Retail Climate Change Mitigation: Life-Cycle Emission and Energy Efficiency Labels and Standards

Prepared for the California Air Resources Board and the California Environmental Protection Agency

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Abstract

Strategies for reducing the life-cycle greenhouse gas (GHG) emissions of products are receiving increasing attention around the world as climate change mitigation policy options. One notable example is the use of product "carbon footprint" labels, the purpose of which is to provide consumers with enabling information for low-carbon purchasing decisions. This project developed a multi-region input-output (MRIO) based life-cycle assessment (LCA) model for California to explore the role that product life-cycle carbon labels and efficiency standards might play in reducing the state's annual GHG emissions. The California MRIO LCA model was applied to estimate the life-cycle GHG emissions of 22 different products in the state, and was further coupled with life-cycle design and technology improvement analyses to estimate the GHG emissions that might be realized for each product under different scenarios. The results suggest that energy-using devices and animal-based food items offered the greatest potential reductions among the products considered, that services might be overlooked as a supply chain GHG emissions reduction opportunity, and that purchase volume, market uptake, and stock turnover considerations are key characteristics affecting the identified GHG emissions reduction opportunities. Data uncertainties were found to be a key issue for conducting such analyses. The methods developed in this study could provide the state with valuable screening capabilities for identifying products and services that hold the greatest potential for in-state emissions reductions under life-cycle GHG emissions policy initiatives.

Executive Summary

Background

Under Executive Order S-3-05, California has committed to reducing its greenhouse gas (GHG) emissions to 80% below 1990 levels by the year 2050. While California has a strong track record of policies aimed at reducing the operational energy use and GHG emissions of retail products (e.g., appliance efficiency standards), little attention has been paid to reducing the GHG emissions of purchased products and services across their entire life cycle (i.e., production, use, and disposal). Such life-cycle emissions may account for up to two-thirds of the annual "carbon footprint" of the typical California household. Thus, the life-cycle GHG emissions associated with retail products might represent an untapped source of potential GHG emissions reductions for the state. Two potential policy models that have been investigated elsewhere, and are aimed at these potential GHG emissions reductions, are: (1) product carbon footprint labels, the purpose of which is to provide consumers with enabling information for low-carbon purchasing decisions, and (2) life-cycle energy use and GHG emissions standards, which would set minimum performance standards for purchased products. However, quantitative estimates of the GHG emissions that might be saved for different products, and where these emissions savings might occur, are lacking in the public domain. Such estimates are critical for sound policy making.

Methods

The project sought to answer two primary research questions:

1. By how much might GHG emissions be reduced across the life-cycle of a given product if carbon labels and/or standards are successful in driving the market to best practice for low carbon and energy efficient life cycles?

2. Of the estimated emissions reductions, how much is likely to occur within California?

To help answer these questions, this project first developed a comprehensive California LCA model to estimate the life-cycle GHG emissions of retail products, both inside and outside the state. This study considered 22 different product cases, which were selected by CARB from a preliminary short list of retail products and services prepared by the research team. The model consists of four primary analysis modules. The first module developed by the research team is a multi-regional input-output (MRIO) model that is capable of estimating the full production (or value) chain energy use and GHG emissions of a wide variety of products. The second module estimates the factory-to-retail transportation phase energy use and GHG emissions of the 22 selected retail products in California. The third module estimates the use phase energy use and GHG emissions of the 22 selected retail products in California based on best-available operational energy use data. The use phase includes product operations and maintenance activities that require energy use; for example, direct electricity use by a computer or indirect electricity use via refrigeration of food items. The fourth module estimates the end of life phase energy use and GHG emissions associated with disposal activities associated with the 22 selected retail products (landfill, waste-to-energy recovery, recycling, and composting). These modules comprise the final California MRIO LCA model developed in this study.

Next, the research team coupled outputs from the California MRIO model with analyses of "best practice" design features and life-cycle technology performance to estimate the life-cycle energy use and GHG emissions of hypothetical "low carbon" versions of the 22 selected products. The low carbon case was an approximation of the minimum life-cycle GHG emissions that are currently realistic for a given product, and was meant to approximate the "best in class" products that may appear on the market in response to California life-cycle GHG emissions labeling and standards programs.

Using results for current and "low carbon" versions of the 22 products, the research team then explored the GHG emissions reductions that might be achieved for the 22 selected products under different consumer adoption and policy scenarios over a five year projection period (2011-2015). Included were a business as usual scenario, a total "low carbon" technical potential scenario, a technical potential scenario that considered the limitations of stock turnover for key products, a carbon label uptake scenario, and a life-cycle product performance standards scenario.

Results

The results suggest that the California MRIO LCA model, when populated with product-specific information and coupled with product-level technical analyses of potential life-cycle design, operations, and technology improvements, could provide the state with valuable screening capabilities for identifying products and services that hold the greatest potential for in-state emissions reductions under life-cycle GHG emissions policy initiatives. For the 22 products considered in this study, it was found that energy-using devices and animal-based food items offered the greatest potential reductions among the products considered. In terms of the estimated potential GHG emissions reductions, 10 of the 22 products considered accounted for 90% of the estimated reduction potential. Of these 10, four were energy-using devices (refrigerator, water heater, flat panel TV, and desktop PC control unit) and four were animal-based food items (beef, milk, cheese, and chicken). The total technical potential for life-cycle GHG emissions for the 22 products was estimated at 29 Tg CO2e over the period 2011-2015, which accounted for stock turnover as a limiting factor for the achievable GHG emissions savings. Restaurants—the only commercial sector expenditure considered in this study—were estimated to be the number one emitter of life-cycle GHG emissions also ranked number one in terms of estimated GHG emissions reduction potential. The implication of these results is that the service sectors might hold large potential for GHG emissions reductions, but services have been largely overlooked to date in carbon labeling and standards initiatives (which have focused strictly on consumer products). The study also found that only roughly two-thirds of the technical potential GHG emissions reductions was likely to occur within the state. When exploring how natural market uptake might affect the potential savings, the study found that even if consumers reacted with the same levels of uptake as products labeled with the ENERGY STAR—which is arguably the greatest labeling success story in the United States—that only roughly 40% of the technical potential might be realized. However, setting best practice standards for life-cycle GHG emissions for product purchases in the state might provide greater reductions—up to 75% of the estimated technical potential—since such standards would limit consumer choice to top performing products. Data uncertainties were found to be a key issue that should be addressed using the methodology moving forward.

Conclusions

The results suggest that the California MRIO LCA model, when populated with product-specific information and coupled with product-level technical analyses of potential life-cycle design, operations, and technology improvements, could provide the state with valuable screening capabilities for identifying products and services that hold the greatest potential for in-state emissions reductions under life-cycle GHG emissions policy initiatives. The results also suggest that significant life-cycle GHG emissions reductions might be achievable via product carbon labels and/or life-cycle standards for the 22 products considered in this study, but future work is needed to address some of the key limitations of the methods and data sources summarized in this report. Key recommendations for addressing these limitations include compilation of more detailed energy use and emissions data for California economic sectors, more detailed data on annual purchases by California households, more precise data on the life-cycle energy use, emissions, and life-cycle system characteristics of specific products, and more in-depth explorations of data and model uncertainties, and their implications for interpreting the results of the analytical methods presented here.

Introduction

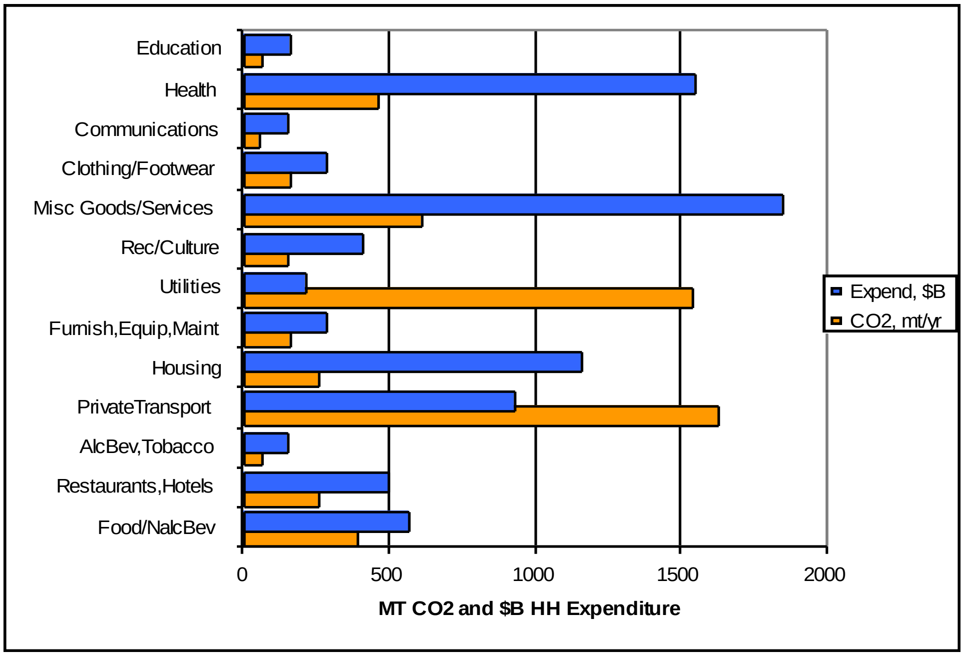
Background

Under Executive Order S-3-05, California has committed to reducing its greenhouse gas (GHG) emissions to 80% below 1990 levels by the year 2050. Given the enormity of this challenge, California (and other world regions that have committed to similar long-term GHG emissions reductions) must consider a broad range of potential GHG reduction policies, including those that extend beyond established practices such as building energy codes, appliance efficiency standards, and vehicle efficiency standards.

Of the annual GHG emissions attributable to the typical U.S. household, roughly one-third is associated with private transportation and another one-third is associated with residential energy use. Figure 1 summarizes recent estimates by Weber and Matthews (2008), which demonstrate this typical household emissions breakdown. The blue bars in the figure indicate the total U.S. household spending in each expenditure category, while the orange bars indicate the estimated annual GHG emissions associated with these expenditures. As can be seen in Figure 1, the remaining one-third of annual emissions is typically attributable to the production and disposal of the goods and services consumed by the household. Such emissions are sometimes referred to as the “embodied emissions” of household consumption. If one considers that the vast majority of residential energy use (and associated GHG emissions) is attributable to purchased appliances such as water heaters, refrigerators, furnaces, and entertainment equipment, the purchases of retail goods and services might comprise as much as two-thirds of annual U.S. household carbon emissions.

While California has a strong track record of policies aimed at reducing the operational energy use and GHG emissions of retail products (e.g., appliance efficiency standards), little attention has been paid to reducing the GHG emissions of purchased products and services across their entire life cycle (i.e., production, use, and disposal). Thus, the life-cycle GHG emissions associated with California’s retail products might represent an untapped source of potential GHG emissions reductions for the state. For example, for personal computers (PCs) in California, Masanet and Horvath (2006) identified manufacturing- and disposal-phase opportunities that could reduce the GHG emissions of California’s PCs by roughly 2.5 million metric tons of CO_2 per year.

Two potential policy mechanisms for reducing the life-cycle GHG emissions of products and services purchased by California households are: (1) the use of so-called “carbon footprint” labels on retail products, which provide the consumer with information on the GHG emissions implications of their purchasing decisions, and (2) the establishment of life-cycle energy efficiency and GHG emissions standards for retail products with significant life-cycle GHG emissions, which would set minimum standards for products sold within the state. The former strategy has gained increasing worldwide momentum, the most visible example being product carbon footprint labels and protocols developed by the UK’s Carbon Trust and the British Standards Institute (BSI 2008). The latter strategy would be modeled after successful standards initiatives for household appliances at the state and federal levels (e.g., California efficiency standards for refrigerators and ENERGY STAR efficiency standards for a wide range of devices).



Source: Weber and Matthews (2008)

Figure 1: Estimated annual GHG emissions for U.S. households (2004)

Effective GHG emissions reduction policies require thorough analysis of the potential benefits and costs of prospective policy measures to ensure that meaningful GHG emissions reductions can be achieved, and that selected policy measures are the most promising among a portfolio of competing policy options. While there appears to be significant potential for reducing California's GHG emissions through management of the life-cycle GHG emissions of retail products and services, quantitative estimates of the GHG emissions that might be saved for different products, and where these emissions savings might occur, are lacking in the public domain.

As a result, the extent to which such labels and standards for retail products could lead to GHG emissions reductions in California is not yet clear. Analytical methods and policy initiatives related to product life-cycle GHG emissions labels and standards are still emerging, so there are few (if any) case study data to draw upon to measure the effectiveness of such programs. Moreover, robust analytical methods to quantify the in-state GHG emissions of California's retail products across their entire life cycle are currently lacking. There is also little direct guidance available to California to design and manage life-cycle GHG emissions labeling and standards programs as such initiatives are just now beginning to enter the global policy debate (see, for example, EC (2007)).

In order for the California Air Resources Board (CARB) to fully assess the opportunities associated with life-cycle GHG emissions labels and standards in relation to the state's AB 32 targets, there is a critical need for exploratory research to (1) develop credible estimates of the potential for in-state GHG emissions reductions associated with the full life-cycle of California's retail products, and (2) analyze the extent to which the identified GHG emissions reductions might be realized through retail product GHG emissions labels and standards in the state.

Research questions and objectives

The objective of this research project is to assess opportunities for reducing California's GHG emissions through the application of life-cycle GHG emissions labels and standards to retail products consumed by Californians.

The project sought to answer two primary research questions:

3. By how much might GHG emissions be reduced across the life-cycle of a given product if carbon labels and/or standards are successful in driving the market to best practice for low carbon and energy efficient life cycles?

4. Of the estimated emissions reductions, how much is likely to occur within California?

To help answer these questions, this project had the following specific research objectives:

Objective 1: Estimation of life-cycle GHG emissions attributable to retail products in California

This research objective was aimed at developing best-available estimates of the annual life-cycle GHG emissions attributable to retail products purchased by Californians. The goal was to develop a comprehensive California LCA model to estimate the life-cycle GHG emissions of retail products, both inside and outside the state. The development of this model involved six primary research tasks.

First, to develop a multi-regional input-output (MRIO) model that is capable of estimating the full production (or value) chain energy use and GHG emissions of a wide variety of products. For the purposes of this project, the production phase is defined as:

- The ***final manufacturing sector***, which refers to the final sector in the production system, which fabricates and/or assembles the finished product for sale and shipment to the retail outlet. For example, in the production of bottled wine, the final manufacturing sector is the winery sector.
- All contributing ***supply chain*** sectors. The supply chain refers to the extended system of sectors upstream of the final manufacturing sector, which supplies the materials, parts, and services necessary for the final manufacturing sector to produce its finished products. Using again the bottled wine example, the supply chain would include all processes for producing the bottles, grapes, labels, corks, etc. purchased by the winery sector.

Second, to apply the MRIO model to estimate the total annual production chain energy use and GHG emissions of a select set of retail products and services purchased by California households. This study considered 22 different product cases, which were selected by CARB from a preliminary short list of retail products and services prepared by the research team.

Third, to develop and apply a methodology to estimate the ***factory-to-retail transportation phase*** energy use and GHG emissions of the 22 selected retail products in California. This life-cycle phase is defined as the transportation of the finished product from the final manufacturer to retail outlet.

