environmental Performance Assessments

Martin C. Heller,[1] *Susan E. M. Selke*,[2] *and Gregory A. Keoleian*[1]

[1]*Center for Sustainable Systems, School of Natural Resources and Environment, University of Michigan, Ann Arbor, MI, USA*
[2]*School of Packaging and Center for Packaging Innovation and Sustainability, Michigan State University, East Lansing, MI, USA*

Keywords:

food packaging
food waste
greenhouse gas (GHG) emissions
industrial ecology
life cycle assessment (LCA)
life cycle energy analysis

Supporting information is linked to this article on the *JIE* website

Summary

Scrutiny of food packaging environmental impacts has led to a variety of sustainability directives, but has largely focused on the direct impacts of materials. A growing awareness of the impacts of food waste warrants a recalibration of packaging environmental assessment to include the *indirect* effects due to influences on food waste. In this study, we model 13 food products and their typical packaging formats through a consistent life cycle assessment framework in order to demonstrate the effect of food waste on overall system greenhouse gas (GHG) emissions and cumulative energy demand (CED). Starting with food waste rate estimates from the U.S. Department of Agriculture, we calculate the effect on GHG emissions and CED of a hypothetical 10% decrease in food waste rate. This defines a limit for increases in packaging impacts from innovative packaging solutions that will still lead to net system environmental benefits. The ratio of food production to packaging production environmental impact provides a guide to predicting food waste effects on system performance. Based on a survey of the food LCA literature, this ratio for GHG emissions ranges from 0.06 (wine example) to 780 (beef example). High ratios with foods such as cereals, dairy, seafood, and meats suggest greater opportunity for net impact reductions through packaging-based food waste reduction innovations. While this study is not intended to provide definitive LCAs for the product/package systems modeled, it does illustrate both the importance of considering food waste when comparing packaging alternatives, and the potential for using packaging to reduce overall system impacts by reducing food waste.

Introduction

While the modern food industry has concerned itself with maintaining food safety and quality, the moral imperative of feeding a rapidly growing population, combined with a maturing recognition of the biophysical planetary limits within which this food must be supplied, has brought acute focus to the problem of food waste. The Food and Agriculture Organization of the United Nations (FAO) estimates that one third of food produced for human consumption is lost or wasted globally (Gustavsson et al. 2011). Food produced and not eaten has an annual carbon footprint of 3.3 gigatonnes of carbon dioxide

Conflict of Interest: The authors declare no conflict of interest.

Address correspondence to: Martin C. Heller, University of Michigan, 3012 Dana Building, 440 Church Street, Ann Arbor, MI 48109-1041, USA. *Email*: mcheller@umich.edu; *Web*: http://css.umich.edu/person/martin-c-heller

DOI: 10.1111/jiec.12743

Editor managing review: Robert Anex

Volume 23, Number 2

equivalents (CO_2-eq.) (if it were a country, it would be the third top emitter after the United States and China) and occupies 30% of the world's agricultural land area (FAO 2013).

In response to these staggering losses, The United Nations Global Sustainable Development Goals (SDG 12.3) include a 50% reduction in per-capita global food waste at the retail and consumer levels by 2030 (United Nations 2015). The U.S. Department of Agriculture (USDA) and U.S. Environmental Protection Agency (US EPA) also announced in 2015 the first U.S. food waste reduction goal, calling for a 50% reduction by 2030 (USDA 2015). An estimated 70 million metric tonnes (MMT) of edible food is lost annually in the United States, with nearly 60% of this occurring at the consumer level (Dou et al. 2016). Greenhouse gas (GHG) emissions associated with production of this food loss are estimated at 1.4 kilograms (kg) CO_2-eq. capita^{-1} day^{-1} (160 MMT CO_2-eq. in annual total), increasing the carbon footprint of the average U.S. diet by 39% (Heller and Keoleian 2015). Meeting ambitious waste reduction goals will require concerted effort from stakeholders throughout the food value chain.

Sustainability efforts aimed at reducing the environmental impact of packaging often overlook the primary role of food packaging: protecting and preserving both perishable and shelf-stable foods. Environmental concerns about packaging tend to focus on the direct environmental impacts of packaging material production and packaging end-of-life (EoL), despite the indication that efforts to reduce indirect impacts of food waste often far outweigh options to reduce direct impacts (Russell 2014; Wikström et al. 2016; Silvenius et al. 2013; Williams and Wikström 2011; Wikstrom et al. 2014). A recent collaborative effort in the United States between business, nonprofit, foundation, and government leaders reports that packaging adjustments alone have the potential to divert 189,000 metric tonnes of food waste annually in the United States, with an economic value of US$715 million; active or intelligent packaging aimed at slowing spoilage offers an additional potential 65,000 metric tonnes of food waste diverted (ReFED 2016).

Life cycle assessment (LCA) is a tool used to account for the emissions and resource use throughout a product's life cycle, including raw material acquisition production, distribution, use, and disposal, and assigns these emission and resource flows to prospective environmental impacts (ISO 2006a). LCA applied to agricultural and food product systems presents a unique set of challenges (Roy et al. 2012; Andersson et al. 1994; Schau and Fet 2008). As these have been addressed over the past decade and a half, there have been exponential increases in the number of reported food LCA studies (Heller et al. 2013).

LCA of food packaging dates back to the earliest applications of the LCA method (Guinee et al. 2011). Yet, limited attention has been given to the balancing act that arises between the environmental impact of producing and disposing of the packaging itself and its ability to moderate food waste—and associated environmental impact—along the food value chain. Wikström and Williams have made significant literature contributions aimed at raising awareness of the importance of considering food waste in food packaging design and sustainability (Williams et al. 2008; Wikstrom and Williams 2010; Williams and Wikström 2011; Williams et al. 2012; Wikstrom et al. 2014; Wikström et al. 2016). They have mathematically described the relationships between environmental impact of food waste and food packaging within a life cycle perspective (Wikstrom and Williams 2010), and established the need to utilize a functional unit based on the food *eaten* in order to account for consumer-level food losses. These authors and others have demonstrated, through specific case studies, the importance of including the environmental impact of wasted food when evaluating packaging systems. As an example, the GHG emissions of bread packaging could be doubled without increasing overall climate impact if the packaging change led to a bread waste reduction of 5% (Williams and Wikström 2011). Not including food waste may lead to contradictory results, favoring larger packaging for geometrical reasons, or less packaging material per unit of food product. In addition, such studies have established the importance of the ratio between the environmental impact of the specific food item and its packaging as a predictive parameter of food waste effects.

The goal of this paper is to consider a large number of food items and their typical packaging configurations using a consistent LCA model in order to map the potential influence of food waste effects on environmental performance. The intention is *not* to provide a definitive impact assessment of the cases studied, but instead to use the best available data to demonstrate the need for consideration of food waste in environmental assessments of food packaging. We expect that this mapping exercise will offer packaging design engineers preliminary guidance on the significance of food waste in optimizing the environmental performance of packaging. We also aim to raise general awareness to the potential role that packaging can play, when properly designed, in reducing food waste and, in turn, the environmental impacts of our food system.

Methods

To orient this research in the existing food LCA literature, we conducted a thorough literature review, extracting GHG emissions and energy demand data across life cycle stages of various foods (see supporting information S1 available on the Journal's website for details of literature review). This literature review provides a basis for a broad exploration of the food to packaging (FTP) environmental impact relationship, defined here as (equation 1):

$$FTP_E = \left[E\left(\frac{\textit{agricultural (farm gate) production}}{\textit{kg food}}\right) + E\left(\frac{\textit{food processing}}{\textit{kg food}}\right)\right] / E\left(\frac{\textit{packaging materials}}{\textit{kg food}}\right) \quad (1)$$

where: E = environmental impact indicator of interest (e.g., GHG emissions, cumulative energy demand [CED]).

Table 1 Foods, primary packaging, and baseline food waste rates considered in this study

Food	*Primary package*	*2013 USDA LAFA food waste rates*[a]	
		Retail	*Consumer*
Spinach	PET clam, 100% virgin PET	14.4%	9%
Spinach	PET clam, 100% PCR PET	14.4%	9%
Ready-to-eat lettuce	LDPE/PP bag	13.9%	24%
NFC OJ	1 L PET, 100% virgin PET	6%	10%
NFC OJ	1 L PET, 100% PCR PET	6%	10%
NFC OJ	1 gal (3.8 L) HDPE, 100% virgin HDPE	6%	10%
NFC OJ	1 gal (3.8 L) HDPE, 100% PCR HDPE	6%	10%
Chopped tomatoes	Steel can	6%	28%
Mushrooms	8 oz (0.24 L) PET tray 100% virgin PET	12.7%	21%
Mushrooms	8 oz (0.24 L) PET tray 100% PCR PET	12.7%	21%
Potatoes	5 lb (2.27 kg) LDPE bag	6.5%	16%
Eggs	PET carton, 100% virgin PET	9%	23%
Eggs	PET carton, 100% PCR PET	9%	23%
Eggs	Paperboard carton	9%	23%
Potato chips	PP bag	6%	4%
Milk	1 gal (3.8 L) HDPE, 100% virgin HDPE	12%	20%
Milk	1 gal (3.8 L) HDPE, 100% PCR HDPE	12%	20%
Milk	1/2 gal (1.9 L) paperboard	12%	20%
Ground turkey	3 lb. (1.36 kg) MAP	3.5%	(35 – 23) = 12%[b]
Ground turkey	3 lb. (1.36 kg) chub	3.5%	(35 – 23) = 12%[b]
Pork	PS tray w overwrap	4.4%	(29 – 24) = 5%[b]
Cheese	PET bag, 100% virgin PET	6%	24%[c]
Cheese	PET bag, 100% PCR PET	6%	24%[c]
Beef	PS tray w LDPE overwrap	4.3%	4%[b]

[a]The USDA reports these as food loss rates, but after correcting for cooking losses, we consider them equivalent to food waste rates. In some cases (NFC OJ, chopped tomatoes, ready-to-eat lettuce, and ground turkey), the waste rates are from a more generic commodity category (orange juice, canned tomatoes, fresh romaine and leaf lettuce, and turkey).
[b]Consumer loss rates modified to account for cooking losses. See above text for description.
[c]Average of all cheeses.
NFC OJ = not-from-concentrate orange juice; PCR = postconsumer recycled.

A literature-based exploration of FTP_{GHGE} is presented in the first *Results* section.

We then developed an LCA model aimed at assessing the system-level balance in environmental impact resulting from food waste/food packaging interplay. The following sections detail the development of that model and the data used in evaluating a range of food/packaging configurations.

Food/Packaging Selection and Food Waste Rates

The foods and packaging configurations under consideration in this mapping exercise are shown in table 1, along with the assumed baseline retail- and consumer-level waste rates, taken from the 2013 USDA Loss Adjusted Food Availability (LAFA) data set (USDA ERS 2013). This selection of foods was chosen by first considering the foods with the largest flows (kg capita^{-1} yr^{-1}) at retail in each food category of the LAFA data set, and then adjusting to provide greater diversity. Primary packaging configurations were chosen (by expert opinion, as no known data exist) to represent the most popular options available in the U.S. marketplace. It is important to note that the USDA LAFA database reports food *loss* rates (losses as a fraction of the food available at each stage), often based on the differences between per-capita availability and survey-based consumption of specific foods. These losses include losses due to cooking that are not differentiated from consumer-level spoilage or plate waste. To account for this in meats, which are expected to be most affected by cooking losses, we considered typical cooking losses as reported by the USDA (Showell et al. 2012). The reported cooking losses (100 – cooking yield %) vary greatly by meat cut and cooking method, but averaging over entries resulted in 23% for turkey, 24% for pork, and 26% for beef. These cooking losses are then subtracted from consumer loss rates from LAFA to provide an estimate of spoilage and plate waste for the meats. However, LAFA reports a consumer loss rate for beef of 20%, lower than many reported cooking loss rates; we therefore assume a consumer waste rate of 4% for beef.

Functional Unit

The functional unit forms the comparative basis of LCA studies and the denominator of presented results, and therefore can influence conclusions drawn from study results. Given the

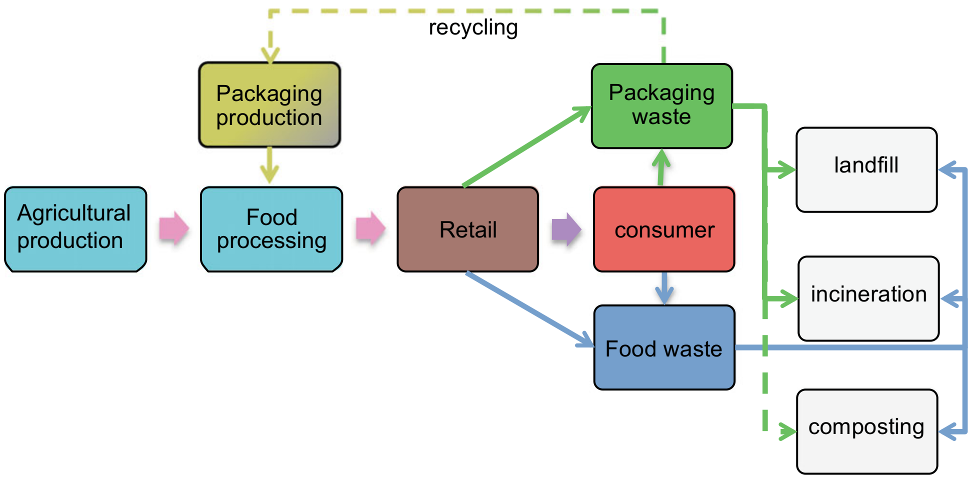

Figure 1 System diagram indicating the life cycle stages to be included in this study. Thick arrows represent tages where transport is included. Colors correspond to those in figure 3 and figure S1 in supporting information S1 on the Web.

focus on food waste in this project, the functional unit should reflect food actually consumed, therefore accounting for waste at all stages. Throughout this study, a functional unit of 1 kg of food consumed is maintained.

System Boundaries

The generic system diagram in figure 1 outlines the stages and processes included in this study. Given the intended focus on packaging trade-offs, food losses/waste at the agricultural production and primary food processing stages are not explicitly considered. The study instead focuses on food loss/waste during retail and consumption stages. As shown in figure 1, the environmental impacts from final disposal of food waste are included, as are the impacts of recycling and/or disposing of packaging waste. Transportation is accounted for between major stages, although generalized assumptions have been made to reasonably represent U.S. national average transportation distances. Refrigeration is included in distribution-, retail-, and consumer-level storage for all foods except potatoes, potato chips, and chopped (canned) tomatoes.

Life Cycle Inventory and Data Sources

In this section, we describe generic modeling and inventory approaches, as well as data sources that are common among case studies. Additional parameters and data sources unique to individual cases are detailed in supporting information S2 on the Web.

Agricultural Production and Food Processing

GHG emissions and CED of food agricultural production and food processing are drawn primarily from existing LCA literature, as detailed in table 2. Table 2 also indicates the boundary condition for which we extracted data within each study. In all cases, contributions from packaging within this literature were excluded from the extracted data and modeled independently as described below. We acknowledge that, in most cases, the literature studies are not representative of U.S. production and therefore serve as a proxy for our cases. This is appropriate given that the intention of the paper is to demonstrate the importance of food waste rather than provide a definitive assessment of the cases considered.

Packaging Production

Inventory data for the production of packaging materials as well as the transformation of materials into packaging forms were taken from the ecoinvent 3.1 database. We include both primary packaging (in direct contact with food product) and the tertiary packaging used in transport and distribution (typically not seen by customers). Secondary packaging, used to aggregate individual packages for retail display or multipack sales, is not included in the configurations studied here. Preference was given to U.S. Life Cycle Inventory Database (USLCI) data sets, where available. Specific processes, the data set origin, and impact factors are shown in table S1 in supporting information S1 on the Web. Note that transport of packaging materials is not included in our assessment.

Gases used in Modified Atmosphere Packaging (MAP) were modeled using ecoinvent processes for liquefied oxygen and liquefied carbon dioxide, applying appropriate densities and expansion ratios, as purified and pressurized gases were not available. While liquefied gases are likely not the source for MAP applications, the impacts based on this modeling approach are negligible, and nonliquefied gas sources are anticipated to have even smaller impacts.

Transport: Processor to Retail

Transportation from processing to retail distribution was modeled using a generic freight trucking process from ecoinvent 3.1, which is based on a tonne-km (kilometer) unit. Impact factors for transportation processes are shown in table S2 in supporting information S1 on the Web. Since many fresh products require refrigerated trucking (and ecoinvent 3.1 does not

Table 2 Values and sources for agricultural production and processing of foods evaluated in this study

	GHG emissions		Nonrenewable CED	
Food	*(kg CO_2eq./kg)*	*Source*	*(MJ/kg)*	*Source*
Spinach	0.18	Average of "Spinach, at farm" Agrifootprint processes for Netherlands and Belgium (Blonk Consultants 2015)	0.66	Average of "Spinach, at farm" Agrifootprint processes for Netherlands and Belgium (Blonk Consultants 2015)
Ready-to-eat Romaine lettuce	0.14	Average of UK and Spain values from (i Canals et al. 2008), at farm gate	10.4	Average of UK and Spain values from (i Canals et al. 2008), at farm gate + 0.0562 kWh/kg lettuce for processing
NFC OJ	0.71	(Dwivedi et al. 2012), at processor gate	8.96	(Beccali et al. 2010), minus 36% attributable to distribution; at processor gate
Chopped tomatoes	0.67	(Del Borghi et al. 2014), at processor gate	9.15	(Del Borghi et al. 2014), at processor gate
Mushroom	1.75	Primary data gathered from Highline Mushrooms, Ontario, CA, and modeled in SimaPro	25.3	Primary data gathered from Highline Mushrooms, Ontario, CA, and modeled in SimaPro
Potatoes	0.20	(Williams et al. 2006; Moudry Jr et al. 2013), average of 4 scenarios, at farm gate	1.27	(Williams et al. 2006), average of 2 scenarios, at farm gate
Eggs	1.7	Average values from four studies: (Pelletier et al. 2013, 2014; Cederberg et al. 2009; Nielsen et al. 2013), at farm gate	12.3	Average values from four studies: (Pelletier et al. 2013, 2014; Cederberg et al. 2009; Nielsen et al. 2013), values for whole egg at farm gate
Potato chips	1.98	(Nilsson et al. 2011), at processor gate	22.8	(Nilsson et al. 2011), at processor gate
Milk	1.05	(Thoma et al. 2013), at processor gate	3.99	(Gronroos et al. 2006), at processor gate
Turkey	5.42	Average of 4 production systems at farm gate from Leinonen et al. (2014), converted to carcass weight using dress yield of 79.13%	29.4	Average of 4 production systems at farm gate from Leinonen et al. (2014), converted to carcass weight using dress yield of 79.13% + 3.85 MJ/(kg dress carcass) for processing (Ramirez et al. 2006)
Pork	6.45	(Thoma et al. 2011), at processor gate, boneless equivalents	22.5	Average of 4 upper midwest U.S. scenarios, (Pelletier et al. 2010), boneless equivalents
Cheese	6.62	(Kim et al. 2013) Based on as-sold basis (incl. moisture), at processor gate	25.2	(Kim et al. 2013) Based on as-sold basis (incl. moisture), at processor gate
Beef	14.5	(Battagliese et al. 2013) adjusted to processor gate, boneless equivalents	30.0	(Battagliese et al. 2013) adjusted to processor gate, boneless equivalents

Note: Included life cycle stages have been truncated in all cases to either at farm gate or processor gate, as indicated for each; impacts exclude packaging and distribution.
NFC OJ = not-from-concentrate orange juice; GHG = greenhouse gas; kg CO_2eq./kg = kilograms of carbon dioxide equivalents per kilogram; CED = cumulative energy demand; MJ/kg = megajoules per kilogram.

offer a process for refrigerated shipping), the trucking process was modified to account for refrigeration by the following.