Fourth, to develop and apply a methodology to estimate the ***use phase*** energy use and GHG emissions of the 22 selected retail products in California based on best-available operational energy use data. The use phase includes product operations and maintenance activities that require energy use; for example, direct electricity use by a computer or indirect electricity use via refrigeration of food items.

Fifth, to develop and apply a methodology to estimate the ***end of life phase*** energy use and GHG emissions associated with disposal activities associated with the 22 selected retail products based on best available data. This life-cycle phase includes product landfill, waste-to-energy recovery, recycling, and composting.

Sixth, to incorporate the above methods into a user friendly California MRIO LCA model that can be applied to assess all phases of the life-cycle of products and services in California for the 22 selected products, as well as for other products of interest to CARB in the future. The model is designed to assess current emissions profiles for each product, which are referred to in the remainder of this report as the 2011 baseline case (or simply "baseline case").

Objective 2: Estimation of life-cycle GHG emissions reductions attainable for retail products in California

This research objective was aimed at estimating the life-cycle energy use and GHG emissions of hypothetical "low carbon" versions of the 22 selected products. The low carbon case is meant as an approximation of the minimum life-cycle GHG emissions that are currently realistic for a given product, and is meant to approximate the "best in class" products that may appear on the market in response to California life-cycle GHG emissions labeling and standards programs. Such "low-GHG" products could be the result of a manufacturer's aggressive pursuit of energy efficiency improvements to product manufacturing methods (e.g., the pursuit of ENERGY STAR plant labels) and product operating characteristics (e.g., the pursuit of ENERGY STAR appliance labels), and the use design for recycling techniques.

Objective 3: Analysis of policy scenarios for retail product labeling and standards programs in California

This research objective was aimed at estimating the GHG emissions reductions that might be achieved for the 22 selected products under different consumer adoption and policy scenarios over a five year projection period (2011-2015). More specifically, the goal was to leverage the California MRIO LCA model (developed to meet Objective 1) with the low carbon product estimates (developed to meet Objective 2) to estimate the potential for GHG emissions by product, life-cycle phase, and region of GHG emission reduction. These estimates were to be made using five different projection scenarios:

1. a ***"business as usual (BAU)" scenario***, which assumes that current per-product life-cycle GHG emissions will stay constant over the projection period , but increase in absolute fashion due to population growth;

2. a ***"low carbon technical potential (TP)" scenario***, which estimates the maximum achievable GHG emissions reductions by product if low carbon versions were maximally deployed;

3. a ***"stock turnover constrained TP" scenario***, which considers that durable goods (e.g., appliances) will likely only be replaced as they reach the end of their useful life, and therefore limit the pace at which low carbon versions of durable goods can be deployed over the projection period;

4. a ***"benchmark market uptake" scenario***, which estimates natural market uptake of the low carbon versions of the 22 products if such products displayed carbon labels, based on benchmarking to market uptake rates of ENERGY STAR labeled products as the best available proxies for labeled products in the United States; and

5. a ***"life cycle product standards" scenario***, which estimates the achievable GHG emissions reductions by product if maximum life-cycle emissions standards were to be set for each of the 22 products, and that would help cap the life-cycle emissions of products eligible to be sold in California.

Objective 4: Recommendations for future work

This research objective aimed to summarize concisely the "lessons learned" through this project, and to offer recommendations for future work. Specific goals were to identify opportunities for improving and expanding the analytical framework developed in this study, to identify data gaps in our analyses that could be filled through future work, and to offer recommendations for next steps and future research that could be pursued by the ARB (or other California agencies) to further its understanding and to build capacity in the area of product GHG emissions labels and standards.

Caveats

It is important to note that the goal of this research project was not to develop an official model for estimating the life-cycle GHG emissions of products and services under any future California carbon labeling and/or life-cycle standards policies. Nor was the project designed to provide precise estimates of the current life-cycle energy use or GHG emissions of the 22 products at the product level. Rather, the research project was designed to provide preliminary estimates of annual product-related GHG emissions at the state level to shed light on where in the world such emissions occur, and roughly what emissions reductions might be achieved through the adoption of best practice, low carbon product production and design practices.

Structure of this report

This report describes the development and results of an extensive modeling effort that involved the compilation, assessment, and synthesis of many different data from a wide range of sources as well as the generation of detailed results for 22 different product cases in five different scenarios. In order for the main body of this report to remain concise and readable, much detailed supporting information is provided in Appendices A through F. Interested readers are referred to the relevant appendix (and appendix tables) for more details on data, methods, and assumptions throughout the report as appropriate.

The Methods section provides a concise description of the overall modeling approach that was developed and applied in this study. Summaries are provided of each of the major components of the California MRIO LCA model, as well as the approaches for applying this model to estimate the current GHG emissions and achievable GHG emissions reductions for each product in different scenarios. Details on the MRIO approach, the 22 product selection process, and product case data derivations are provided in Appendices A through E.

The Results and Discussion section provides a concise summary of major results, and their interpretation with respect to the research goals of this study. Detailed results for each product case are provided in Appendix F.

The Summary and Conclusions section provides the research team's perspectives on the major findings of this study, their implications for better understanding the potential for product carbon labels and standards as a policy mechanism for reducing GHG emissions, and the limitations of the model, methods, and data presented in this report.

The Recommendations section builds upon the Summary and Conclusions section by describing opportunities for future work to overcome some of this study's key limitations, as well as for advancing the state's understanding of product and service related life-cycle GHG emissions.

Methods

This section provides an overview of the methods, data sources, and assumptions that were employed to meet the research objectives of this study. This section first provides a broad overview of the modeling methodology and structure of the California MRIO LCA model that was developed as a core deliverable of this project, and submitted to CARB in MS Excel format. Next, the key components of the model—which correspond to the life-cycle phases to be modeled under Objective 1—are described in terms of the key methods and data sources that comprise them. The application of the model to estimate both baseline (i.e., current) and low carbon technical potential GHG emissions of the 22 products is then discussed. Finally, the approaches used to project state-level GHG emissions under the five scenarios of Objective 3 are described. As mentioned in the Introduction section, to be concise the reader is referred to the relevant appendices for more details throughout this section.

Overview

Figure 2 provides a schematic of the overall modeling approach that was developed to meet Objective 1, and which was implemented in the California MRIO LCA model. Major model modules are indicated by bold boxes, and the results generated by the different modules are indicated by the double-lined boxes at the right of the figure. Arrows are meant as schematic indicators of the order of calculations in the model, and each module and key output is labeled numerically for ease of interpretation.

Given that the model is designed to estimate the total annual life-cycle GHG emissions associated with a product in the baseline case – which includes the annual purchases, use, and disposal of a given product – the baseline life-cycle GHG emissions are calculated as:

> *Annual GHG emissions from manufacture and transport of purchases of product X + annual use phase GHG emissions of product X + annual end of life phase GHG emissions of product X*

The same calculation scheme applies for estimating the total annual life-cycle energy use associated with a given product. The first term is estimated via the MRIO and transportation modules based on the annual purchases of product X in California, which is a function of California population and purchasing habits. The second term is estimated based on the unit energy consumption (UEC) module and the annual stock (i.e., installed base) of product X. The latter value is derived based on the estimated lifetime of product X, the annual purchases, the disposal rate, and population growth. The third term is estimated via the end of life (EOL) module and the assumed product lifetime, waste management system characteristics, and the mass fraction of product X that is disposed of using a particular EOL pathway.

Details on the methods, data, and assumptions for each module are described later in this section.

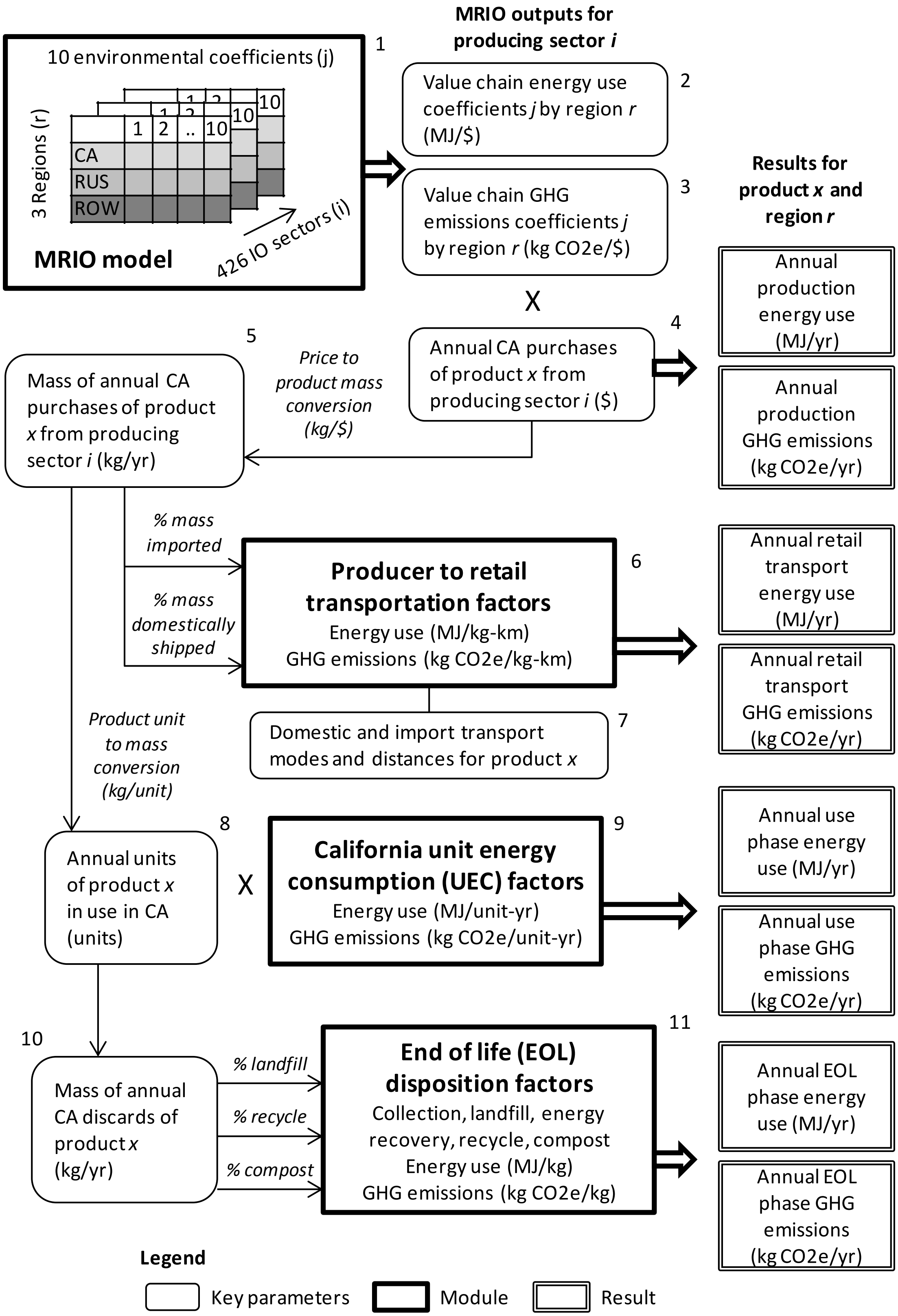


Figure 2: Schematic of the CA MRIO LCA model

Using the numeric labels in Figure 2, the basic methodology can be described as follows:

The MRIO module (1) estimates the production (i.e., value) chain energy use and emissions coefficients, in units of megajoules per dollar (MJ/$) and kilograms of carbon dioxide equivalents per dollar (kg CO_2e/$) attributable to 426 input-output (IO) sectors associated with a dollar of production from the final manufacturing IO sector *i*. The MRIO model estimates coefficients for 10 different types of fuels and GHG emissions, and within three different regions of production *r* (within California, outside California but within the United States, and outside the United States).

The production chain coefficients for energy use (2) and GHG emissions (3) are multiplied by the estimated annual purchases of a product *x* (4), which gives results for the annual production energy use and emissions associated with total purchases of product *x* in California for the baseline year (2011).

Using conversion factors (5), the total annual purchases of product *x* are converted to units of product mass (in kg), which are used to calculate the annual energy use and GHG emissions of factory-to-retail transportation for product *x* in the transportation factors module (6). This module estimates the impacts of transport activities related to products that are sourced both domestically and internationally, based on transportation mode intensity and distance factors (7).

Using conversion factors as well as information on the total installed stock of product *x* (8), the total annual purchases or mass shipped for product *x* is converted into product units for those products that consume energy during the use phase. The California UEC module (9) estimates the total annual energy use and GHG emissions associated with the installed stock of product *x* using California-specific energy use data.

Finally, the mass of product *x* that is disposed of each year is broken down into mass fractions going to landfill, recycling, and composting based on user-supplied data (10). The EOL factors module (11) estimates the annual energy use and GHG emissions associated with total disposals of product *x* by disposition pathway.

The methodology illustrated in Figure 2 provides estimates of the total life-cycle energy and GHG emissions associated with a given product *x*, by life-cycle phase, region of emission, fuel use type, and GHG emissions type for 2011 baseline case.

Figure 3 provides a schematic of how the California MRIO LCA model was utilized to estimate life-cycle energy use and GHG emissions associated with the 2011 low carbon technical potential case for each of the 22 products considered in this study, and in fulfillment of Objective 2. As suggested in Figure 3, five major life-cycle product opportunities were considered (as applicable) for each of the 22 products: (1) supply chain energy efficiency and GHG emissions abatement measures; (2) final manufacturing energy efficiency and GHG emissions abatement measures; (3) product design measures, including considerations for materials and operational energy efficiency and their effects on the transportation, use, and disposal life-cycle phases; (4) transportation efficiency measures; and (5) end of life considerations. Each opportunity area is highlighted within a dashed oval in Figure 3.