The majority of medium to large vehicles use self-contained refrigeration units that utilize a self-contained diesel engine. Various sources estimate the fuel consumption of these compressor engines to be 1 to 5 liters (L) per hour (Tassou et al. 2012; Roibás et al. 2014); we chose a value of 2 L per hour diesel consumption. Assuming an average operating truck speed of 56.3 miles (90.6 km) per hour (Statista 2015) and 6 hours of idling per day (Gaines et al. 2006), or 6 hours every 1,013 miles (1,630 km), we estimate a diesel consumption of 0.0295 L/km. In addition, a refrigerant leakage of 0.0052 grams of R134a/km (Roibás et al. 2014) was also assumed.

Transport distance from unspecified processors to retail outlets across the country is difficult to determine accurately. Where no additional information was available to estimate otherwise, transport distance was based on "average miles per shipment" in Table 2: "Shipment Characteristics of Temperature Controlled Shipments by Three-Digit Commodity for the United States: 2012" in the 2012 Commodity Flow Survey

(U.S. Department of Transportation 2015). Specific transport distances for each food are reported in supporting information S2 on the Web.

Retail Energy Use

Energy use (and associated emissions) at retail are divided into two pieces: refrigeration, and all other energy uses, including space heating and cooling, ventilation, water heating, lighting, cooking, and office equipment and computers. "Food sales" sector data from the 2003 U.S. Energy Information Administration (EIA) Commercial Buildings Energy Consumption Survey (U.S. Energy Information Administration 2006) are used to represent nonrefrigeration energy use. This energy use is then allocated to product categories on an economic basis. While a physical basis for allocation (likely area in this case) is preferred where possible according to International Organization for Standardization (ISO) 14044 standards (ISO 2006b), the complexity and variability of the national food retail sector prohibits such methods here. To perform the economic allocation, total annual national sales at retail for the food in question (e.g., beef) is divided by total supermarket sales (US$475,317 million in 2013 according to Progressive Grocer's Annual Consumer Expenditures Study [Progressive Grocer 2014]). This ratio is multiplied into the energy-use numbers and then divided by total annual kg of food commodity sold at retail to arrive at an energy use per kg. It was assumed that space heating, water heating, and cooking utilize natural gas, whereas all other end uses utilize electricity (U.S. national grid average).

While refrigeration energy is available through the above source, because packaging configuration can influence impacts, it is desirable to allocate it on a more physical (rather than economic) basis to individual food products. We estimate energy use for specific commercial refrigeration equipment via the U.S. Department of Energy equipment standards (U.S. Department of Energy 2014). This document provides maximum daily energy consumption (kilowatt-hours [kWh]/day) for various equipment categories, for example: for "vertical open equipment" with "remote condensing" operating at "medium temperature (38°F = 3.3°C)," the standard energy level is given by (equation 2):

$$\mathrm{E}\,(\mathrm{kWh/day}) = 0.66 \times TDA + 3.05 \qquad (2)$$

where TDA = total display area of the case, in square feet.

Appropriate equipment types and sizes are chosen for each food type, and the energy use per day is allocated to an individual product with the ratio of consumer facing area per kg for the product in question to TDA. This value is then averaged annually and nationally by multiplying by 365 and by total number of retail stores (37,716 in 2014 [FMI 2014]) and divided by the kg of food commodity sold annually at retail (i.e., annual throughput).

Refrigerant leakages also contribute to global warming. US EPA estimates annual U.S. supermarket refrigeration leakage to be 397 kg/year and assumes R-404A to be the typical commercial refrigerant used (US EPA 2011). To estimate the refrigerant leakage per kWh of refrigeration energy used, this value is divided by the total annual refrigeration energy for food sales (U.S. Energy Information Administration 2006). This leakage per kWh is then multiplied by the refrigeration energy consumption as calculated above to allocate a portion of the leakage to a given product.

Transport: Retail to Home

The 2009 National Household Transportation Survey (Santos et al. 2011) reports that the average vehicle trip length for shopping is 6.4 miles. We use this distance as a proxy for average grocery trips, and utilizing a process for "transport in passenger car with internal combustion engine" from ecoinvent 3.1, we allocate this transportation burden to the individual product in question on an economic basis (total annual sales of product in question/total annual supermarket sales).

Home Refrigeration

The 2009 Residential Energy Consumption Survey (U.S. Energy Information Administration 2013) reports that the annual energy consumption per household by refrigerators is 1,259.9 kWh, and the average refrigerator volume is 22 cubic feet (ft^3) (0.62 cubic meters). The annual energy use is divided by 365 to provide a daily energy use and allocated to the food product in question based on a volume fraction (volume per kg of food package in question divided by 22 ft^3). While packaging offers varying shelf life stability to foods, residence time in home refrigeration is determined largely by consumer behavior, and no empirical data are available. Rather than introduce subjective uncertainty, we have assumed a default of 4 days in home refrigeration for all foods requiring refrigeration.

Food Waste Rates

The rate of food wastage at retail and consumer stages is central to the trade-off explored in this study. They are also challenging to quantify. Consumer-level food waste at the individual product level is, for all practical purposes, unavailable. Gathering such data would require extensive (and expensive) surveying, and is outside of the scope of this project. In this study, we rely on waste rates from the USDA's LAFA data set (USDA ERS 2013), provided at the food commodity level, as an estimate for product-specific waste rates (see table 1). These represent the best estimate for food loss at the consumer level, considered broadly as a national average.

End-of-Life Disposal of Food and Packaging

Modeling of EoL disposal of food and packaging follows the US EPA's Waste Reduction Model (WARM; version 13) (US EPA 2015). The WARM model uses a life cycle approach to estimate energy use (or credit) and GHG emissions associated with recycling, combustion, composting, and landfilling of different materials. While the WARM model uses the avoided burden approach to credit recycling by the offset of virgin material (US EPA 2015), in our model we account for the influence of recycled content in material production via a recycled content (or cut-off) method. Thus, recycling aids the system by avoiding EoL burdens from landfill or incineration,

Table 3 Modeled fractions of disposal pathways for various materials

Material	*Recycled*[a]	*Landfilled*[c]	*Combusted*[c]
Food	4.8%[b]	78.1%	17.1%
PET	24.2%	62.2%	13.6%
HDPE	16%	68.9%	15.1%
PVC	0	82%	18%
LDPE	11.5%	72.6%	15.9%
PP	2.1%	80.3%	17.6%
PS	3.8%	78.9%	17.3%
PLA	0[b]	82%	18%
Steel	72.2%	22.8%	5.0%
Aluminum can	54.6%	37.2%	8.2%
Aluminum foil	0	82%	18%
Glass	34.1%	54.0%	11.9%
Corrugated cardboard	90.9%	7.4%	1.6%
Other paper	24.7%	61.7%	13.6%
wood	25.1%	61.4%	13.5%

[a]Recycling rates for the year reported (2012) from US EPA MSW data tables (US EPA 2014).
[b]Represents percentage composted.
[c]Derived by subtracting recycling fraction and distributing remaining by national average municipal solid waste (MSW) disposal ratio: 82% landfill, 18% incineration.
PET = polyethylene terephthalate; HDPE = high-density polyethylene; PVC = polyvinyl chloride; LDPE = low-density polyethylene; PP = polypropylene; PS = polystyrene; PLA = polylactic acid.

but does not result in a material displacement credit at the EoL process.

US EPA Municipal Solid Waste data (US EPA 2014) were used to establish the default fractions distributed to recycling (or composting), landfill, and combustion pathways. These fractions are based on U.S. national averages for 2012. The fractions used in the model are shown in table 3.

Impact Assessment Methods

This study focused on global warming potential (GWP) and nonrenewable CED. Energy demand is a valuable indicator in considering food/packaging systems as many other impact categories correlate with energy (Huijbregts et al. 2006), and, given the embodied energy of packaging materials, energy demand provides information not captured by GHG emissions. GWP was characterized using the Intergovernmental Panel on Climate Change (IPCC) 2013 GWP 100a method (IPCC 2013). Nonrenewable CED was calculated using the method published by ecoinvent version 2.0 (Frischknecht and Jungbluth 2003): Nonrenewable fossil, nonrenewable nuclear and nonrenewable biomass energy demands were summed in the results presented, although sums throughout are dominated by fossil nonrenewable energy demand. Inventory data necessary to evaluate additional indicators of interest, such as water and land use or eutrophication, were not available for agricultural production of all of the food types considered, and therefore this demonstration assessment focuses on GHG emissions and nonrenewable CED.

Results

Food to Packaging Environmental Impact Relationship: Literature Review

While impacts associated with agricultural production dominate many food life cycles, this can vary significantly depending on food type and scenario specifics, as revealed in a review of existing literature that applies LCA to various food product chains. Figure 2 presents the FTP GHG emissions (FTP_{GHGE}) ratio for a large number of food products, aggregated by food type. The FTP_{GHGE} ratio was calculated according to equation (1), using GHG emissions data across life cycle stages extracted from the literature (see supporting information S1 on the Web for details of the literature review and calculations). For reference, the cases evaluated in this study are also included in figure 2.

While large variation clearly exists, general trends in figure 2 are informative: Cereals, dairy, fish and seafood, and meats have large FTP ratios relative to other food types. When FTP ratios are high, it is more likely for changes in packaging configuration that lead to food waste reduction to result in net system decreases in environmental impact, even when packaging impacts increase.

Characterization of Food/Packaging Life Cycles

Figure 3 provides the distribution of GHG emissions across the full cradle-to-grave life cycle stages for the food/packaging combinations modeled in this study. Note that contributions due to food waste accumulate across the life cycle, but are represented as a separate "stage" in figure 3 in order to demonstrate their relative contribution. Foods in figure 3 are ordered left to right by the percent contribution from food production and processing. Thus, on the left are foods where GHG emissions from producing the consumed portion are small relative to the contribution from other stages (packaging, transport, and accumulated food waste impacts). Foods on the right are dominated by food production impacts. Lettuce and orange juice show disproportionately high distribution burdens because it was assumed that they were produced in a single U.S. location and distributed to the continental U.S. population; upwards of 75% of U.S. lettuce is produced in California whereas 90% of U.S. orange juice is made from Florida-grown oranges. The distribution of nonrenewable CED across life cycle stages is provided in figure S1 in supporting information S1 on the Web. The trend is similar to figure 3, although packaging production represents a

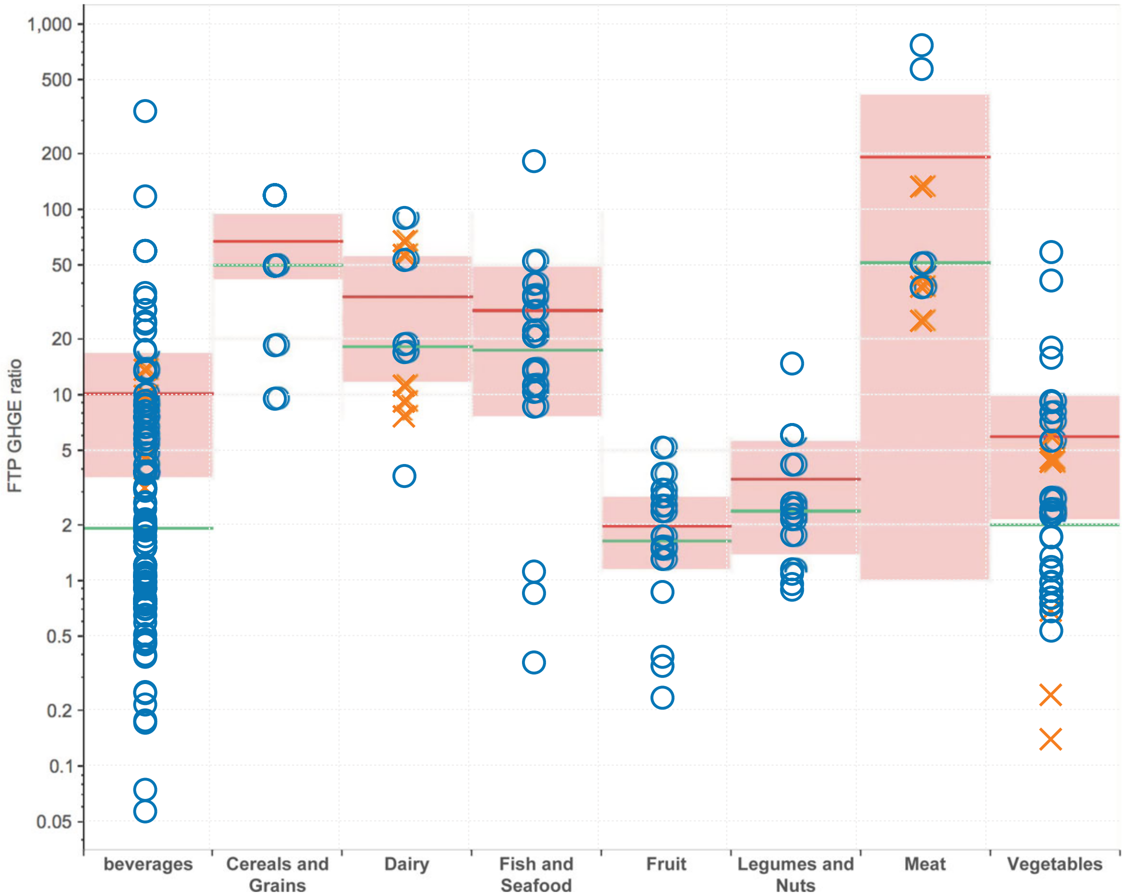

Figure 2 Demonstration of the "food to packaging" (FTP_{GHGE}) ratio for a large number and variety of foods and packaging configurations (beyond those identified in table 1 and assessed in this study). See supporting information S1 on the Web for details of the literature review and calculations. The vertical scale is presented as logarithmic in order to compactly show a wide range of values. Red horizontal bars represent average values for each food grouping, and boxes are 95% confidence intervals around the average. Green horizontal bars represent median values for each food grouping. The cases modeled in the current study are shown as orange "x"s for reference. Other foods, packaging configurations, system boundaries, and background data conditions represented in this figure are as reported in the literature and do not reflect the current study.

larger percentage of energy demand due to the embodied energy of the packaging materials.

Demonstrating Influence of Food Waste: Hypothetical Waste Rate Reduction Scenarios

The underlying premise in including the impact of food waste in evaluating packaging environmental performance is that improvements in packaging that can reduce food waste may result in net system environmental benefits even if the impacts of the packaging system itself increase. To demonstrate the relationships between environmental impacts of food production, packaging, and food waste, we assume a 10% hypothetical reduction in the baseline waste rates and use the LCA model to calculate the relative increase in packaging system impacts (primary and tertiary packaging material production, packaging disposal) that could be afforded by such waste reductions. Figure 4 shows this increase in GHG emissions associated with the packaging system that would break even with 10% reductions in retail waste rate, consumer waste rate, or both. In figure 4, this permissible increase in packaging GHG emissions is plotted against FTP_{GHGE} for the food/packaging combinations. A trend begins to emerge in 4 At very low FTP_{GHGE}, limited increases in packaging impacts are permitted with food waste reduction. At high FTP_{GHGE}, large increases in packaging impacts can be tolerated if they lead to such food waste reductions. While there is a notable trend with FTP_{GHGE}, this ratio alone is not predictive of system response to a reduction in food waste rate; the magnitude of the baseline waste rate is also important. However, the exercise does begin to map out the space available for changes in packaging systems. As changes in packaging would likely also influence processing-, distribution-, retail-, disposal-, and consumer-stage behaviors, this available "space" should be considered conceptually as not just for packaging materials per se, but for all of these associated factors.

Figure 5 gives the same relationships but with non-renewable CED. A similar trend exists, but the influence of food waste

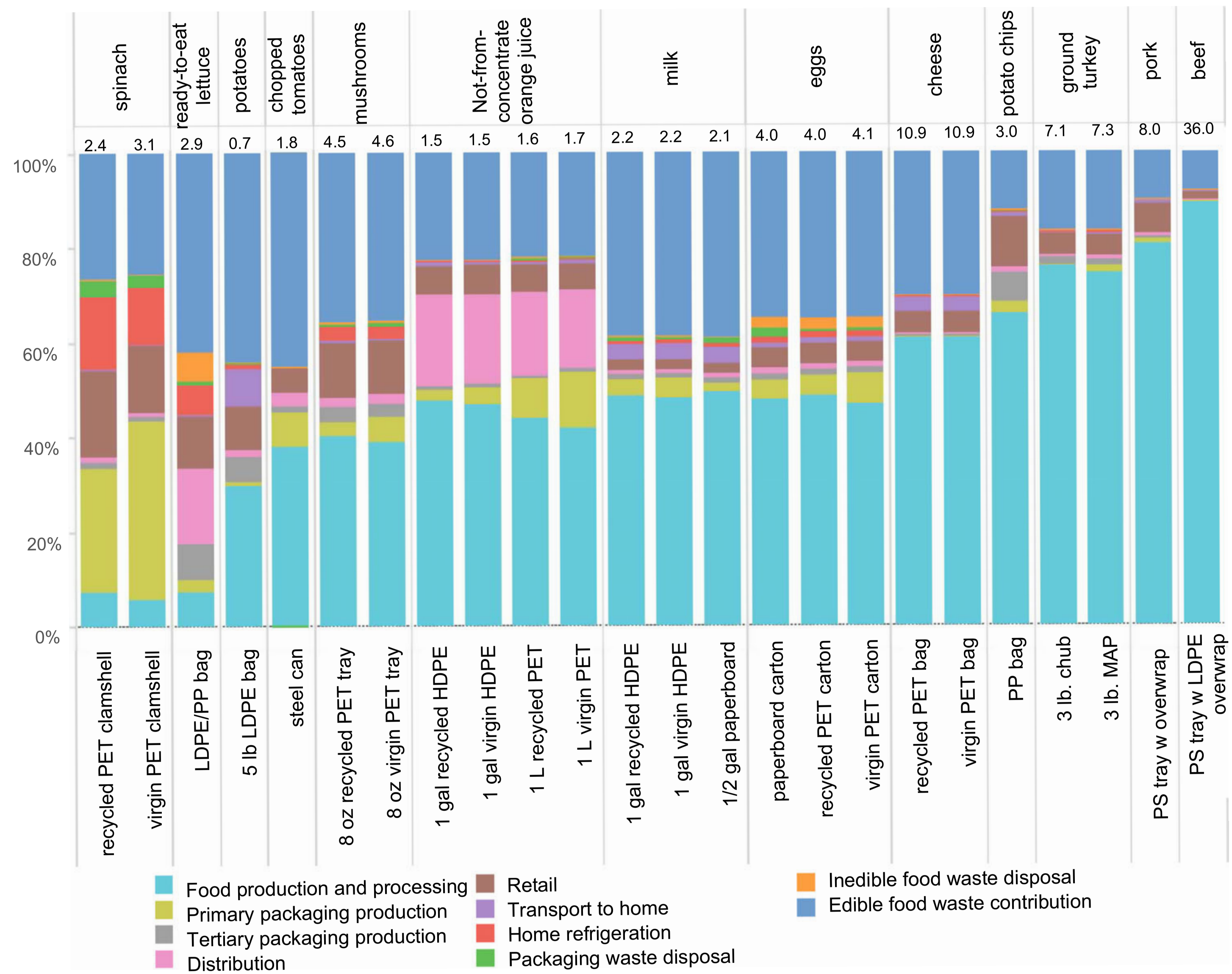

Figure 3 Distribution of GHG emissions across cradle-to-grave life cycle stages for the food/package combinations in table 1. Values above bars represent total GHG emissions in kg CO_2-eq. (kg consumed)-1. Note that "edible food waste contribution" includes emissions associated with edible retail- and consumer-level food waste accumulated throughout the life cycle: production, packaging, distribution, retail, refrigeration, and disposal. GHG = greenhouse gas; kg CO_2eq. = kilograms of carbon dioxide equivalents.