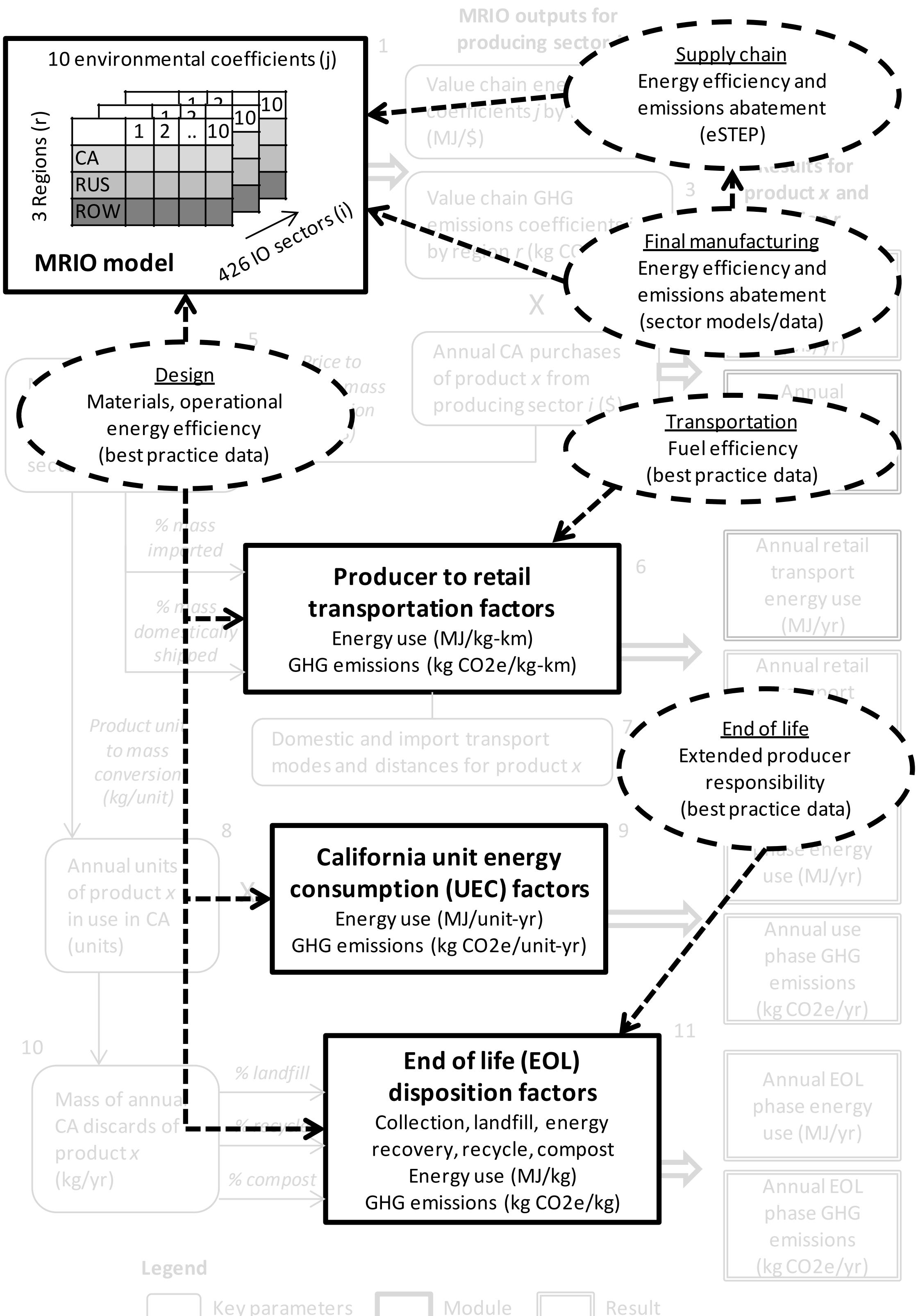


Figure 3: Schematic of the low carbon analysis approach

To estimate supply chain energy efficiency and GHG emissions abatement measures, the research team employed the supply chain technology potentials methodology developed in Masanet et al. (2009a), which is referred to as the Supply Chain Technology Potentials Model for Energy, Emissions, and the Environment (eSTEP). The eSTEP method and data were used to adjust the environmental coefficients in the MRIO model according to estimated "best practice" energy efficiency and emissions intensities of various IO sectors.

Given that for many products the final manufacturing sector accounts for significant fractions of the production chain energy and GHG emissions footprints, the MRIO model was also augmented (when possible) with best practice energy efficiency and emissions intensities for the 22 final manufacturing IO sectors considered in this study using sector-specific case studies and data.

Design changes and their implications for production chain, transportation, use phase, and end of life phase energy use and emissions for each product were considered with according adjustments made to the aforementioned modules as needed. Design changes were limited to best practices that could be verified in the literature for the 22 product case studies, rather than formulated based on conjecture about the universe of design opportunities for each product.

Best practice fuel efficiency data were employed to estimate the achievable energy use and GHG emissions intensity reductions for the transportation phase.

Further details on the methods, data sources, and assumptions associated with each module of the California MRIO LCA model, and the low-carbon product analysis approach, are provided in the remainder of this section.

MRIO model

This section provides a brief overview of the methods pertinent to the results and discussion presented in later sections.

Economic IO models were first developed by Leontief (1936) to aid manufacturing planning. Using linear algebra common in the economics literature [1], the models estimate all purchases and activities in a supply chain leading up to final manufacture in an industry. When the economic IO model is augmented with environmental information in matrix form, it estimates upstream life cycle environmental impacts of production activities by any sector in the economy. The basic IO model derives the total economic purchases (i.e., supply chain) across an economy required to make a desired output. Once the supply chain is calculated, environmental emissions can be estimated by multiplying the output of each sector by its environmental impact per dollar of output:

$$\mathbf{b_i} = \mathbf{R_i(I+A+AA+AAA+...)y} = \mathbf{R_i(I\text{-}A)^{-1}y} \qquad \textit{Eq. 1-1}$$

where bi is the vector of environmental burdens (such as GHG emissions for each production sector), Ri is a matrix with diagonal elements representing the emissions per dollar of output for each sector, I is the identity matrix (a table of all zeros except for the diagonal entries containing a 1), A is the direct requirements matrix (with rows representing the required inputs from other sectors to make a unit of output), and y is the vector of desired production or “final demand.” Terms in Equation 2-1 represent the production of the desired output itself (I⍰y), contributions from the direct or first level (“tier-1”) suppliers (A⍰y), those from the second level (“tier-2”) indirect suppliers (A⍰A⍰y), and so on. Direct emissions from a sector (i.e. Scope 1), including on-site emissions from activities like natural gas or petroleum combustion, and indirect emissions (Scopes 2 and 3) upstream of a sector can be calculated using modified forms of Equation 1-1 by selectively including certain terms from the equation and certain elements in the A matrix [6].

Uncertainties in Input-Output Models

As with any calculation based on IO-LCA methods, this method has substantial uncertainties related to sectoral aggregation; price, temporal, and spatial variation; and several other issues, as discussed elsewhere [18, 24, 32]. Since discussions of uncertainties inherent in IO-LCA can be found in other literature, and they are not unique to the approach used in this work, this section will focus on the type of uncertainty particularly relevant to application of these findings to carbon footprint protocol design—sector aggregation uncertainties.

Sector aggregation occurs when technically and environmentally distinct operations are combined into larger groups to form a sector or groups of sectors [2]. In the basic IO framework on which our methods are based, sector aggregation is decided by the BEA in compiling IO tables from survey data. In most cases, more detailed disaggregated IO data are not publicly available. Working with each sector’s environmental intensity information faces similar challenges. Because original environmental or energy consumption data sources may be organized by different industry classification systems (e.g., NAICS), bridging them with IO sector classification system inevitably results in losing information about more detailed sectors, and in some cases, introduces additional uncertainties from disaggregation of sectors. Even if sector aggregation is not the most significant issue, the environmental intensity information used in the model represents a national average of all the companies in the sector. It does not account for the variability in different companies' actual environmental intensity. For example, the carbon intensity of electricity production can vary significantly depending on the geographic region and the respective energy generation portfolio for the electricity grid. Under a certain sector aggregation scheme, aggregation errors may vary substantially depending on the sector or product considered [3]. Using data specific to the footprinting entity will produce more accurate footprint results, though allocation issues can confound even such specific data, as discussed in [4].

Multi-Regional Input-Output Model

This section describes the construction of the MRIO model used in this study, which consists of three regions, each at a different geographical scale: the "California" region is at the sub-nation scale, the "rest of the US" (RUS) region is at the national scale, and the "rest of world" (ROW) region is at the global scale. Each region is mutually exclusive even though one at a higher scale may encompass another at a lower scale geographically.

The mathematics of MRIO models are well known and described in detail elsewhere [52, 65, 67, 68, 71, 72]. In general, existing multi-regional models belong to two types: those dealing with regions smaller than national tables, and those dealing with several nations and the trade activities among them. Little literature exists on combining sub-national modeling with multi-national modeling, but in theory the development should be similar to existing methods for sub-national and multi-national modeling [5, 6].

The MRIO model is constructed in several steps. In the first step, the domestic and import portions of the EIO-LCA model are separated using the use table compiled by BEA [7]. In the second step, because input-output data are systematically compiled by states, a proxy data set for economic activities is identified to provide the basis for separating production and consumption in California from the rest of the US. In the third step, the Simple Location Quotient (LQ) method, a technique developed by IO researchers to estimate economic activities as proportion of total needs at regional level in the absence of survey data, is used to allocate California and RUS supplies that are used to meet California demands. In the fourth steps, the Employment Ratio Method (ERM) is used to allocate California and RUS supplies that are used for industry demands in the RUS region. Finally, all the components of the MRIO model is put together into a multi-regional A matrix. These methods are described in details in Appendix B.2.

Putting together California and RUS portions of the inter-industry transaction matrix, obtained from multiplying the domestic A matrix (Ad) with a vector of LQ factors, a vector of ERM factors, a vector of 1–LQ, and a vector of 1–ERM, a multi-regional A matrix is created to represent the inter-regional transaction of the industries in the California, RUS, and ROW regions. (Please refer to Appendix B.2 for a detailed description of the methods.) The 3-region A matrix and its 9 compartments are illustrated in Figure 4.

$$
\begin{array}{rc}
 & \begin{array}{ccc} \text{California} & \text{Rest of the US} & \text{Rest of World} \end{array} \\
\begin{array}{r} \text{California} \\[2em] \text{Rest of the US} \\[2em] \text{Rest of World} \end{array} &
\begin{pmatrix}
\begin{gathered} \mathbf{A}_{\mathbf{CA\text{-}CA}} \\ \langle LQ_{CA} \rangle \times A_d \end{gathered} & \begin{gathered} \mathbf{A}_{\mathbf{CA\text{-}US}} \\ \langle ERM_{CA} \rangle \times A_d \end{gathered} & \begin{gathered} \mathbf{A}_{\mathbf{CA\text{-}ROW}} \\ 0 \end{gathered} \\
\begin{gathered} \mathbf{A}_{\mathbf{RUS\text{-}CA}} \\ \langle 1 - LQ_{CA} \rangle \times A_d \end{gathered} & \begin{gathered} \mathbf{A}_{\mathbf{US\text{-}US}} \\ \langle 1 - ERM_{CA} \rangle \times A_d \end{gathered} & \begin{gathered} \mathbf{A}_{\mathbf{US\text{-}ROW}} \\ 0 \end{gathered} \\
\begin{gathered} \mathbf{A}_{\mathbf{ROW\text{-}CA}} \\ A_m \end{gathered} & \begin{gathered} \mathbf{A}_{\mathbf{ROW\text{-}US}} \\ A_m \end{gathered} & \begin{gathered} \mathbf{A}_{\mathbf{ROW\text{-}ROW}} \\ A \end{gathered}
\end{pmatrix}
\end{array}
$$

Figure 4: Three-region inter-industry transaction matrix, which consists of 9 blocks representing the transactions among the 3 regions

This 3-region model assumes that only one national border is crossed in the production of any good. This means that goods produced abroad use no CA or RUS-made components and no goods are exported from the US only to be returned in goods imported to the US. Therefore, the ROW-CA and ROW-US compartments are filled with the US import portions of the A matrix (Am), while the CA-ROW and US-ROW compartments are filled with zeros. This assumption has been tested previously in the literature and found to produce reasonable error levels [64, 68, 75]. Further, the rest of US and rest of world segments are assumed to produce goods similar to the US as a whole, such that the US model can be used as a proxy for these production technologies. Therefore, the ROW-ROW compartment is filled with the original A matrix from the basic US model.

This economic model is converted to an environmental model using vectors of environmental emissions per unit of economic output (in this case, energy and GHG emissions in metric tons of CO2 equivalents/ $1 million). California energy and GHG emissions vectors were constructed from a variety of data sources and are described in detail. US vectors for both GHG emissions and energy were taken from Carnegie Mellon University's Economic Input-Output Life-Cycle Assessment (EIO-LCA) model (Carnegie Mellon University Green Design Institute, 2008) which is thought to be the most comprehensive LCA model available for the United States. Because this model relies on data supplied by the Bureau of Economic Analysis, the most recent comprehensive information that is available is for the benchmark year 2002. Modifications to the MRIO model are described in detail below.

Modifications to California Energy Related GHG Emissions Data

California manufacturing energy use and greenhouse gas (GHG) emissions data were obtained from the CARB Greenhouse Gas Emissions Inventory. The allocation for this data is at a less detailed level than the EIO-LCA data and is approximately consistent with three digit North American Industrial Classification System (NAICS). In order to adjust the 428 IO sectors, scaling factors were created in a series of steps which are highlighted in equations 4 through 7. Using scaling factors does not produce results that are different than using raw data to calculate environmental vectors but rather they make implementation of aggregated data much simpler. For this model, GHG emissions factors were calculated from energy use based on the factors highlighted in Table 1.

Table 1: GHG emissions factors for various fuels

Fuel to GHG Conversions	Coal	Natural Gas	Petroleum	Biomass/Waste
(g CO2e / MJ)	**89.5**	**50.3**	**69.4**	**89.5**

The first step in this adjustment process for California's environmental vectors was to aggregate the IO data into sectors which were equivalent to the CARB sectors that are used in the GHG Emissions Inventory. This was done using Eq. 1-2.

$$E_{US\ CARB\ sector} = \frac{\sum_m E_n * Output_n^{US}}{\sum_m Output_n^{US}} \quad \textit{Eq. 1-2}$$

Equation 1-2 is used to calculate environmental burdens in the United States per million dollars at an equivalent level of aggregation as the existing CARB data where E is an environmental impact factor which has burden per million dollars, n is an IO sector within m which is the set of IO sectors that fall into a given CARB sector, and Output is the industrial output for IO sector n.

The next step in the process is to translate the CARB GHG Emissions Inventory data into impact vectors with units of environmental burden per million dollars.

$$E_{CA\ CARB\ sector} = \frac{Energy_{CA\ CARB\ sector}}{\sum_m Output_n^{US} * ERM_n} \quad \textit{Eq. 1-3}$$

Equation 1-3 is used to calculate energy use in California for a given CARB sector per million dollars where Energy is the total energy use in a CARB sector in the state of California, Output is the industrial output for sector n in a given region (US), and ERM is the employment ratio multiplier for sector n to scale the US industrial output to reflect the industrial output in California.

Following the calculation of the California industrial output per million dollars, a scaling factor was calculated by dividing equation 1-2 by equation 1-3 for a given CARB sector.

$$Scaling\ Factor_{CARB\ sector} = \frac{E_{CA\ CARB\ sector}}{E_{US\ CARB\ sector}} \quad \textit{Eq. 1-4}$$

The final step in the adjustment of the California vectors is to apply the CARB scaling factors calculated in equation 1-4 to the original US IO environmental vectors.

$$E_n^{CA} = E_n^{US} * Scaling\ Factor_{CARB\ sector} \quad \text{Eq. 1-5}$$

Equation 1-5 is used to calculate the final environmental impacts per million dollars for a given IO sector n based on the scaling factors calculated in equation 1-4 for each CARB sector.

Adjusting California's commercial IO sectors was similar but less complicated than the adjustments to the manufacturing sectors. Electricity data and natural gas data were obtained from the Commercial Building Energy Consumption Survey (CBECS) for the US on a square footage of floor space basis. Similar data was obtained from the California Commercial End-Use Survey (CEUS). Simple division of data from each survey gives a scaling factor similar to the one in equation 6. This data can then be applied in a manner that is similar to equation 7 in order to scale electricity and natural gas data for the state of California. Lacking California specific data for petroleum, biomass and waste, and nonfossil electric, the US data was left unaltered for these fuels.

The final adjustment to the California energy and resulting GHG emissions data comes in the "power generation and supply" sector which reflects purchased power from off-site sources. Each fuel in the EIO-LCA model was adjusted to reflect differences in the California energy grid relative to the national average electricity grid mix which is found in the EIO-LCA model.

$$Fuel_n^{CA} = Fuel_n^{US} * \frac{\frac{Fuel^{CA}}{All\ Fuels^{CA}}}{\frac{Fuel^{US}}{All\ Fuels^{US}}} \quad \textit{Eq. 1-6}$$

Equation 1-6 is used to recalculate the amount of energy per million dollars is supplied from energy purchase in California based on DOE's eGrid data for 2004 (United States Environmental Protection Agency, 2007) where Fuel represents the amount of a given fuel is used in the electricity grid in the region of interest and All Fuels represent the total fuel used in that region.