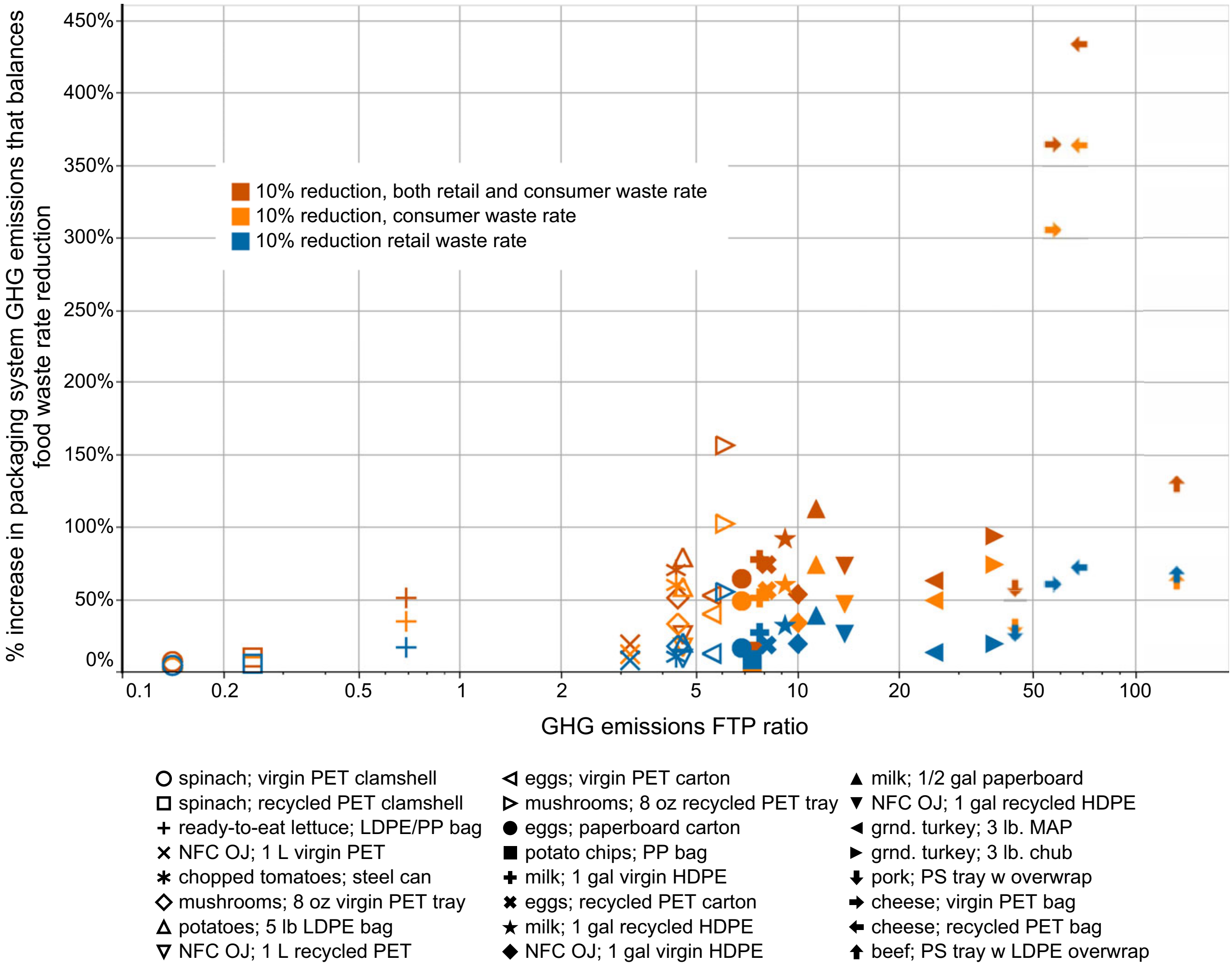

Figure 4 Demonstration of the increase in GHG emissions associated with the packaging system (primary, tertiary, and disposal) that would balance a 10% reduction in food waste rate at the retail (blue symbols), consumer (orange symbols), and retail plus consumer (red symbols) level for the food: packaging combinations in table 1. The allowable percent increase in packaging GHG emissions is plotted against FTP_{GHGE} (food production GHG emissions to packaging production GHG emissions, calculated without food waste contributions). Note that the x-axis is logarithmic merely to display a wide range of values efficiently. GHG = greenhouse gas.

is not as strong for non-renewable CED, largely because of the embodied energy in packaging materials (which does not present itself in GHG emissions) and the agricultural emissions not related to fossil fuel use (enteric methane and field-level N_2O emissions). Because of these factors, the difference between energy demand for food production and energy demand for packaging production is smaller, resulting in lower values of FTP_{CED}.

As a reminder, the intention of this paper and the demonstrations presented in figures 4 and 5 are meant to show general patterns and to highlight the influence that food waste can have in food/packaging systems; care must therefore be taken in drawing far-reaching or detailed conclusions. For assessments meant to serve as the basis for intervention, LCA should be performed using specific data and parameters for both the packaging and food product in question.

Sensitivity Analysis

To demonstrate the influence of parameter uncertainty on system performance, we have considered a ±20% change in individual parameters. The resultant change in overall system GHG emissions and CED for the cases of spinach in polyethylene terephthalate (PET) clamshell (low FTP) and ground turkey in MAP packaging (high FTP) are presented in table S4 in supporting information S1 on the Web. Agricultural production and processing dominate the system contributions to total CED and GHG emissions in the turkey case (see figure 3 and figure S1 in supporting information S1 on the Web), and therefore show strong sensitivity to parameter perturbations (18% and 13% changes in system GHG emissions and CED, respectively, from 20% changes in agricultural production impact per kg). Sensitivity of total GHG emissions to agricultural production contributions is greater than that of CED due to non-energy-related GHG emissions (nitrous oxide [N_2O],