Table 2: Relative energy consumption per million dollars in California and the US in the power generation and supply sector

Region	Coal (MJ/$)	Natural Gas (MJ/$)	Petroleum (MJ/$)	Nonfossil Electric (MJ/$)	Total (MJ/$)
US	79	23	4	3	109
CA	2	69	4	4	78

The California electricity grid also has lower carbon intensity than the US average in its mix of electricity generation portfolio [9]. The emission factor for California's power generation sector is scaled down from the US emission factor by the ratio of carbon intensities in California and the US average (385/650) [10]. In terms of the ROW emission factors, while previous research suggests that the average mix of imports into the United States may have a higher CO2 intensity than domestically-made goods for some commodity classes [11], comparing such factors across countries is difficult due to several factors including exchange rates and lack of data [12]. Creating a dataset that would provide information about the environmental burdens of each IO sector would be an incredible undertaking and lies well outside of the scope of this research. One relevant study finds that food manufacturing outside of the United States had carbon intensities ranging from 30% to 300% of the US intensities (Weber & Matthews, 2008). The same article found that mining intensity ranged from 5% to 220%. While knowledge of the embedded emissions of internationally produced goods is important for other uses, it is nonessential to understanding the potential for California goods and services to reduce emissions through carbon labels. Region-specific environmental vectors can be constructed for each country that the United States trades with however, because life cycle emissions are likely to only be affected if a product is produced and purchased in California, the accuracy of environmental burdens worldwide is far less important to this analysis than the accuracy of environmental burdens in California and RUS.

Adjustments to California Process (non-fuel combustion) Emissions Data

Process emissions for carbon dioxide, methane, nitrous oxide, hydrofluorocarbons, perfluorocarbons, and sulfur hexafluoride were obtained from the CARB GHG Emissions Inventory. The CARB sectors that these process emissions were allocated to were used to replace corresponding IO sectors from the MRIO model by dividing the emissions data by the California industrial economic output for that sector (ERM * US Economic Output). Three sectors were excluded from this dataset as they were obvious outliers likely from mismatching sectoral definitions between the IO sectors and the CARB sectors.

Agricultural process emissions in the CARB GHG Inventory were allocated much differently than they are in the MRIO model and its underlying data. The CARB GHG Inventory aggregates this data based on agricultural processes such as enteric fermentation of cattle, manure management of swine, and agricultural residue burning of field crops. The IO data in the model is aggregated to individual sectors such as grain farming, milk production, and poultry and egg production which are derived from EPA's 2008 document entitled "Inventory of U.S. Greenhouse Gas Emissions and Sinks 1990-2008" (United States Environmental Protection Agency, 2010). The Process Emissions from this source were replaced with California specific emissions and then used to recalculate emissions per million dollars for individual IO sectors. Greater than 98% of the emissions factors in the model were either altered to reflect California specific process emissions or were considered to not have any process emissions for a given greenhouse gas in a specific sector. For the remaining sectors where emissions data was not available or data was excluded as an outlier, the emissions factor for the entire U.S. was used.

RUS vectors needed to reflect energy and greenhouse gas emissions data for the United States excluding the state of California. This adjustment took the total US energy consumption by fuel, subtracted the energy use in California, and finally divided by the industrial output for the US minus the industrial output for California.

$$E_n^{RUS} = \frac{(E_n^{US} * Output_n^{US}) - (E_n^{CA} * Output_n^{CA})}{(Output_n^{US} - Output_n^{CA})} \quad \textit{Eq. 1-7}$$

Equation 1-7 is used to calculate the RUS energy use vectors on a per million dollar basis.

Uncertainties of MRIO Model

As with any calculations based on IO-LCA methods, the model has substantial uncertainties related to sectoral aggregation and allocation, source data, environmental multipliers, price variation, inter-industry transaction data, temporal and spatial variations, import assumptions, proportionality assumption, and several other issues. Since discussions of uncertainties inherent in IO-LCA can be found in other literature [18, 24, 32, 33, 79, 86, 87], and they are not unique to the approach used in this work, this section will focus on three types of uncertainty particularly relevant to the approach: price uncertainty, regionalization assumption, and variation in environmental intensities.

Price Uncertainty

The implementation of the environmental intensity matrix in the basic EIO-LCA model assumes that every sector pays the same price for the commodities they purchase as inputs. Therefore, when calculating the total supply chain environmental impacts of a sector, the same set of "environmental intensity per dollar output" factors are applied systematically to the entire economy regardless of who are the purchasers of that commodity. In reality, there is variation in the prices paid by each sector, and large purchasers are likely to pay less (on a per unit of commodity basis) than small purchasers. Assuming that each sector pays the same price can skew the results by underestimating the environmental impacts of large purchasers, while overestimating the impacts of small purchasers that pay the higher unit price. Moreover, the price of the same commodity purchased by customers in the same sector may also vary by regions (states, counties, or metropolitan areas). While there are probably price variations for all commodity and sector combinations in the economy, this section focuses the price uncertainty discussion on electricity prices since some information about electricity price variation is available.

In the basic US model, an economy-wide average electricity price of $0.072/kWh and a national-average carbon intensity of electricity generation are assumed for all customers. However, 2002 EIA Energy Power Monthly data [15, 16] indicate that average residential customers pay $0.0844/kWh, commercial customers pay $0.0789/kWh, and industrial customers pay $0.0488/kWh [17]. Electricity price paid by detailed IO sectors can also be estimated using US Economic Census Fuels and Electric Energy Data [17, 18].

There is no easy way to adjust for the variation in electricity prices in the basic US IO model because the results produced by the model are the sums for all the power generation nodes in the entire supply chain tree (i.e., the sectoral sum of power generation sector at any tiers) without distinction in each power generation node's downstream customer. Other IOA techniques, such as Structural Path Analysis (SPA) and mixed-unit input-output (MUIO) models [19], can be employed to evaluate the effects of price variation in the calculated environmental impact results, but they are not in the scope of the current work, and more work can be done in this area in the future.

Nevertheless, the potential environmental impacts resulting from price variations can be qualitatively inferred by comparing the industry-specific prices to the national-average electricity price. Theoretically, for the same dollar spent on electricity purchase, a customer that pays x times the national average electricity price should contribute to electricity consumption of 1/x times the national-average. For example, if an industrial customer pays only 68% (=$0.0488/$0.072) of the economy-wide average price, for the same dollar spent on electricity, this industrial customer would contribute to 1.48 (=1/(68%)) times more electricity consumption than the results obtained from the basic EIO-LCA model, where the electricity/$ factor is the same for all sectors in the economy. Similarly, for a commercial customer that pays 108% (=$0.0782/$0.072) times the economy-wide average price, for each dollar spent on electricity, the commercial customer contributes to only 92% (=1/(1.08%)) of the electricity consumption calculated in the basic US IO model. Likewise, a residential customer pays 117% (=$0.0844/$0.072) times the economy-wide average electricity will contribute to only 85% of the electricity consumption calculated by the basic EIO-LCA model. In summary, the amount of electricity consumed by an average residential customer is potentially overestimated by approximately 15%, an industrial customer is on average potentially underestimated by 48%, while a commercial customer is roughly overestimated by 8%. Such effects may accumulate positively or negatively (accumulate in one direction or canceled out) through the layers in the supply chain. A preliminary study by Weber using a MUIO model shows that on a total supply chain basis, manufacturing sectors' total supply chain footprint can be underestimated by 15%, while the service sectors' total footprint can be underestimated by 2-3% on average [19]. From the consumer's perspective, their carbon footprint attributed to furnishings and household equipment can be 16% higher while the footprint attributed to utilities are 12% lower if price variation is considered [19].

Regionalization Assumption

Because IO tables are not available at the sub-national level, regionalization techniques are employed in the construction of the MRIO model, and they inevitably contribute to an additional source of uncertainty in the analysis. The LQ and ERM regionalization techniques utilize proxy to infer the inter-industry transactions in the regions at the sub-national level by allocating inputs to suppliers in certain regions. Depending on allocation assumptions that LQ and ERM techniques are based, the model may assign more or less inputs to suppliers in different regions than in the real world. Similar uncertainty exists for the California household expenditure vector, which also uses LQ to fill in gaps in survey-data. These regionalization uncertainties are difficult to quantify without validation using survey-based data, which in themselves have other data-related uncertainties associated with them.

The underlying assumption of the LQ method is that if a region's production of a certain commodity is equal to or greater than the region's demand for that commodity, the region is self-sustaining and will use supplies produced within the region to meet its own demand. This assumption essentially ignores consumer preference for commodity (goods or services) produced in certain region due to specific preferences for taste, brand name, or other factors. For example, using the LQ method, California's supplies of wines exceed California industry sectors' demands for wines, hence it is inferred that California sectors do not purchase wines from the RUS region under the LQ assumption. However in reality, some California businesses or consumers may prefer wines produced in Oregon, but such transregional purchase is not being accounted in the model. Another example is hotel accommodation. LQ method assumes that California sectors purchase no hotel accommodation from RUS because California suppliers of hotel exceed the demands by California sectors. However, a non-trivial amount of hotel accommodation purchases by California businesses may in fact be for out-of-state business trips, and California consumers may prefer vacationing in non-California locale such as Las Vegas, resulting in little purchases of hotel accommodation in its home state. The LQ assumption may be reasonable for commodities that are mostly indistinguishable in their regions of origin (e.g. sugar, paper, coke) but it may be misleading for commodities with distinct consumer preferences for regions of production.

As the result of the LQ assumption, the MRIO model would overestimate the amount of supply chain carbon footprints occurred in California. It is difficult to quantify the amount of overestimation and its full supply effects without survey data. However, a study conducted by Burress that compared economic activities obtained by survey data with estimations by LQ found that the LQ assumption overstated the economic activities in Kansas by approximately 20% [20].

On the other hand, the ERM method assumes that sectors in the RUS region obtain input supplies from California in proportion to the shares of California production with respect to RUS. While California goods' market penetration may not be so evenly distributed throughout the US as the ERM method assumes, and there may be sub-regional differences in California good's penetration in the broader RUS region (e.g. although ERM method indicates that 19.4% of petroleum refineries outputs consumed by RUS sectors came from California, due to transportation distance and pipeline network, Florida may actually get a higher share of gasoline from Louisiana refineries, while Utah may get a higher share of gasoline from California than from Louisiana), the uncertainty resulted from this assumption is likely to average out over the supply chain and even-out within sub-regions as the RUS region represents a conglomeration of 49 diverse states.

Variation in Environmental Intensity

Another source of uncertainty in the MRIO model came from the assumption that technology and environmental intensity of production in the different regions are similar to US's national average. It does not account for the differences in intensities between different producers in the US or abroad. Variations in environmental intensity can be due to differences in production technologies, electricity generation mix in the grid, emission control technologies, environmental regulations, and producer price and currency valuation (for which the environmental intensity per monetary unit factors are based on) [2].

With increasing globalization of the supply chain of consumer products and a growing amount of imports from developing nations, this limitation of IO models inevitably becomes increasingly more important, and it has implications for the treatment of the ROW region, which is a conglomerate of all the US's trading partner nations. (Detailed discussions on uncertainties in the treatment of ROW region in MRIO models can be found in [12, 21]) For example, toys manufactured in China would use a mix of electricity supplies in the local grid, which could have environmental impacts much different than modeled here. Weber and Matthews [22] found that the CO2 intensities of food manufactured outside of US range from 30% to 300% of the US intensities, depending on the origination country, and the intensities of foreign mining range from 5% to 220% for CO2. They conducted a sensitivity analysis by assuming the ROW is represented by the most CO2-intensive and least CO2-intensive countries and found that such variation can result in 20% difference of total embodied emissions of CO2 for households [22]. In a multi-directional MRIO model consisting of several European nations and a ROW region, Lenzen et al. found that the total CO2 multiplier (kg/US$) of utilities (including electricity, gas, and district heat) can range from 0.16 for Norway to 11.29 for Germany, while the ROW is 9.31 on a global average [21]. Williams et al. also found that the energy intensities for making steel in China can be more than 8 times higher than in US [2]. For goods that have significant shares of import from environmentally intensive nations, the life cycle environmental impacts from the ROW region will be higher than the results presented here.

For this project, the energy and emissions factors needed for the model were input-output model multipliers on an "effect per million dollar basis". As all core effects of an IO model are in dollars, these multipliers convert economic results into effects such as energy used or GHG emissions.

The MRIO model described above requires energy and GHG databases for the California, Rest of US, and Rest of World submodels. For the Rest of US portion of the model, the energy and GHG emissions are used with permission from the 2002 Benchmark Input Output Model of the US Economy of the EIO-LCA project at Carnegie Mellon (described below). Likewise the Rest of World model uses these same energy and emissions factors as applied to its 426 sectors. Finally, the California portion of the MRIO model borrows heavily from the core US model with some adjustments as described below.

Energy and GHG Emissions Factors of the United States

The energy data used in the 2002 US Benchmark Commodity by Industry EIO-LCA model is derived from several additional sources, generally for three aggregated sectors (minerals, manufacturing, transportation). The energy/fuel data are also the main required underlying data sources to estimate GHG emissions for the sectors.

For the 11 mineral sectors (whose first 3 digits start with 211-213), the 2002 Fuel and Electric Energy Report published by the U.S. Census Bureau [Census 2002b] was used. This document reports fuel and electricity usage in physical units (e.g., short ton, barrel, cubic feet, gallon and kWh) as well as in some cases economic expenditures for the mineral sectors in 2002. Fuels presented in this report include electricity, coal, natural gas, and various petroleum-based fuels, which we again aggregate into the fuels listed above. Sectoral fuel use was calculated in terajoules (TJ) using the conversion factors shown in Appendix A.1 (Table A-1).

For the 14 agricultural sectors (sectors whose first 3 digits start with 111 and 112) the 2002 Census of Agriculture, specifically Table 59 was used [USDA 2002]. This document reports fuels as one category, "gasoline, fuels, and oils" and electricity usage in terms of expenditure by each of the NAICS codes included in the table. The 1997 Census of Agriculture included more detailed fuel expenditure information listing four fuel categories: gasoline and gasohol, diesel, natural gas, and LPG, fuel oil, kerosene, motor oil, grease, etc [USDA 1997]. The 1997 allocation of fuels within each sector was used to disaggregate the “gasoline, fuels, and oils” category within the 2002 Census. Expenditures were converted into physical units using values presented in Appendix A.1 (Table A-2).

Manufacturing Sectors

The electricity and fuel use for manufacturing sectors (representing 279 of the 426 sectors in the model) were estimated using data from the 2002 Manufacturing Energy Consumption Survey (MECS) [EIA 2006]. This report presents fuel and electricity usage in trillion BTU, in 3 to 6 digit NAICS forms with physical units of BTU. Note that the specific MECS data required is for non-feedstock use of energy and fuels; as an example we do not consider feedstock use of petroleum for making plastics to be a use of petroleum.

In the case of the detailed fuel data estimates, they were allocated from the 3-digit to 6-digit sector level by considering the dollar purchases of the fuels of each commodity sector in the model from the relevant industry sectors (i.e., from the 2002 US Benchmark IO Use Table). This assumption implicitly presumes that sectors within an aggregate industry sector have similar costs of energy. For example, in the 311 Food sector, if sector 311111 represented 90% of the dollar purchases of all the sectors beginning with 311 from power generation and supply in the use table, then 90% of the electricity use would be allocated to sector 311111.

Transportation Sectors

Energy use of the 11 transportation sectors was estimated using data from the use table as well as Transportation Energy Data Book (edition 26), published by the U.S. Department of Energy [USDOE 2007, Table 2.5] which reports consumption of energy by fuel type and transportation mode in trillion BTUs for 2002. The modes include Highway (auto, motorcycle, bus, light truck, other truck) and Non-Highway modes (air, water, pipeline, and rail) of transportation. Fuels presented were gasoline, diesel fuel, liquefied petroleum gas, jet fuel, residual fuel oil, natural gas, and electricity. Energy use by automobiles, motorcycles, and light trucks (in orange below) were assumed to be out of scope and excluded since these vehicles are not generally used for production of goods and services (with the exception of corporate fleets used in service sectors, see “all other sectors” below).

All other sectors

The sectoral economic values of consumption of coal, electricity, and natural gas of the roughly 100 sectors not covered by the sources above were estimated from the purchases of electricity and fuels from the 2002 CxI Use table at the Detailed level and then divided by the wholesale prices listed in Appendix A.1 to estimate the resource use in physical units.

As a result of the indirect estimates of energy use from this method, the estimates for these sectors are thus more uncertain than the other sectors. For example, the coal purchased by the wholesale trade sector is listed at $4 million, which is then adjusted by the average cost for coal paid by electric utilities (not a specific value for the wholesale trade sector), then converted to a value of 3.3 trillion BTU.

The overall validation of the energy use data with national level and sectoral level totals is summarized in Appendix A.1.7.

Emissions of GHG Emissions by Sector

GHG emissions in general were estimated for the IO sectors based on either direct estimation of GHG emissions from fossil fuel combustion, or from other public EPA data on process GHG emissions for various GHG-intensive sectors where the emissions come from non-fossil combustion. The GHG emissions are separated into: CO2 emissions (fossil and process sources), methane (CH4), nitrous oxide (N2O), and hydrofluorocarbons (HFCs). Note that the latter three types of GHG emissions come largely from non-fossil combustion and thus are not separated into fossil and process emissions. Appendix A.2 discusses how the estimates for each category were made, and apply to all sectors.

The primary fuel use and resulting greenhouse gas emissions for the power generation and supply sector in the California region of the model were adjusted to accurately portray the grid mix within the state rather than the national average values. The national average of 109 TJ per million dollars was maintained but a grid mix consisting of 50% natural gas, 27% nonfossil electric, 21% biomass and waste fuel, 1% petroleum, and 1% coal replaced the average national mix of fuels [eGRID 2007]. Changing the grid mix also changed the greenhouse gas emissions factors for power generation and supply based on the following factors from the EPA GHG Inventory; 94,453 metric tons per trillion Btu for coal, 53,056 metric tons per trillion Btu for natural gas, and 73,216 metric tons per trillion Btu for petroleum products and biomass/waste fuel (see Appendix A Table A-8 for further details). This leads to total carbon dioxide equivalent emissions of 5,086 metric tons of carbon dioxide equivalent per million dollars. This estimate for total greenhouse gas emissions is consistent with other literature estimates that suggest that the California electricity grid has about 60% of the carbon intensity of the national average [Masanet 2005].

Product selection

The final 22 products considered in this study are summarized in

Table 3. These products were ultimately selected by CARB, based on a preliminary short list of interesting product case studies proposed by the research team. The research team identified its preliminary short list of candidate products based on five primary screening criteria. Each criterion is listed and discussed briefly below.

1. The product should have significant total life-cycle GHG emissions. Since the team can only focus on 22 products in detail, it gave priority to products that are likely to be major contributors to one's retail product carbon footprint (e.g., personal computers and beef and dairy products) or that come from product categories that are major factors in one's retail carbon footprint (e.g., food and eating out).

2. There must be credible data available for the product to facilitate the life-cycle modeling, improvement potentials assessments, and scenario projections objectives of our study. To perform the detailed assessments of 22 products, the team needs to use existing case studies, data, and models. So the team only chose products for which such detailed assessments are practical given existing resources.

3. The product should be from a sector for which there is a significant manufacturing and/or design/management presence in California. Priority is given to industries that are important from both economic and GHG emissions perspectives in California, and that will continue to be so over the near-term (i.e., through 2020). This criterion helped to ensure that the chosen products would be relevant to California's economy and GHG emissions reduction targets, both now and in the future.

4. The final list of products should represent a diverse mix of products and supply chain characteristics. This criterion helped ensure breadth of product coverage, so that the team could explore how GHG emissions footprints and reduction potentials might vary across product classes.

5. There should be a few products for which consumption by California residents is growing. Such products are likely to be of greater GHG emissions importance to California in the future (e.g., flat panel televisions).

Of these five criteria, the first two (significance of emissions and data availability) were given the greatest priority in the initial screening process. The final three criteria were then used to fine tune the preliminary short list.

The full methodology for arriving at the 22 product in

Table 3 is summarized in Appendix C.

Table 3: 22 products selected by CARB

Industry/sector	Product	Industry/sector	Product
Apparel	Men's dress shirt	Food	Beef
Appliances	CFL		Bread
	Refrigerator		Canned tomatoes
	Water heater		Cheese
Beverages	Beer		Milk
	Soft drink		Chicken
	Wine		Tortillas
Chemicals	Paint	Forestry	Paper towels
Commercial	Restaurant		Wooden cabinet
Electronics	Flat panel TV	Minerals	Masonry cement
	Hard disk drive		
	Personal computer		

Baseline case analysis

This section provides a concise overview of the major assumptions associated with the baseline analysis depicted in Figure 2, which was described in the Methods section. Many of the assumptions summarized in this section are based on in-depth data analysis and synthesis, the details of which are described in the appendices. The reader is referred to the relevant appendix (and table) for further details as appropriate. Furthermore, the descriptions below are indexed to the relevant numerical labels in Figure 2 for ease of interpretation; indices are indicated in brackets below.

Production phase [1, 2, 3, 4]

The first step in estimating the production chain energy use and GHG emissions of each product was to derive unit-level price conversions, expressed in $2002 producer prices. This conversion is necessary to ensure compatibility with the 2002 IO matrix employed in the California MRIO model, and also to convert between annual units purchased and consumer expenditures depending on which type of data were available to estimate annual purchases of products in California for a given product.

Table 4 summarizes the product units selected in this study, as well as the $2002 producer prices that were derived for each product based on estimated 2011 purchases. Detailed derivations are provided in Appendix E, Table E-1. The values in Table 4 were used to estimate total 2011 purchases of each of the 22 products, which are summarized in Table 5. Detailed derivations for total annual purchases are provided in Appendix E, Table E-2.

Table 4: Producer price (2002) to product unit conversions

Industry/sector	Product	Unit	$ 2002 per unit
Apparel	Men's dress shirt	1 dress shirt	$28
Appliances	CFL	1 15w CFL	$1.72
	Refrigerator	1 refrigerator	$548
	Water heater	Gas fired tank water heater	$191
Beverages	Beer	12 oz. bottle	$0.36
	Soft drink	16 oz. bottle	$0.32
	Wine	750 ml bottle	$2.98
Chemicals	Paint	1 gallon	$11.14
Commercial	Restaurant	$ 2002	n/a
Electronics	Flat panel TV	1 LCD TV	$826
	Hard disk drive	1 external drive	$119
	Personal computer	1 control unit	$1225
Food	Beef	1 kg	$2.87
	Bread	1 kg	$1.51
	Canned tomatoes	0.5 liter can	$0.40
	Cheese	1 kg	$3.31
	Milk	1 gallon	$2.04
	Chicken	1 kg	$1.90
	Tortillas	1 kg	$1.05
Forestry	Paper towels	1 kg	$1.76
	Wooden cabinet	1 cabinet	$131
Minerals	Masonry cement	Metric ton	$108

Table 5: Estimated 2011 California retail purchases

Industry/sector	Product	Annual California purchases (in $2002 producer prices)
Apparel	Men's dress shirt	$152 million
Appliances	CFL	$8.1 million
	Refrigerator	$526 million
	Water heater	$126 million
Beverages	Beer	$259 million
	Soft drink	$345 million
	Wine	$1.3 billion
Chemicals	Paint	$61 million
Commercial	Restaurant	$27.7 billion
Electronics	Flat panel TV	$2.2 billion
	Hard disk drive	$53 million
	Personal computer	$2.9 billion
Food	Beef	$1.8 billion
	Bread	$201 million
	Canned tomatoes	$170 million
	Cheese	$449 million
	Milk	$953 million
	Chicken	$671 million
	Tortillas	$358 million
Forestry	Paper towels	$77 million
	Wooden cabinet	$850 million
Minerals	Masonry cement	$56 million

Factory to retail transportation phase [5, 6, 7]

Factory to retail energy use and GHG emissions were estimated in a two-step process. First, mass conversion factors were derived for each of the 22 products (except for restaurants, the unit for which is a dollar spent), which are summarized in Table 6. Detailed derivations are summarized in Appendix E, Table E-3. Second, based on the total estimated mass of annual shipments for each product, the transportation mode-distance intensity factors summarized in Table 7 were used in conjunction with estimated modes and transport distances by product for both domestically-sourced and imported products.

The factory-to-retail transportation energy and GHG emissions for domestically-produced produced products were estimated using U.S. Census Bureau's 2002 Commodity Flow Survey (U.S. Census 2002b), which contains data on modes, average distances, and intensities for key U.S. commodities, and average energy and GHG emissions intensities for U.S. trucking, rail, and water transportation from the U.S. Life

Cycle Inventory Database (NREL 2011). The relevant Commodity Flow Survey data are summarized in Table 8, and the average intensities from the US LCI Database are summarized in Table 7.

Table 6: Product mass conversions

Industry/sector	Product	Unit	Unit mass (kg)
Apparel	Men's dress shirt	1 dress shirt	0.3
Appliances	CFL	1 15w CFL	0.14
	Refrigerator	1 refrigerator	133
	Water heater	Gas fired tank water heater	53
Beverages	Beer	12 oz. bottle	0.55
	Soft drink	16 oz. bottle	0.4
	Wine	750 ml bottle	1.5
Chemicals	Paint	1 gallon	4.1
Commercial	Restaurant	$ 2002	n/a
Electronics	Flat panel TV	1 LCD TV	9.4
	Hard disk drive	1 external drive	1
	Personal computer	1 control unit	9
Food	Beef	1 kg	1
	Bread	1 kg	1
	Canned tomatoes	0.5 liter can	0.43
	Cheese	1 kg	1
	Milk	1 gallon	3.8
	Chicken	1 kg	1
	Tortillas	1 kg	1
Forestry	Paper towels	1 kg	1
	Wooden cabinet	1 cabinet	23
Minerals	Masonry cement	Metric ton	1000

Table 7: Energy and GHG emissions intensities for common freight modes

Mode	Energy (MJ/t-km)	CO2 (kg/t-km)
Transport, barge, diesel powered	3.49E-01	2.81E-02
Transport, combination truck, diesel powered	9.90E-01	7.99E-02
Transport, ocean freighter, residual fuel oil powered	2.06E-01	1.60E-02
Transport, train, diesel powered	2.36E-01	1.89E-02

Source: NREL (2011)

The factory-to-retail energy use and emissions associated with imported products were estimated by identifying major producing countries of imports by product, estimating the tonnage shipped and shipping distances between source country and U.S. ports, and using the energy use and emissions intensities for freight modes in Table 7. The fraction of product purchases that are assumed to be met by imports was determined from the "rest of world" purchases estimated by the MRIO model for the final manufacturing sector of each of the 22 products. Source countries for food products were obtained from the FAO TradeSTAT database (FAO 2011), which provides such data for a wide range of food commodity global trade flows on an annual basis. Source countries for all other products were estimated using the U.S. Census Bureau foreign trade statistics. For imports by sea, the distance between source country and U.S. ports was estimated using global port-to-port data from the U.S. National Imagery and Mapping Agency (NIMA 2001). For land imports (e.g., from Canada or Mexico) were estimated based on land distances to California from major shipping ports using Google Maps. Finally, the energy use and emissions intensity factors in Table 7 were coupled with the source country and distance data to arrive at the estimates summarized in Table 9.

Table 8: U.S. commodity flow data for domestic shipments

Industry/sector	Product	Truck		Rail		Water		Truck and rail		Truck and water	
		10^3 t	10^6 t-km	10^3 t	10^6 t-km	10^3 t	10^6 t-km	10^3 t	10^6 t-km	10^3 t	10^6 t-km
Apparel	Men's dress shirt	931	1,387	-	-	-	-	-	-	13	19
Appliances	CFL	642	873	-	-	-	-	-	-	-	-
	Refrigerator	1,838	2,412	-	-	-	-	18	51	-	-
	Water heater	1,838	2,412	-	-	-	-	18	51	-	-
Beverages	Beer	36,567	23,204	1,368	3,107	-	-	1,884	5,444	143	919
	Soft drink	36,567	23,204	1,368	3,107	-	-	1,884	5,444	143	919
	Wine	36,567	23,204	1,368	3,107	-	-	1,884	5,444	143	919
Chemicals	Paint	67,183	47,566	5,052	6,842	-	-	2,976	6,914	112	420
Commercial	Restaurant	-	-	-	-	-	-	-	-	-	-
Electronics	Flat panel TV	1,838	2,412	-	-	-	-	18	51	-	-
	Hard disk drive	1,838	2,412	-	-	-	-	18	51	-	-
	Personal computer	1,838	2,412	-	-	-	-	18	51	-	-
Food	Beef	57,517	47,690	503	1,439	-	-	396	1,002	191	362
	Bread	71,220	43,487	7,116	11,812	-	-	3,728	6,364	51	217
	Canned tomatoes	12,876	10,010	2,144	3,272	-	-	961	2,168	30	18
	Cheese	12,876	10,010	2,144	3,272	-	-	961	2,168	30	18
	Milk	12,876	10,010	2,144	3,272	-	-	961	2,168	30	18
	Chicken	57,517	47,690	503	1,439	-	-	396	1,002	191	362
	Tortillas	71,220	43,487	7,116	11,812	-	-	3,728	6,364	51	217
Forestry	Paper towels	45,874	24,647	632	1,020	324	35	859	1,792	60	141
	Wooden cabinet	150,902	59,653	12,683	27,027	-	-	6,214	15,611	-	854
Minerals	Masonry cement	20,532	5,260	2,233	2,366	-	-	244	341	-	-

Table 9: Energy and GHG emissions intensity estimates for imports

Industry/sector	Product	Top origin countries for imports					Energy (MJ/kg)	CO2 emissions (kg CO2e/kg)
Apparel	Men’s dress shirt	China	Canada	Mexico	Japan	Germany	2.3	0.19
Appliances	CFL	China	India	Korea			2.1	0.17
	Refrigerator	Italy	Mexico	China	Germany		1.7	0.13
	Water heater	China	Mexico	Korea	Canada		2.5	0.2
Beverages	Beer	Mexico	Netherlands	Canada	Germany	Belgium	2.5	0.2
	Soft drink	Austria	Switzerland	Canada	Mexico	Netherlands	2.9	0.25
	Wine	Italy	Australia	France	Argentina	Chile	1.8	0.14
Chemicals	Paint	Canada	Germany	Japan	UK	China	1.8	0.14
Electronics	Flat panel TV	Mexico	China	Japan	Taiwan	Korea, South	3.4	0.27
	Hard disk drive	China	Thailand	Singapore	Malaysia		2.3	0.18
	Personal computer	China	Mexico	Malaysia	Japan	Taiwan	2.7	0.21
Food	Beef	Canada	Mexico				0.91	0.07
	Bread	Mexico	Canada	Germany	India	China	0.76	0.06
	Canned tomatoes	Canada	Turkey	Israel	Mexico		1.88	0.15
	Cheese	Italy	New Zealand	France	Argentina	Netherlands	1.5	0.12
	Milk	Canada	Mexico	Greece	France	UK	1.7	0.13
	Chicken	Canada	Chile	Israel			2.0	0.16
	Tortillas	Mexico	Canada				1.9	0.15
Forestry	Paper towels	Canada	Germany	Finland	China		1.85	0.15
	Wooden cabinet	China	Vietnam	Canada	Mexico	Malaysia	2.1	0.17
Minerals	Masonry cement	Canada	China	Korea	Mexico		1.62	0.13

Product use phase [8, 9]

Data on installed stocks of energy using products were derived using two primary approaches. For direct users of energy (e.g., appliances), the research team relied on the most recent California Residential Appliance Saturation Survey (RASS) (KEMA 2010). The RASS provides statewide estimates on the installed stocks and UECs of all major residential appliances, including all direct energy using products considered in this study. For indirect users of energy, which are defined as products whose normal use requires the operation of energy using devices (e.g., clothes washers and dryers for the men's dress shirt and refrigeration for milk), estimates were made based on assumed lifetimes and per-product indirect energy requirements. The use phase assumptions for each relevant product are summarized in Table 10. The detailed derivations of these assumptions are summarized in Appendix E, Table E-4.