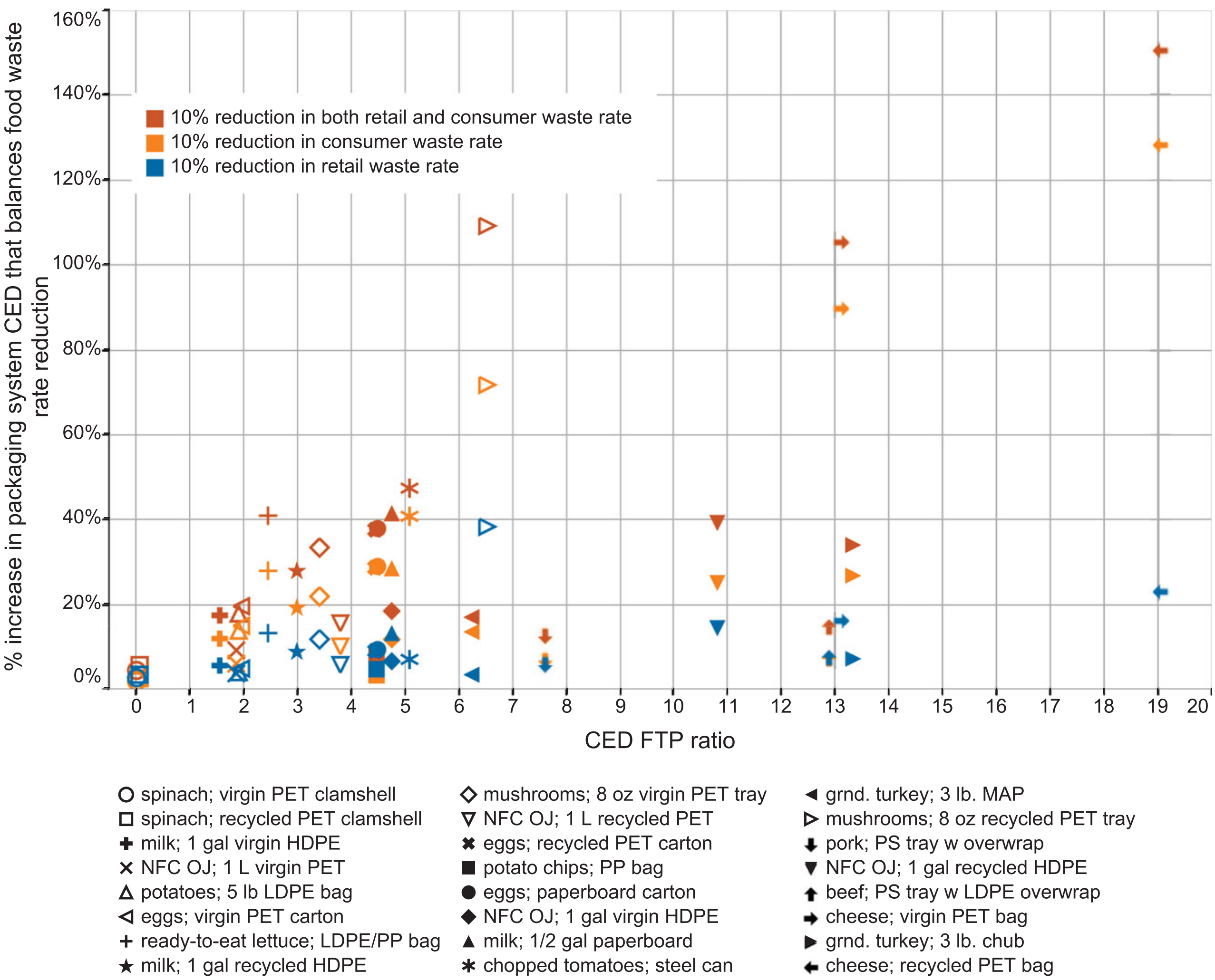

Figure 5 Demonstration of the increase in nonrenewable CED associated with the packaging system (primary, tertiary, and disposal) that would balance a 10% reduction in food waste rate at the retail (blue symbols), consumer (orange symbols), and retail plus consumer (red symbols) level for the food: packaging combinations in table 1. The allowable percent increase in packaging energy demand is plotted against the FTP ratio (food production energy demand to packaging production energy demand, calculated without food waste contributions). CED = cumulative energy demand.

methane) in agriculture as well as embodied feedstock energy contributions from packaging materials that do not have a direct GHG emissions component. With exception to agricultural production- and consumer-level food waste rate, all other modeling parameters in the turkey case demonstrate less than 3% change in total impacts from 20% parameter perturbations. Total system impacts for spinach show much lower sensitivity to agricultural production GHG emissions and CED and increased sensitivity to primary packaging weight. The spinach case is also more sensitive to changes in food waste rates and parameters that are associated with the impacts of refrigeration. Due to the lower FTP ratio for spinach, the case is more sensitive to the weight of primary packaging, but still less than 10% change in total impacts from a 20% perturbation. This analysis suggests that uncertainties in model parameters are likely to have notably less influence on results than the anticipated uncertainty in food waste rates.

Discussion

This study analyzes a group of generic foods in typical packaging configurations in order to demonstrate the influence of food waste on product system (food plus packaging) environmental performance. The underlying implication is that changes in food packaging configurations aimed at reducing food waste at the retail and consumer level can reduce environmental impacts of the product system even with increases in the impact of the packaging itself.

Packaging Design and Food Waste

Food packaging design can influence food waste in a variety of ways. The most obvious, of course, is through protecting food from mechanical damage (e.g., bruising, crushing) and physical-chemical degradation (e.g., oxidation, microbial spoilage). Countless examples of packaging that extend product shelf-life exist, but consumer preference often interferes with optimization of shelf-life extension (consider, e.g., vacuum packaging of beef). Packaging can also influence food waste at the consumer-level *beyond* its ability to postpone spoilage. A survey of Swedish households determined that 20% to 25% of household food waste was related to packaging design attributes, including the attributes *easy to empty* and *contains the correct quantity* (Williams et al. 2012). Additional packaging attributes that can influence food waste include *resealability*, *easy to: open, grip, and dose*, and communication of *food safety/freshness information* (Wikstrom et al. 2014; Lindh et al. 2016). When such attributes are considered from the standpoint of reducing food waste, the potential of packaging to improve system environmental performance may be realized.

Food-to-Packaging Ratio: Useful Indicator?

In figures 4 and 5, the FTP ratio offers a general orienting trend to the role of food waste reduction in total system environmental impact. Figure 2 provides a broader perspective on the variability of FTP_{GHGE} across food types, based on literature reported food LCAs. Consideration of this ratio, even at a scan-level approximation using best available data, may help packaging engineers direct attention to appropriate impact reduction strategies. At very low FTP ratios, it is likely preferable to focus attention on reducing the impact of the packaging—through lightweighting, alternative material selection, etc.—as food waste reduction will not have significant influence on the total system environmental performance. At very high FTP ratios, where emissions or resource use of food production are much larger than that of the packaging, emphasis on food waste reduction will likely yield larger system benefits. At intermediate FTP ratios, trade-offs require evaluation on a case-by-case basis. Key product chain characteristics, most notably heated greenhouse production and air freighting, are important to consider in such a scan-level approximation, however, as they could greatly influence food production impacts. For example, tomatoes grown in heated greenhouses can have carbon footprints 2 to 3 times those grown in open field or under unheated, protective structures (Webb et al. 2013; Theurl et al. 2014; Page et al. 2012). One example of air-freighted green beans places the carbon footprint at 20 to 26 times that of regional production without air freight (Sim et al. 2007).

Study Limitations

The differences between results based on nonrenewable CED and GHG emissions also emphasize the danger of relying on single environmental category assessments, especially when involving agricultural products. While it may be common with industrial products for other impact categories to trend with fossil fuel use, biological and field-level emissions in agriculture can disrupt this trend. Speaking very generally, we can expect food product system eutrophication and water-use impacts to be dominated by agricultural production; other categories such as acidification potential, ozone depletion potential, photochemical smog potential, and human health impacts such as respiratory effects will require case-by-case evaluation.

We use food loss data from the USDA LAFA data set as our baseline estimates of retail- and consumer-level food waste rates. This data set is the only known collection that provides a consistent estimate of food losses across all food commodities in the U.S. diet, but it certainly presents challenges. First is the generic nature of food commodity categories. For example, the relatively high consumer loss rate for turkey likely reflects whole turkeys prepared for holidays and special occasions and may not be as reflective of the ground turkey products considered here. Second, LAFA reports food *losses*, which include avoidable food waste (spoilage, plate waste) as well as unavoidable losses of moisture and fat from cooking. We have attempted to account for these cooking losses with meats, but available estimates are strongly dependent on specific cuts of meat and cooking methods and, in the case of beef at least, do not appear to be compatible with LAFA reported losses. We have gathered actual retail-level waste rates from a U.S. regional food retailer for the foods considered here to compare against LAFA data. These waste rates, averaged over 2 years of sales at circa 200 storefronts, are notably smaller (factor of 10 or more) than the LAFA loss rates in most cases (see table S3 in supporting information S1 on the Web for values). Meats (turkey, pork, and beef) are the exception, where LAFA retail loss rates are close to the empirical values collected from our retail partner. At this stage, it is impossible to determine whether our gathered data reflect a more efficient retail business and the LAFA data are more appropriate national averages for retail losses. As indicated by the sensitivity results in table S4 in supporting information S1 on the Web, uncertainty of food waste rates can have a moderate influence on LCA results and should be taken into consideration before drawing conclusions.

Future Research and Data Needs

The above concerns signal the need for high-quality food waste rate data. Numerous efforts to improve our understanding of food waste are underway, including an international standard for food loss and waste accounting and reporting (Food Loss & Waste Protocol 2016), improved measurements by the Food Waste Reduction Alliance (Food Waste Reduction Alliance 2016), efforts to make decision makers and consumers aware of food waste through the Save Food Initiative (Save Food Initiative 2016), and others. Repeated analyses highlight the challenges presented by food date labeling schemes that vary in terminology and application from region to region, and are largely misunderstood by industry and consumers, leading to significant unnecessary food waste. A recent review of the history and current practices of date labeling concludes with a call to action to move toward uniformity in date labeling (Newsome et al. 2014). Innovations in "intelligent" packaging strive to augment or replace date labeling through various indicator technologies that sense, detect, or record changes in the product, the package, or its environment (Vanderroost et al. 2014;

Realini and Marcos 2014), whereas the emerging field of "responsive" food packaging is designing stimuli response systems enabling real-time food quality and food safety monitoring or remediation (Brockgreitens and Abbas 2016). These technologies may likely offer additional means for packaging to reduce food waste, but also further emphasize the need for LCA of the product/package system to assure net environmental benefits.

Establishing accurate consumer-level food waste rates is extremely difficult, especially for specific products. Conducting household surveys can be costly and laden with methodological challenges (Edjabou et al. 2015). A growing body of information on consumer behavior and psychology with regard to both packaging and food waste provides a starting point for initiatives and packaging design aimed at reducing consumer-level food waste (Aschemann-Witzel et al. 2015; Stancu et al. 2016; Secondi et al. 2015; Neff et al. 2015; Martinho et al. 2015; Wikström et al. 2016). Trade-offs between consumers' desire for convenience, consumer perceptions of packaging, food waste generation, and whole product chain environmental impact have also been explored by comparing ready-to-eat meals with meals prepared at home (Hanssen et al. 2017).

Conclusion

Investments in packaging have the potential to reduce overall environmental impacts associated with food production, distribution, and consumption, through reducing food loss and waste. A systems approach using LCA will help to determine the potential benefits and guide packaging design decisions. The hypothetical waste rate reduction scenarios presented here begin to map out the opportunity space available for packaging innovations that lead to reduced food waste and net system impact reduction, even when impacts of the packaging system increase. For some foods, such as leafy greens, where agricultural production burdens are small and FTP ratios as small, net system impacts are sensitive to packaging production impacts, whereas in other cases, such as meats, with high FTP ratios, food wastes dominate the trade-off with packaging impacts. This study provides a framework to evaluate the environmental trade-offs between package configurations and food waste that can also be used to explore other relevant impacts such as water stress. Ongoing improvements in food waste data collection are needed to fully inform packaging design decision making.