Table 10: Baseline per-product UEC data

Industry/sector	Product	Unit	UEC
Apparel	Men's dress shirt*	1 dress shirt	2 kWh/yr
			0.1 therms/yr
Appliances	CFL	1 15w CFL	16.4 kWh/yr
	Refrigerator	1 refrigerator	772 kWh/yr
	Water heater	Gas fired tank water heater	195 therms/yr
Beverages	Beer*	12 oz. bottle	0.04 kWh/yr
	Soft drink*	16 oz. bottle	0.06 kWh/yr
	Wine*	750 ml bottle	0.1 kWh/yr
Electronics	Flat panel TV	1 LCD TV	290 kWh/yr
	Hard disk drive	1 external drive	11 kWh/yr
	Personal computer	1 control unit	380 kWh/yr
Food	Beef*	1 kg	0.05 kWh/yr
	Cheese*	1 kg	0.05 kWh/yr
	Milk*	1 gallon	0.2 kWh/yr
	Chicken*	1 kg	0.05 kWh/yr

* = indirect consumers of energy

To convert from UECs to annual GHG emissions, the emissions factors in Table 11 were employed in the California MRIO LCA model. These factors were derived in Masanet et al. (2009a).

Table 11: Use phase emission factors

Fuel	Emission factor
Electricity	0.40 kg CO2e/kWh
Natural gas	5.92 kg CO2e/therm

End of life phase [10, 11]

The end of life phase module requires estimates of the annual mass of product discards in California each year, product mass composition, and the mass fractions for each of the three disposition paths considered in the model (landfill, compost, and recycle). Detailed derivations of these parameters for each product are summarized in Appendix E, Table E-5.

To estimated landfill energy use and emissions, a California-specific landfill model developed in (Masanet et al. 2005) was employed in this study. The landfill model estimates the energy use and GHG emissions associated with the disposal of both biodegradable and non-biodegradable waste in California landfills, based on recent California landfill profile data. For biodegradable products, the method estimates the average amount of electricity generated from energy recovery landfills as well as fugitive emissions of landfill methane generation. Appendix D contains the detailed calculations and data sources behind this method; Table 12 summarizes the resulting average energy use and emission factors employed in the California MRIO LCA model.

Table 12: Landfill modeling methodology summary

End of life aspect/value		Applicability/notes
Landfill disposal		For all mass sent to landfill
Energy use	0.6 MJ/kg waste	Includes collection, transport, and landfill equipment operations
GHG emissions	0.05 kg CO2e/kg waste	
Landfill gas generation, emissions, and use		Additional energy use and GHG emissions associated with biodegradable portion of mass sent to landfill
Energy use	-0.7 MJ/kg waste	Electricity generated from landfill gas
GHG emissions	1.1 kg CO2e/kg waste	Average net emissions

To estimate the GHG emissions “credits” associated with materials recycling and composting, factors from the U.S. EPA’s Waste Reduction Model (WARM) were employed in this study (U.S. EPA 2010a). These factors are summarized in

Table 13 and elaborated upon in Appendix E, Table E-5.

Table 13: Recycling and compost factors from WARM and other sources

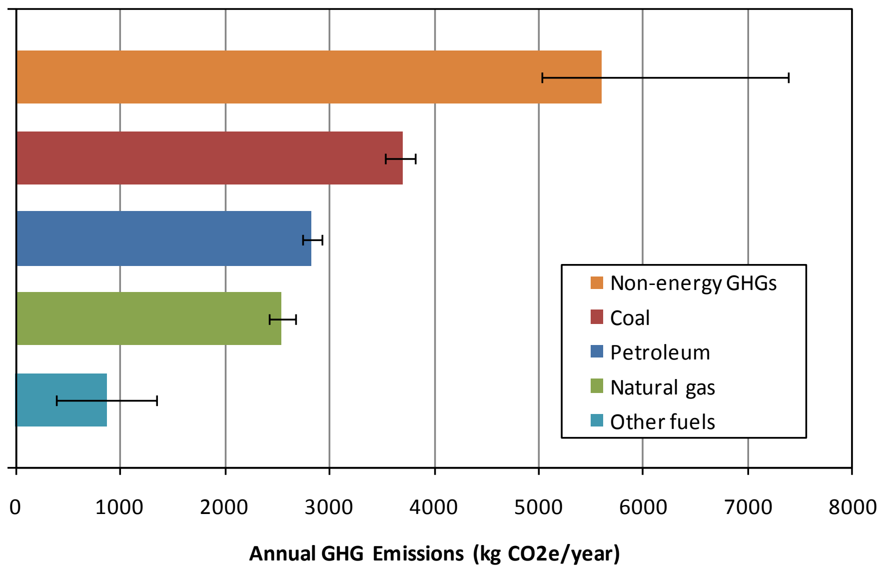
Material	GHG Emissions per Short Ton of Material Recycled (MTCO2E)	GHG Emissions per Short Ton of Material Composted (MTCO2E)	Source(s)
Steel Cans	(1.80)	n/a	EPA (2010a)
Glass	(0.28)	n/a	EPA (2010a)
HDPE	(1.38)	n/a	EPA (2010a)
PET	(1.52)	n/a	EPA (2010a)
Dimensional Lumber	(2.46)	n/a	EPA (2010a)
Food Scraps	n/a	(0.20)	EPA (2010a)
Personal Computers	(2.26)	n/a	EPA (2010a)
Refrigerators	(2.45)	n/a	EPA (2010a), ISIS (2007)
Flat panel TVs	(1.38)	n/a	EPA (2010a), IVF (2007)
Water heaters	(1.99)	n/a	EPA (2010a), Lu et al.(2011)
CFLs	(1.50)	n/a	EPA (2010a), VITO (2009)

Uncertainty assessment

Clearly, the California MRIO LCA modeling method described above relies on many different data from a diversity of different sources. Thus, the uncertainties associated with the method are likely to be significant. In an ideal world, all data sources used in the methodology would provide compatible uncertainty information, and this information would be used for a robust uncertainty analysis of the model's results. In reality, such uncertainty data are rarely available, especially given the diversity of data sources employed in this study.

To date, the only published study that has attempted to quantify the uncertainties associated with hybrid IO LCA models for carbon footprinting is Masanet et al. (2009a). Thus, the research team employed the approach and data assumptions of the Masanet et al. (2009a) study to at least partially estimate the parameter uncertainty associated with the California MRIO LCA modeling results for the baseline case of each of the 22 products. The methodology considers parameter uncertainties associated with: (1) use phase energy consumption and emissions, (2) fuel coefficient uncertainties in the MRIO model, and (3) non-energy GHG emissions coefficient uncertainties in the MRIO model.

Although less than comprehensive, the preliminary parameter uncertainty estimates employed in this project provide at least some idea of the minimum parameter uncertainty associated with the model's estimates. Figure 5 summarizes the uncertainty estimates for the IO modeling results in the Masanet et al. (2009a) study, which suggest that even partial parameter uncertainties are significant in the modeling framework, especially for non-energy GHG emissions. This project did not address modeling uncertainty, however, which is another key source of uncertainty inherent in the MRIO-based method.



Source: Masanet et al. (2009a)

Figure 5: Estimated annual supply chain GHG emissions per California household by source category

Use phase energy consumption and emissions uncertainty

The GHG emission factor for electricity was based on information from Marnay et al. (2002), which presented fuel data for electricity generation and estimates for average carbon intensity of California electricity (including imported electricity) from three different models. The fuel data from Marnay et al. (2002) were coupled with average GHG emission factors by fuel from CARB's California GHG emissions inventory.

However, no uncertainty data for the California GHG emissions inventory estimates for electricity generation could be found in the public domain. Thus, the research team estimated 95% uncertainty ranges for electricity generated from different fuel types based on data from the Intergovernmental Panel on Climate Change's (IPPC's) GHG emission factor database and the U.S. EPA's national GHG emissions inventory.

The GHG emission factor for residential natural gas combustion in California was based on emission factors obtained from the California GHG emissions inventory. As for the GHG emission factors for electricity generation, no uncertainty data for the California GHG emissions inventory estimates for natural gas combustion could be found in the public domain. Thus, the research team estimated 95% uncertainty ranges for residential natural gas combustion based on data from the IPPC's GHG emission factor database (IPCC 2008) and the U.S. national GHG emissions inventory (U.S. EPA 2008a).

The resulting 95% confidence intervals on the average emission factors are listed in Table 14.

Table 14: Estimated uncertainty ranges for UEC emission coefficients

Emission factor	Unit	Value	95% Confidence Interval	
			Lower	Upper
Electricity	0.40 kg CO2e/kWh	0.40	0.38	0.44
Natural gas	5.92 kg CO2e/therm	5.92	5.71	6.34

Masanet et al. (2009a) estimated confidence intervals for the RASS technology saturation and average end use UEC data used to estimate product UECs in this study. Ninety-five percent confidence intervals were estimated for each technology saturation assumption, based on survey sampling error estimates provided by KEMA-Xenergy et al. (2004) for the different sample populations in the RASS study. (These sample populations were based on California utility territories and metered versus non-metered households).

The RASS study did not explicitly estimate standard errors for its average end use UEC estimates. However, the regression analysis approach used by the RASS study team to estimate average end use UECs is analytically similar to the regression approach used by the U.S. Department of Energy to estimate end use UECs in its quadrennial U.S. Residential Energy Consumption Survey (RECS) (U.S. DOE 1983). Thus, the research team used published standard errors for average end use UECs from the 2001 RECS (U.S. DOE 2003) as proxies for RASS end use UEC standard errors in this project. The resulting 95% confidence intervals on the baseline UECs are listed in Table 15.

Table 15: Estimated uncertainty ranges for baseline UECs

Industry/sector	Product	Unit	UEC	95% Confidence Interval
Apparel	Men's dress shirt*	1 dress shirt	2 kWh/yr	+/- 8%
			0.1 therms/yr	+/- 15%
Appliances	CFL	1 15w CFL	16.4 kWh/yr	+/- 7%
	Refrigerator	1 refrigerator	772 kWh/yr	+/- 7%
	Water heater	Gas fired tank water heater	195 therms/yr	+/- 14%
Electronics	Flat panel TV	1 LCD TV	290 kWh/yr	+/- 7%
	Hard disk drive**	1 external drive	11 kWh/yr	+/- 12%
	Personal computer	1 control unit	380 kWh/yr	+/- 8%

* Uncertainty ranges are derived from RASS data for washers and dryers

** Uncertainty range is derived from RASS data for home office equipment

Fuel coefficient uncertainties in the MRIO model

The Masanet et al. (2009a) study also compiled parameter uncertainty information for data used to construct fuel and fuel end use coefficients in the 2002 U.S. national EIO-LCA model, when such uncertainty information existed. For the fuel coefficients, the Masanet et al. (2009a) study constructed 95% confidence intervals for the following fuels and IO sectors, which were used in this study:

- all fuels for the manufacturing IO sectors, based on survey standard error data from the 2002 MECS (U.S. DOE 2005)
- electricity and petroleum use for the construction IO sectors, based on survey standard error data from the 2002 U.S. Economic Census (U.S. Census Bureau 2005).

Non-energy GHG emissions coefficient uncertainties in the MRIO model

Based on Masanet et al. (2009a), this study utilizes U.S. EPA (2004) national inventory data that contains estimates of non-energy related GHG emissions from over forty different sources, along with 95% confidence intervals for each estimate. The estimated confidence intervals for many of these data are significant; for example, the range for methane emissions from landfills is +/-30%, the range for methane emissions from natural gas systems is +/-40%, and the range for process-related CO_2 emissions from iron and steel production is +78%/-58%. Such uncertainties are currently unavoidable given the state of measurement and estimation techniques for these GHG inventory data; however, they also represent important parameter uncertainties in the modeling framework of this study.

To construct 95% confidence intervals for non-energy GHG emissions in the supply chain model, the research team first compiled 95% confidence interval estimates from U.S. EPA (2004) for each important emissions source. Next, the research team mapped these uncertainties to MRIO sector-level non-energy GHG emission coefficients.

The resulting uncertainty estimates for the baseline case results are presented in the Results section of this study.

Low carbon technical potential case analysis

As summarized in Figure 3, five major life-cycle product opportunities were considered (as applicable) to estimate the low carbon technical potential for each of the 22 products: (1) supply chain energy efficiency and GHG emissions abatement measures; (2) final manufacturing energy efficiency and GHG emissions abatement measures; (3) product design measures, including considerations for materials and operational energy efficiency and their effects on the transportation, use, and disposal life-cycle phases; (4) transportation efficiency measures; and (5) end of life considerations.

For each opportunity category, the research team limited its estimates to verifiable best practice technologies and approaches so that the low carbon technical potentials would represent emissions reductions that are technologically feasible in the five-year projection period. Thus, the technical potential can be thought of as an upper bound of energy efficiency and GHG emissions reduction potential in a technical feasibility sense, regardless of cost or other limitations. However, the general framework discussed below could be employed with more aggressive technology assumptions to estimate low carbon potentials for technologies or approaches that are emerging or currently under development.

Similar to the baseline case analysis description, this section provides a concise overview of the major assumptions associated with analysis approach depicted in Figure 3, which was described in the Methods section. Many of the assumptions summarized in this section are based on in-depth data analysis and synthesis, the details of which are described in the appendices. The reader is referred to the relevant appendix (and table) for further details as appropriate.

Supply chain improvements

To estimate best practice supply chain technology improvement potentials, this study utilized the eSTEP modeling methodology that is summarized in Masanet et al. (2009a, 2009c). The model currently includes best practice technology energy savings data for a range of energy efficiency measures in different IO sectors, and for different energy end uses. It also contains key measures for non-energy related greenhouse gas emissions in several IO sectors. A summary of the broad IO sectors, fuels, and non-energy GHG emissions covered by best practice technology data in the eSTEP model is provided in Table 16.