Funding Information

This research was sponsored by the Center for Packaging Innovation and Sustainability at Michigan State University.

References

Andersson, K., T. Ohlsson, and P. Olsson. 1994. Life cycle assessment (LCA) of food products and production systems. *Trends in Food Science & Technology* 5(5): 134–138.

Aschemann-Witzel, J., I. de Hooge, P. Amani, T. Bech-Larsen, and M. Oostindjer. 2015. Consumer-related food waste: Causes and potential for action. *Sustainability* 7(6): 6457–6477.

Battagliese, T., J. Andrade, I. Schulze, B. Uhlman, and C. Barcan. 2013. *More Sustainable Beef Optimization Project: Phase 1 Final Report*. Florham Park, NJ, USA: BASF Corporation. http://bit.ly/NCBA_Report. Accessed January 2015.

Beccali, M., M. Cellura, M. Iudicello, and M. Mistretta. 2010. Life cycle assessment of Italian citrus-based products. Sensitivity analysis and improvement scenarios. *Journal of Environmental Management* 91(7): 1415–1428.

Blonk Consultants. 2015. Agri-Footprint 2.0. www.agri-footprint.com/. Accessed 2016.

Brockgreitens, J. and A. Abbas. 2016. Responsive food packaging: Recent progress and technological prospects. *Comprehensive Reviews in Food Science and Food Safety* 15(1): 3–15.

Cederberg, C., U. Sonesson, M. Henriksson, V. Sund, and J. Davis. 2009. *Greenhouse gas emissions from Swedish production of meat, milk and eggs 1990 and 2005*. Report No. 793. Borås, Sweden: SIK-Institutet för livsmedel och bioteknik.

Del Borghi, A., M. Gallo, C. Strazza, and M. Del Borghi. 2014. An evaluation of environmental sustainability in the food industry through life cycle assessment: The case study of tomato products supply chain. *Journal of Cleaner Production* 78: 121–130.

Dou, Z., J. D. Ferguson, D. T. Galligan, A. M. Kelly, S. M. Finn, and R. Giegengack. 2016. Assessing US food wastage and opportunities for reduction. *Global Food Security* 8: 19–26.

Dwivedi, P., T. Spreen, and R. Goodrich-Schneider. 2012. Global warming impact of Florida's Not-From-Concentrate (NFC) orange juice. *Agricultural Systems* 108: 104–111.

Edjabou, M. E., M. B. Jensen, R. Götze, K. Pivnenko, C. Petersen, C. Scheutz, and T. F. Astrup. 2015. Municipal solid waste composition: Sampling methodology, statistical analyses, and case study evaluation. *Waste Management* 36: 12–23.

FAO (Food and Agriculture Organization of the United Nations). 2013. *Food wastage footprint: Impacts on natural resources Summary Report*. Rome: Natural Resources Management and Environment Department, Food and Agriculture Organization of the United Nations. www.fao.org/nr/sustainability/food-loss-and-waste/en/. Accessed January 2014.

FMI (Food Marketing Institute). 2014. Supermarket facts. www.fmi.org/research-resources/supermarket-facts. Accessed August 4 2015.

Food Loss & Waste Protocol. 2016. Food Loss and Waste Accounting and Reporting Standard, Version 1.0. http://flwprotocol.org/. Accessed July 2016.

Food Waste Reduction Alliance. 2016. Analysis of U.S. food waste among food manufacturers, retailers, and restaurants. www.foodwastealliance.org/about-our-work/assessment/. Accessed March 2014.

Frischknecht, R. and N. Jungbluth. 2003. *Implementation of Life Cycle Impact Assessment Methods*. *Final report ecoinvent 2000*. Duebendorf, Switzerland: Swiss Centre for LCI. www.ecoinvent.ch.

Gaines, L., A. Vyas, and J. L. Anderson. 2006. Estimation of fuel use by Idling commercial trucks. In 85th Annual Meeting of the Transportation Research Board, 22–26 January, Washington, DC.

Gronroos, J., J. Seppala, P. Voutilainen, P. Seuri, and K. Koikkalainen. 2006. Energy use in conventional and organic milk and rye bread production in Finland. *Agriculture, Ecosystems and Environment* 117(2–3): 109–118.

Guinee, J. B., R. Heijungs, G. Huppes, A. Zamagni, P. Masoni, R. Buonamici, and T. Rydberg. 2011. Life cycle assessment: Past, present, and future. *Environmental Science & Technology* 45(1): 90–96.

Gustavsson, J., C. Cederberg, U. Sonesson, R. van Otterdijk, and A. Meybeck. 2011. *Global food losses and food waste*. Rome: Food and Agriculture Organization of the United Nations.

Hanssen, O. J., M. Vold, V. Schakenda, P.-A. Tufte, H. Møller, N. V. Olsen, and J. Skaret. 2017. Environmental profile, packaging intensity and food waste generation for three types of dinner meals. *Journal of Cleaner Production* 142(pt 1): 395–402.

Heller, M. C. and G. A. Keoleian. 2015. Greenhouse gas emission estimates of U.S. dietary choices and food loss. *Journal of Industrial Ecology* 19(3): 391–401.

Heller, M. C., G. A. Keoleian, and W. C. Willett. 2013. Toward a life cycle-based, diet-level framework for food environmental impact and nutritional quality assessment: A critical review. *Environmental Science & Technology* 47(22): 12632–12647.

Huijbregts, M. A. J., L. J. A. Rombouts, S. Hellweg, R. Frischknecht, A. J. Hendriks, D. van de Meent, and J. Struijs. 2006. Is cumulative fossil energy demand a useful indicator for the environmental performance of products? *Environmental Science & Technology* 40(3): 641–648.

Canals, L. M., I. Muñoz, A. Hospido, K. Plassmann, S. McLaren, G. Edwards-Jones, and B. Hounsome. 2008. *Life cycle assessment (LCA) of domestic vs. imported vegetables. Case studies on broccoli, salad crops and green beans*. CES working paper 01/08. Guildford, UK: Centre for Environmental Strategy (CES), University of Surrey.

IPCC (Intergovernmental Panel on Climate Change). 2013. *Climate Change 2013. The Physical Science Basis. Working Group I contribution to the Fifth Assessment Report of the IPCC*. Geneva, Switzerland: IPCC. www.ipcc.ch/. Accessed August 2016.

ISO (International Organization for Standardization). 2006a. *Environmental management—Life cycle assessment—Principles and framework*. ISO 14040:2006. Geneva, Switzerland: International Organization for Standardization.

ISO (International Organization for Standardization). 2006b. *Environmental management—Life cycle assessment—Requirements and guidelines*. ISO 14044:2006. Geneva, Switzerland: International Organization for Standardization.

Kim, D., G. Thoma, D. Nutter, F. Milani, R. Ulrich, and G. Norris. 2013. Life cycle assessment of cheese and whey production in the USA. *The International Journal of Life Cycle Assessment* 18(5): 1019–1035.

Leinonen, I., A. G. Williams, and I. Kyriazakis. 2014. Comparing UK turkey production systems using analytical error propagation in uncertainty analysis. Paper presented at 9th International Conference on Life Cycle Assessment in the Agri-Food Sector, San Francisco, CA, USA. http://lcafood2014.org/proceedings/LCA_Food_2014_Proceedings.pdf. Accessed November 2016.

Lindh, H., H. Williams, A. Olsson, and F. Wikström. 2016. Elucidating the indirect contributions of packaging to sustainable development: A terminology of packaging functions and features. *Packaging Technology and Science* 29(4–5): 225–246.

Martinho, G., A. Pires, G. Portela, and M. Fonseca. 2015. Factors affecting consumers' choices concerning sustainable packaging during product purchase and recycling. *Resources, Conservation and Recycling* 103: 58–68.

Moudry Jr., J., Z. Jelinkova, M. Jaresova, R. Plch, J. Moudry, and P. Konvalina. 2013. Assessing greenhouse gas emissions from potato production and processing in the Czech Republic. *Outlook on Agriculture* 42(3): 179–183.

Neff, R. A., M. L. Spiker, and P. L. Truant. 2015. Wasted Food: U.S. consumers' reported awareness, attitudes, and behaviors. *PLoS One* 10(6): e0127881.

Newsome, R., C. G. Balestrini, M. D. Baum, J. Corby, W. Fisher, K. Goodburn, and F. Yiannas. 2014. Applications and perceptions of date labeling of food. *Comprehensive Reviews in Food Science and Food Safety* 13(4): 745–769.

Nielsen, N., M. Jørgensen, and I. Rasmussen. 2013. *Greenhouse Gas Emissions from Danish Organic Egg Production estimated via LCA Methodology*. Aarhus, Denmark: Knowledge Center for Agriculture, Poultry. www.landbrugsinfo.dk/fjerkrae/klima-og-energi/sider/klima_uk_okoaeg.pdf. Accessed 22 February 2018.

Nilsson, K., V. Sund, and B. Florén. 2011. *The environmental impact of the consumption of sweets, crisps and soft drinks*. Copenhagen: TemaNord. www.scp-knowledge.eu/sites/default/files/knowledge/attachments/TN2011509 web%5B1%5D.pdf. Accessed October 2013.

Page, G., B. Ridoutt, and B. Bellotti. 2012. Carbon and water footprint tradeoffs in fresh tomato production. *Journal of Cleaner Production* 32: 219–226.

Pelletier, N., M. Ibarburu, and H. Xin. 2013. A carbon footprint analysis of egg production and processing supply chains in the Midwestern United States. *Journal of Cleaner Production* 54: 108–114.

Pelletier, N., M. Ibarburu, and H. Xin. 2014. Comparison of the environmental footprint of the egg industry in the United States in 1960 and 2010. *Poultry Science* 93(2): 241–255.

Pelletier, N., P. Lammers, D. Stender, and R. Pirog. 2010. Life cycle assessment of high- and low-profitability commodity and deep-bedded niche swine production systems in the Upper Midwestern United States. *Agricultural Systems* 103(9): 599–608.

Progressive Grocer. 2014. 68th Annual Consumer Expenditures Survey. *Progressive Grocer* 93(7): 40–50.

Ramirez, C., M. Patel, and K. Blok. 2006. How much energy to process one pound of meat? A comparison of energy use and specific energy consumption in the meat industry of four European countries. *Energy* 31(12): 2047–2063.

Realini, C. E. and B. Marcos. 2014. Active and intelligent packaging systems for a modern society. *Meat Science* 98(3): 404–419.

ReFED. 2016. *A Roadmap to Reduce U.S, Food Waste by 20 Percent*. www.refed.com. Accessed March 2016.

Roibás, L., A. Elbehri, and A. Hospido. 2014. Carbon footprint along the Ecuadorian banana supply chain: Methodological improvements and calculation tool. Paper presented at 9th International Conference LCA of Food, 8–10 October, San Francisco, CA, USA. http://lcafood2014.org/proceedings/LCA_Food_2014_Proceedings.pdf. Accessed November 2016.

Roy, P., T. Orikasa, M. Thammawong, N. Nakamura, Q. Y. Xu, and T. Shiina. 2012. Life cycle of meats: An opportunity to abate the greenhouse gas emission from meat industry in Japan. *Journal of Environmental Management* 93(1): 218–224.

Russell, D. A. 2014. Sustainable (food) packaging—An overview. *Food Additives & Contaminants, Part A* 31(3): 396–401.

Santos, A., N. McGuckin, H. Y. Nakamoto, D. Gray, and S. Liss. 2011. *Summary of Travel Trends: 2009 National Household Travel Survey*. FHWA-PL-11-022. Washington, DC: U.S. Department of Transportation. http://nhts.ornl.gov/2009/pub/stt.pdf. Accessed April 2015.

Save Food Initiative. 2016. www.save-food.org/. Accessed 3 January 2017.

Schau, E. M. and A. M. Fet. 2008. LCA studies of food products as background for environmental product declarations. *The International Journal of Life Cycle Assessment* 13(3): 255–264.

Secondi, L., L. Principato, and T. Laureti. 2015. Household food waste behaviour in EU-27 countries: A multilevel analysis. *Food Policy* 56: 25–40.

Showell, B. A., J. R. Williams, M. Duvall, J. C. Howe, K. Y. Patterson, J. M. Roseland, and J. M. Holden. 2012. *USDA Table of Cooking Yields for Meat and Poultry*. Beltsville, MD, USA: USDA Agricultural Research Service. www.ars.usda.gov/nutrientdata. Accessed August 2016.

Silvenius, F., K. Grönman, J. M. Katajajuuri, R. Soukka, H. K. Koivupuro, and Y. Virtanen. 2013. The role of household food waste in comparing environmental impacts of packaging alternatives. *Packaging Technology and Science* 27(4):277–292.

Sim, S., M. Barry, R. Clift, and S. J. Cowell. 2007. The relative importance of transport in determining an appropriate sustainability strategy for food sourcing. *The International Journal of Life Cycle Assessment* 12(6): 422–431.

Stancu, V., P. Haugaard, and L. Lähteenmäki. 2016. Determinants of consumer food waste behaviour: Two routes to food waste. *Appetite* 96: 7–17.

Statista. 2015. Average operating truck speed on selected U.S. interstate highways in 2009. www.statista.com/statistics/195100/average-operating-truck-speed-on-selected-us-interstate-highways/. Accessed 12 August 2015.

Tassou, S. A., G. De-Lille, and J. Lewis. 2012. *Food transport refrigeration*. Uxbridge UK: Brunel University, Centre for Energy and Built Environment Research, School of Engineering and Design. www.grimsby.ac.uk/documents/defra/trns-refrigeenergy.pdf. Accessed August 2015.

Theurl, M. C., H. Haberl, K. H. Erb, and T. Lindenthal. 2014. Contrasted greenhouse gas emissions from local versus long-range tomato production. *Agronomy for Sustainable Development* 34(3): 593–602.

Thoma, G., R. E. Martin, D. Nutter, R. Ulrich, C. Maxwell, J. Frank, and C. East. 2011. *National Life Cycle Carbon Footprint Study for Production of US Swine*. Des Moines, IA, USA: National Pork Board. www.pork.org/filelibrary/NPB Scan Final - May 2011.pdf. Accessed October 2013.

Thoma, G., J. Popp, D. Nutter, D. Shonnard, R. Ulrich, M. Matlock, and F. Adom. 2013. Greenhouse gas emissions from milk production and consumption in the United States: A cradle-to-grave life cycle assessment circa 2008. *International Dairy Journal* 31(s1): S3–S14.

U.S. Department of Energy. 2014. *Technical Support Document: Energy Efficiency Program For Consumer Products And Commercial And Industrial Equipment. Commercial Refrigeration Equipment*. Washington, DC: U.S. Department of Energy.

U.S. Department of Transportation. 2015. *2012 Commodity Flow Survey*. EC12TCF-US. www.census.gov/econ/cfs/2012/ec12tcf-us.pdf. Accessed May 2015.

U.S. Energy Information Administration. 2006. *2003 Commercial Buildings Energy Consumption Survey*. www.eia.gov/consumption/commercial/data/2003/pdf/e01a.pdf. Accessed April 2015.

U.S. Energy Information Administration. 2013. *2009 Residental Energy Consumption Survey*. Washington, DC: U.S. Department of Energy. www.eia.gov/consumption/residential/data/2009/c&e/ce3.1.xlsx. Accessed May 2015.

US EPA (U. S. Environmental Protection Agency). 2011. *Profile of an Average U.S. Supermarket's Greenhouse Gas Impacts from Refrigeration Leaks Compared to Electricity Consumption*. www.epa.gov/sites/production/files/documents/gc_averagestoreprofile_final_june_2011_revised_1.pdf. Accessed April 2015.

US EPA (U. S. Environmental Protection Agency). 2014. *Municipal Solid Waste Generation, Recycling, and Disposal in the United States: Facts and Figures for 2012*. EPA-530-F-14-001. Washington, DC: United States Environmental Protection Agency Solid Waste and Emergency Response. https://www.epa.gov/sites/production/files/2015-09/documents/2012_msw_dat_tbls.pdf. Accessed 22 February 2018.

US EPA (U. S. Environmental Protection Agency). 2015. *Documentation for Greenhouse Gas Emission and Energy Factors Used in the Waste Reduction Model (WARM), version 13*. https://archive.epa.gov/epawaste/conserve/tools/warm/pdfs/WARM_Documentation.pdf. Accessed 21 February 2018.

United Nations. 2015. Sustainable Development Goals. https://sustainabledevelopment.un.org/sdgs. Accessed 4 April 2017.

USDA (U.S. Department of Agriculture). 2015. USDA and EPA Join with Private Sector, Charitable Organizations to Set Nation's First Food Waste Reduction Goals. www.usda.gov/wps/portal/usda/usdamediafb?contentid=2015/09/0257.xml&printable=true&contentidonly=true. Accessed August 2016.

USDA ERS (U.S. Department of Agriculture Economic Research Service). 2013. Loss-adjusted food availability documentation. USDA Economic Research Service. www.ers.usda.gov/data-products/food-availability-per-capita-data-system/loss-adjusted-food-availability-documentation/. Accessed September 2016.

Vanderroost, M., P. Ragaert, F. Devlieghere, and B. De Meulenaer. 2014. Intelligent food packaging: The next generation. *Trends in Food Science & Technology* 39(1): 47–62.

Webb, J., A. G. Williams, E. Hope, D. Evans, and E. Moorhouse. 2013. Do foods imported into the UK have a greater environmental impact than the same foods produced within the UK? *The International Journal of Life Cycle Assessment* 18(7): 1325–1343.

Wikstrom, F. and H. Williams. 2010. Potential environmental gains from reducing food losses through development of new packaging—A life-cycle model. *Packaging Technology and Science* 23(7): 403–411.

Wikstrom, F., H. Williams, K. Verghese, and S. Clune. 2014. The influence of packaging attributes on consumer behaviour in food-packaging life cycle assessment studies—A neglected topic. *Journal of Cleaner Production* 73: 100–108.

Wikström, F., H. Williams, and G. Venkatesh. 2016. The influence of packaging attributes on recycling and food waste behaviour—An environmental comparison of two packaging alternatives. *Journal of Cleaner Production* 137: 895–902.

Williams, A., E. Audsley, and D. Sandars. 2006. *Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities*. Defra project report IS0205. http://randd.defra.gov.uk/Default.aspx. Accessed July 2015.

Williams, H. and F. Wikström. 2011. Environmental impact of packaging and food losses in a life cycle perspective: A comparative analysis of five food items. *Journal of Cleaner Production* 19: 43–48.

Williams, H., F. Wikström, and M. Löfgren. 2008. A life cycle perspective on environmental effects of customer focused packaging development. *Journal of Cleaner Production* 16(7): 853–859.

Williams, H., F. Wikström, T. Otterbring, M. Löfgren, and A. Gustafsson. 2012. Reasons for household food waste with special attention to packaging. *Journal of Cleaner Production* 24: 141–148.

Supporting Information

Supporting information is linked to this article on the *JIE* website:

Supporting Information S1: This supporting information provides details of data sources and impact factors for packaging materials and transportation, results on energy demand distribution, and food waste rates utilized compared with empirically collected values. It also includes results from a sensitivity analysis and the methods and citation sources for Figure 2 in the main article.

Supporting Information S2: This supporting information provides the food/packaging scenario modeling parameters, and a reference list of sources used.