The eSTEP model was used to generate potential reductions in fuel use and emissions for all manufacturing, commercial, agricultural, mining, and water treatment sectors in the California MRIO model as a means of approximating the potential supply chain emissions reductions a final manufacturer might drive throughout its supply chain by sourcing its inputs only from “low carbon” supply chain partners. As such, it provides an upper bound estimate on best practice supply chain emissions savings, since it assumes that best practices will be adopted at all sectors in its supply chain, whether those sectors are primary or very distant suppliers.

The analytical framework of eSTEP is illustrated schematically in Figure 6. For each of the 22 products, the estimated supply chain reductions by supply chain sector, fuel, emission, and end use are provided in Appendix F. For more information on the eSTEP methodology, the reader is referred to Masanet et al. (2009a).

Table 16: eSTEP measures by sector and fuel

Manufacturing (electricity, natural gas, coal, and petroleum)	
Conventional Boiler Use	Facility HVAC
CHP and/or Cogeneration Process	Facility Lighting
Process Heating	Onsite Transportation
Process Cooling and Refrigeration	Conventional Electricity Generation
Machine Drive	Other
Electro-Chemical Processes	
Commercial (electricity)	
Space Heating	Cooking
Cooling	Refrigeration
Ventilation	Office Equipment
Water Heating	Computers
Lighting	Other
Commercial (natural gas)	
Space Heating	Cooking
Water Heating	Other
Agriculture (electricity, natural gas, petroleum)	
Motors	Machinery
Lighting	Other
Onsite transport	
Water treatment (electricity)	
Pumping systems	Other
Mining (petroleum, electricity)	
Mining vehicles	Conveyors
Pumps	
Non-energy GHG emissions abatement	
Improved fertilizer management	Manure methane capture
Reduce leaks in natural gas distribution	Reduced/improved refinery flaring
Reduced SF6/PFC in electronics	Landfill gas recovery
Reduced SF6 emissions in power systems	

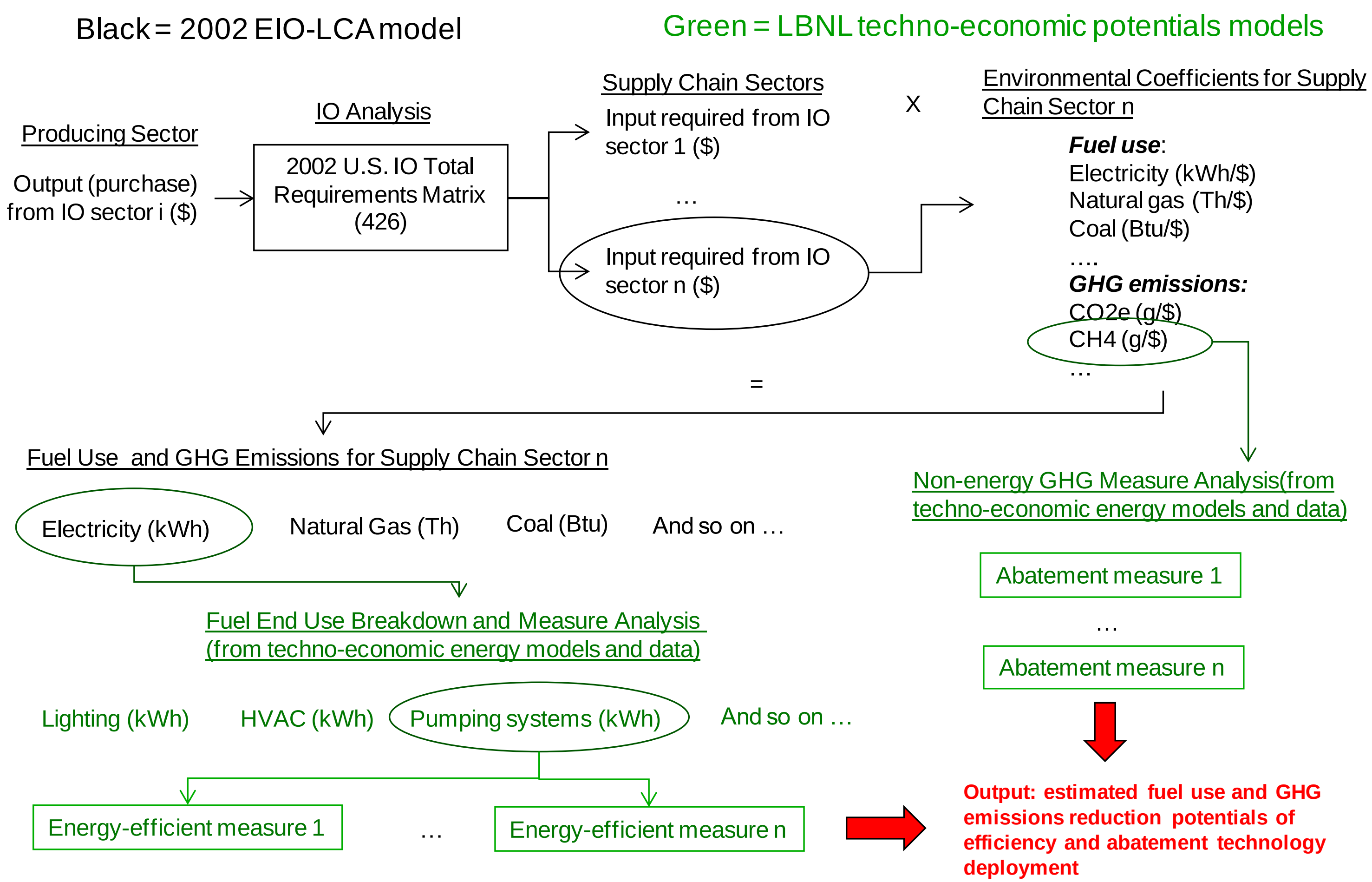


Figure 6: Schematic of the eSTEP supply chain model

Final manufacturing improvements: design

The design improvements considered in the low carbon technical potential cases were restricted to verifiable best practices in the literature. Still, for most of the 22 products considered in this study, viable design options were found to exist that hold potential for reducing produce life-cycle energy use and GHG emissions. For several energy using products, as expected best practice design was found to focus primarily on reducing product operational energy use. For such products, no other design considerations were considered given the dominance of use phase energy consumption in the product life-cycle footprint.

As illustrated in Figure 3, design changes can have the potential to reduce life-cycle energy use and GHG emissions at more than one phase. For example, reductions in materials mass can result in lower production, transportation, and end of life impacts. Such multi-phase effects were considered where appropriate in this study.

The major best practice design considerations that were included in this study are summarized in Table 17. A variety of best practice design considerations were identified, including materials mass reductions, increased use of recycled materials, new design configurations, design for recycling, and use phase energy efficiency improvement. More detailed discussion of each consideration, and the related analytical assumptions, are provided in Appendix E, Table E-7.

Final manufacturing improvements: operations

Because for many products the final manufacturing sector accounts for significant fractions of the production chain energy and GHG emissions footprints, the MRIO model was also augmented (when possible) with best practice energy efficiency and emissions intensities for the 22 final manufacturing IO sectors considered in this study using sector-specific case studies and data. Given that the eSTEP model is primarily focused on energy efficient technology improvements to cross-cutting systems (e.g., lighting, boilers, pumps, and motors), the research team took special care to ensure that the technology potentials estimated for each final manufacturer reflected the best practices identified in the literature. This was done not only to improve the comprehensiveness of the low carbon technical potential case analysis, but also because it is likely that final manufacturers who react to carbon labels and/or product standards will first look to their own facilities for energy use and GHG emissions reductions.

Table 18 summarizes the best practice energy efficiency reductions that were estimated for each of the final manufacturing sectors considered in this study. More detailed information on the derivation of these estimates is provided in Appendix E, Table E-6.

Table 17: Summary of product case assumptions for low-carbon design features

Industry/sector	Product	Low-carbon design features
Apparel	Men's dress shirt	- Increased use of recycled polymer fibers - Increased hydrophobic characteristics for reduced drying time
Appliances	CFL	- Maximal use phase energy efficiency
	Refrigerator	- Maximal use phase energy efficiency through improved controls, insulation, compressors, heat exchangers, and seals.
	Water heater	- Natural gas fired tankless design
Beverages	Beer	- Lightweight bottle - Increased recycled content
	Soft drink	- Increased recycled content
	Wine	- Lightweight bottle - Increased recycled content
Chemicals	Paint	n/a
Commercial	Restaurant	n/a
Electronics	Flat panel TV	- Maximal use phase energy efficiency - Design for recycling
	Hard disk drive	- Maximal use phase energy efficiency - Design for recycling
	Personal computer	- Maximal use phase energy efficiency - Design for recycling
Food	Beef	- Reduced packaging through vacuum packing - Reduced packaging through lightweight trays
	Bread	- "Right sized" loaves - Prominent "best by" date
	Canned tomatoes	- Lightweight cans
	Cheese	n/a
	Milk	- Reduced packaging mass
	Chicken	- Reduced packaging through vacuum packing - Reduced packaging through lightweight trays
	Tortillas	- "Right sized" loaves - Prominent "best by" date

Industry/sector	Product	Low-carbon design features
Forestry products	Paper towels	- Increased use of recycled fibers - Half-sheet rolls
	Wooden cabinet	- Design for take-back and refurbishment
Minerals	Masonry cement	n/a

Table 18: Estimated final manufacturing efficiency improvements

Industry/sector	Product	Estimated percent reduction	
		Electricity use	Fuel use
Apparel	Men's dress shirt	30%	25%
Appliances	CFL	25%	20%
	Refrigerator	25%	20%
	Water heater	25%	20%
Beverages	Beer	15%	25%
	Soft drink	15%	20%
	Wine	25%	25%
Chemicals	Paint	20%	20%
Electronics	Flat panel TV	25%	20%
	Hard disk drive	25%	20%
	Personal computer	25%	20%
Food	Beef	20%	30%
	Bread	20%	30%
	Canned tomatoes	30%	35%
	Cheese	25%	20%
	Milk	30%	30%
	Chicken	20%	30%
	Tortillas	20%	30%
Forestry	Paper towels	15%	20%
	Wooden cabinet	15%	20%
Minerals	Masonry cement	30%	5%

Transportation improvements

We assume that, on average, trucking fuel consumption can be reduced by 10% through the adoption of technologies for reduced drag and improved aerodynamics as estimated in (U.S. DOE 2006). This 10% reduction in fuel consumption was applied to all supply chain trucking fuel use in all regions of the MRIO model, as well as to factory-to-retail trucking transportation in the low carbon case.

Use phase improvements

Table 19 summarizes the estimated achievable reductions in operational energy use for those products that are direct consumers of energy in the use phase. For products that are covered by the ENERGY STAR labeling program, the research team identified the lowest energy consuming version of the case study product in the ENERGY STAR lists of approved products. For the remaining products, best practice estimates from other sources were employed. More detailed information on the derivation of these estimates is provided in Appendix E, Table E-8.

Table 19: Estimated reductions in use phase energy use

Industry/sector	Product	Estimated % reduction in operational energy use
Appliances	CFL	20%
	Refrigerator	30%
	Water heater	15%
Electronics	Flat panel TV	30%
	Hard disk drive	20%
	Personal computer	35%

End of life phase improvements

End of life phase improvements were considered in this study in cases where design changes would lead to changes to product bills of materials or mass reductions. End of life considerations, where applicable, are summarized along with the design derivations in Appendix E, Table E-7.

Scenario projections

Objective 3 of this project was to develop projections of potential GHG emissions reductions achievable under different scenarios over the period 2011 to 2015. A five year projection period was chosen to indicate potential emissions reductions that might be attained in the near term through carbon labels and life-cycle standards. Given that technologies change steadily over time, a five year projection period was also deemed to be reasonably credible for the best practice technologies that were considered in the low carbon technical potential cases. Longer-term projections periods would require more developed analyses of likely technological change and innovation pathways (see for example Masanet et al. 2009b).

Two bounding scenarios were chosen to capture a credible range of possible life-cycle GHG emissions in the near term for the 22 products under study. The first is the "business as usual (BAU)" scenario, which assumes that baseline per-product annual life-cycle GHG emissions (results for which are summarized for each product in Appendix F) will stay constant over the projection period, but increase in absolute fashion due to population growth. The population growth assumptions that were used to construct this scenario are summarized in Table 20. As indicated in Table 20, the BAU scenario assumes that the purchases, installed stock, and disposals of each product will increase by 1.25% per year based on recent California population projections from the California Department of Finance. As such, the BAU scenario is meant to serve as an upper bound on life-cycle GHG emissions over the projection period as it assumes no emissions reductions are achieved for the 22 products.

Table 20: Population growth assumptions

California population characteristic	Value	Source/notes
2010 total state population (persons)	37.3 million	U.S. Census (2011b)
2010 total state households	12.2 million	U.S. Census (2011b)
2010 average persons per household	2.91	U.S. Census (2011b)
Projected annual population growth rate (%)	1.25	California DOF (2011)
2015 projected state population (persons)	39.7 million	Derived based on above data
2015 projected state households	13.6 million	Derived based on above data

The second bounding scenario the "low carbon technical potential (TP) scenario," which estimates the maximum achievable GHG emissions reductions over the projection period. It does so by assuming that the California purchases and installed stocks of all 22 products would switch to the low carbon technical potential case (results for which are summarized in Appendix F) in 2011, and all years moving forward. The effects of population growth are included in this scenario, but the limitations stock turnover are not (i.e., all durable goods are instantly replaced by low carbon versions in 2011). As such, this scenario is meant to serve as an illustrative lower bound on life-cycle GHG emissions over the projection period as it assumes maximum possible emissions reductions are achieved for the 22 products.

In likelihood, even if low carbon versions of all 22 products were to appear in the marketplace immediately, the actual annual life-cycle GHG emissions associated with each product case would fall somewhere between the BAU and low carbon TP scenarios. This study considered three additional scenarios to explore different emissions trajectories considering: (1) the effects of natural stock turnover; (2) possible carbon label uptake by consumers; and (3) possible life-cycle emissions standards.

The "stock turnover constrained TP" scenario was developed to considers that durable goods (e.g., appliances) will likely only be replaced as they reach the end of their useful life, and therefore limit the pace at which low carbon versions of durable goods can penetrate the installed stock over the projection period. This scenario was constructed by assuming that all California purchases would switch to the low carbon technical case for each product, but that the installed stock of energy using durable goods (i.e., refrigerators) would change over at its natural retirement rate. The assumed lifetimes of energy using durable goods are summarized in Appendix E. As such, this scenario provides an upper bound estimate of possible emissions reductions over the projection period if stock turnover rates remain constant. Comparing the low carbon TP and stock turnover constrained TP scenarios sheds light on possible emissions savings that might be realized through policies and incentives for early retirement of current technologies.

The "benchmark market uptake" scenario estimates natural market uptake of the low carbon versions of the 22 products if such products displayed carbon labels. Given that carbon labeling initiatives are relatively new around the world, credible data on actual purchases of labeled products are currently lacking. Thus, the research team chose to adopt the market uptake rates of ENERGY STAR labeled products as the best available proxies for labeled products in the United States, given that the ENERGY STAR label enjoys high consumer brand recognition and that historical market penetration data for ENERGY STAR labeled products are available. Table 21 summarizes the 2010 market shares of different ENERGY STAR products (U.S. EPA 2011f). This scenario was constructed by assuming that all 22 products would bear carbon labels, and that the low carbon versions of each product would be purchased by California consumers at the same rates as the ENERGY STAR data in Table 21. As such, this scenario is meant to illustrate one possible emissions trajectory if consumers reacted to carbon labels as well as they do to the ENERGY STAR product label.

Table 21: 2010 ENERGY STAR market shares

Industry/sector	Product	ENERGY STAR® market share
Appliances	CFL	20%
	Refrigerator	50%
	Water heater	12%
Electronics	Flat panel TV	77%
	Personal computer	47%
ENERGY STAR average (all products)		44%

Source = U.S. EPA (2011f)

Lastly, the "life cycle product standards" scenario estimates the achievable GHG emissions reductions by product if maximum life-cycle emissions standards were to be set for each of the 22 products, and that these standards would determine eligibility for products to be sold in California. This scenario is meant to mirror California appliance standards, which have historically led to significant reductions in the state's energy use by limiting the choices of California consumers to appliances that achieve high levels of energy efficiency. The research team looked again to the ENERGY STAR program to determine a realistic life-cycle GHG emissions standard that sits somewhere between the baseline (i.e., current) case and the low carbon technical potential case for each product. Based on the approach used by the ENERGY STAR for Industry Program, in which industrial plants that are in the top quartile of energy performance nationwide can apply for the ENERGY STAR label, the research team chose to set the life-cycle GHG emissions of each product at 75% less than the baseline case and 25% more than the low carbon technical potential case for each product.

Results and Discussion

This section provides a summary of key results that are aimed at answering the research questions of this study, namely:

1. By how much might GHG emissions be reduced across the life-cycle of a given product if carbon labels and/or standards are successful in driving the market to best practice for low carbon and energy efficient life cycles?

2. Of the estimated emissions reductions, how much is likely to occur within California?

As mentioned in the Introduction, to keep the main body of this report concise, detailed results from the 2011 baseline and low carbon technical potential case analyses for each product are summarized in Appendix F. In total, over 150 different figures are presented in Appendix F which displays the case results for life-cycle energy use and GHG emissions by product, life-cycle phase, region, fuel type, emission type, sector, and IO sector. Figure 7 summarizes the 2011 baseline case results for the 22 products; the top half presents the baseline case results for annual GHG emissions by region and the bottom half presents the same results by product life cycle phase.

Clearly seen is the wide range in results between the considered products; the California product with the lowest estimated annual life-cycle GHG emissions (around 20,000 Mg CO2e/yr) is over two orders of magnitude lower than that with the highest estimated annual life cycle GHG emissions (restaurants, at roughly 13 million Mg CO2e/yr). However, the difference in annual expenditures on each of these two items by Californians is also over two orders of magnitude. As designed, the study was meant to explore the life-cycle emissions of a wide variety of product that are important to California for multiple environmental and economic reasons. However, the span suggests that future studies might consider using the California MRIO LCA model as a screening tool to identify products with large annual life-cycle GHG emissions profiles for more targeted analysis.

The highest ranking products in Figure 7 are those which are either significant consumers of use phase energy, or food-related items. Of the food-related items in this study, those that have significant animal inputs in their supply chain (beef, cheese, milk, and chicken) are seen to have much higher estimated life-cycle GHG emissions. These results are not surprising, given the large contributions that non-energy related GHG emissions in the life cycles of these products (e.g., from manure, fertilizers, and enteric fermentation). While it was not unexpected that these energy using and food-related products would appear at the top of the list, the results provide quantitative evidence of the GHG emissions that are generated by California consumers with purchasing these products, as well as how important they are from a GHG emissions perspective compared to other products purchased in the state.

A number of carbon labeling initiatives have been focused on food to date (e.g., see work by the Carbon Trust on various food products and initiatives for wine, breweries, and others (Sathaye et al. 2009)), and the first labeled product (a bag of Walker's potato chips) was also a food product. However, the results in Figure 7 suggest that certain food products might be far more important for California than others from a life-cycle GHG emissions perspective. However, the animal-related food products that rank high in the list also pose significant challenges for GHG emissions abatement, given the difficulties in measuring and managing emissions from land use, animal waste, and enteric fermentation (as opposed to energy efficiency at a food facility).

Perhaps more interesting is that restaurants were estimated to have the largest life-cycle GHG emissions among the expenditures considered in this study. As mentioned earlier, restaurants also represent the single largest annual expenditure by Californians in this study. However, the magnitude of the estimated restaurant emissions suggests that life-cycle GHG reduction efforts, which have almost exclusively focused on products to date, should also be expanded to consider the service industries as a target for potentially large reductions.

The regional results in the top half of Figure 7 suggest that the majority of estimated annual emissions occur in California. The “undefined” region was applied exclusively to end of life emissions that might occur outside of California (e.g., recycling), but the region for which could not be established or predicted. There are three major reasons for the predominance of emissions in California. First, the study deliberately chose products that were likely to have large production, use, or end of life emissions in California. Second, as explained in the Methods section, the LQ method used in the MRIO model might overestimate the production phase emissions occurring for some products that are produced in California. However, for the animal-related products that appear at the top of the list, there is indeed a large manufacturing presence in California and these products (beef, cheese, milk) are likely to be purchased locally. Third, as seen in the bottom half of Figure 7, in-state use phase emissions dominate the life-cycle of four of the top seven products. Interestingly, the results suggest that transport and end of life emissions are far less important than production and use phase emissions for the chosen products.

The results in Figure 7 convey the usefulness of the California MRIO LCA model for identifying potentially large sources of product-related GHG emissions, and further identifying which region and life-cycle stages are the most important for each product. As such, the model should prove useful in identifying products that should be further studied for potential life-cycle GHG emissions reductions that would benefit the state under different policy mechanisms. Although this project only analyzed 22 products, the California MRIO LCA model could be applied to as many as 423 different product and service sectors.

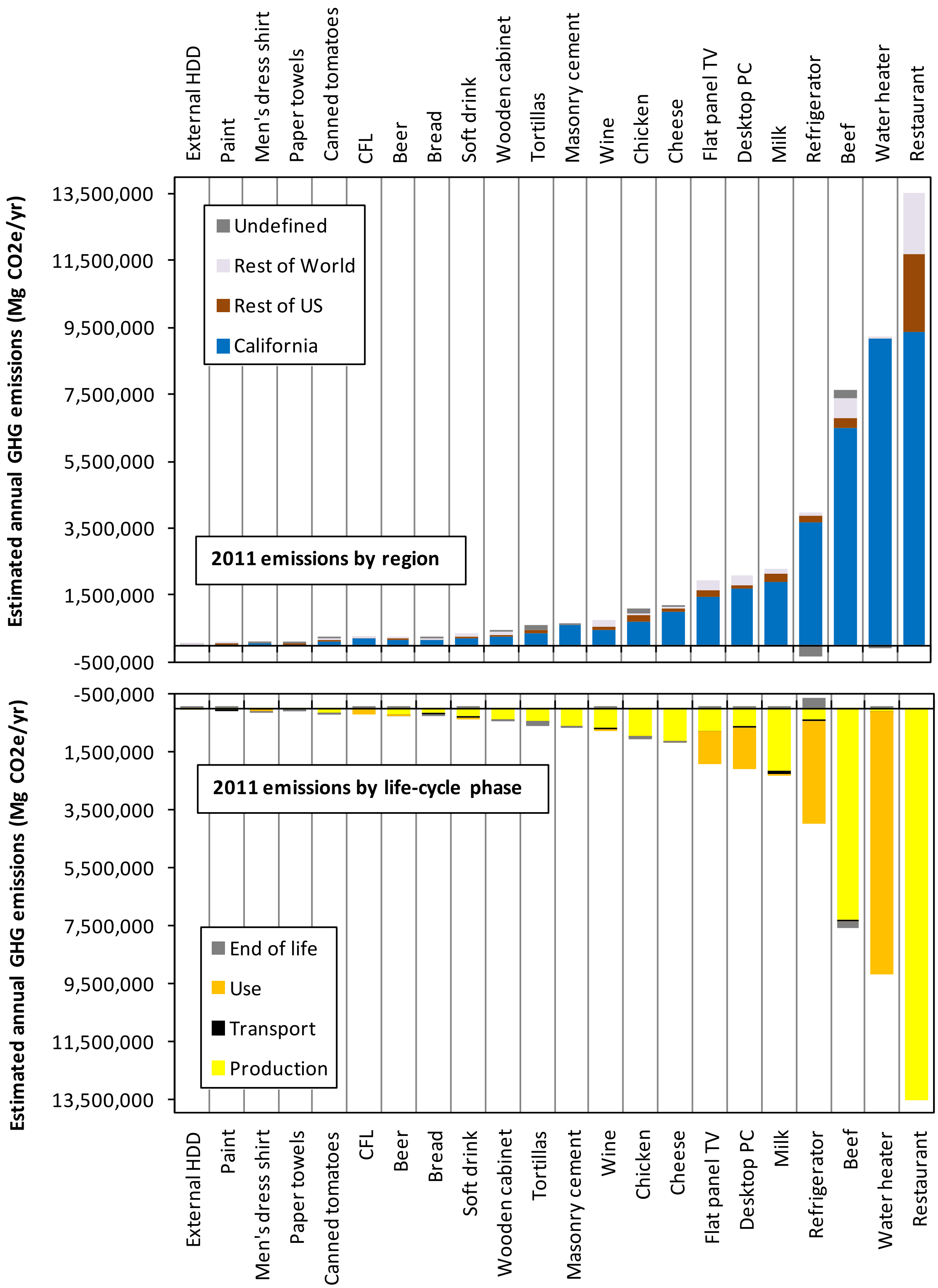


Figure 7: 2011 baseline case results for the 22 products by region and life-cycle phase

Figure 8 summarizes the results of the low carbon technical potential case for the 22 products. Not surprisingly, the estimated potentials for emissions reductions are greatest for the largest emitting products seen in Figure 7. These results are also interesting when compared to the estimated potential reductions by product in Appendix F. For some products, the study estimated that significant savings might be achievable on a per-product basis (e.g., over 40% savings were estimated for bread and tortillas in Appendix F). However, much larger potential emissions reductions might be achievable for products with lower per-unit potential (e.g., only 10% for beef in Appendix F) if they have large life-cycle GHG emissions and are purchased in large annual quantities in the state. In other words, a small per-unit reduction for some products might lead to large reductions at the level of the state.

The results in Figure 8 demonstrate the utility of the study's methods for assessing product carbon labels and standards. The combination of the California MRIO LCA model—which provides much-needed estimates on the region of origin for GHG emissions—and the low-carbon manufacturing and design estimation approaches in this study can allow decision makers to identify and rank candidate products based on their potential for reductions in the state (rather than non-region specific per-product absolute emissions estimates from standard LCA approaches). For example, Figure 7 suggests that milk has higher annual life-cycle GHG emissions than desktop control units or flat panel TVs, but Figure 8 suggests that desktop control units and flat panel TVs might have larger potential for GHG emissions reductions in California through established best practices than milk would.

A key caveat to interpreting the potentials for use phase GHG emissions savings in Figure 8 is that some of the predicted potentials are already targeted by existing standards and policies. For example, new energy efficiency standards for flat panel TVs in California will account for a significant fraction of the estimated savings compared to the 2011 baseline case. Because this study identified best practice energy efficiency beyond existing standards, however, there would still be reduction potential associated with driving the market to best practice beyond established standards. Future studies should look into the marginal potential savings that low-carbon product policies might deliver beyond existing standards (which was beyond the scope of this study).

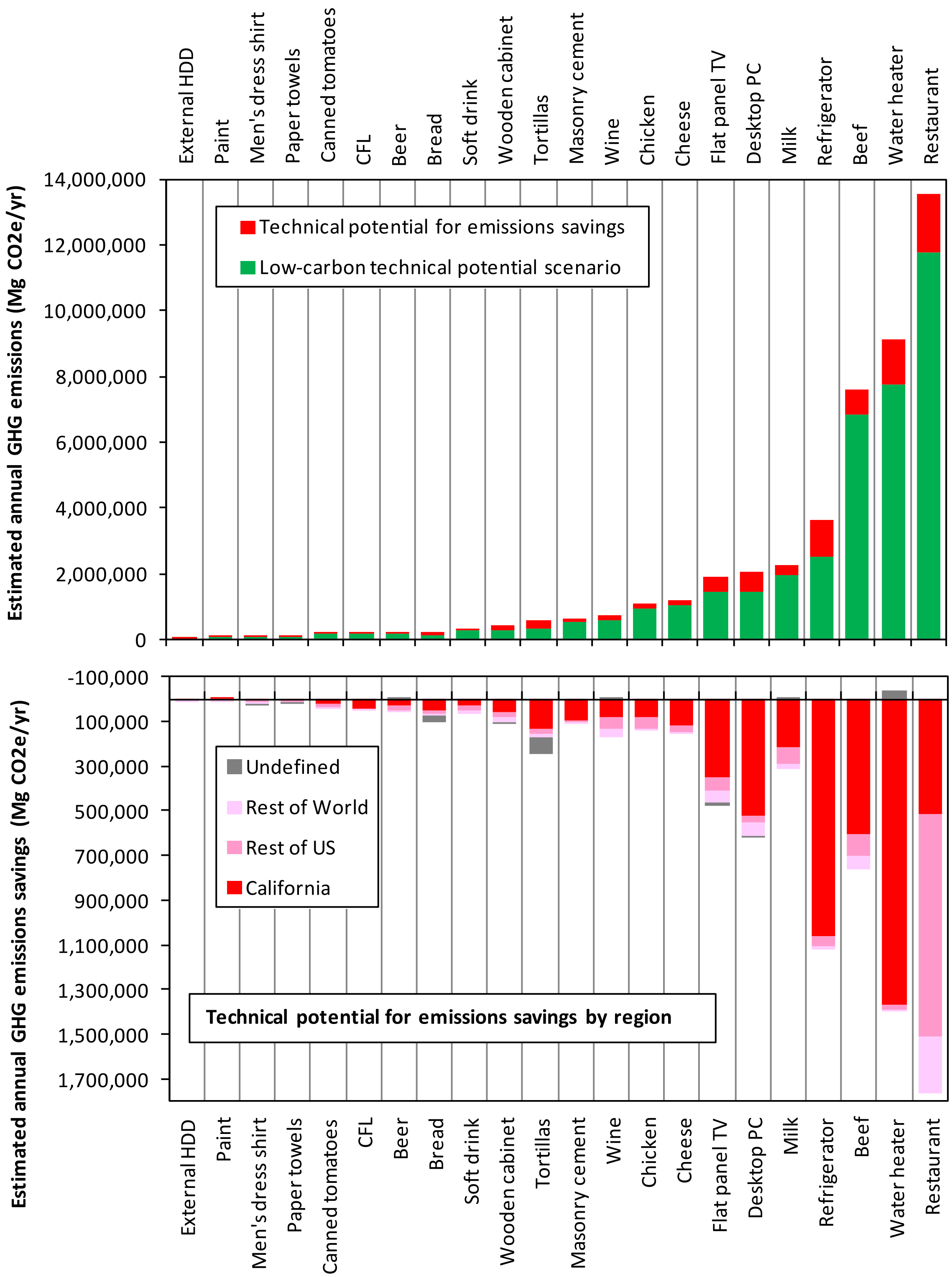


Figure 8: 2011 low carbon technical potential case results for the 22 products

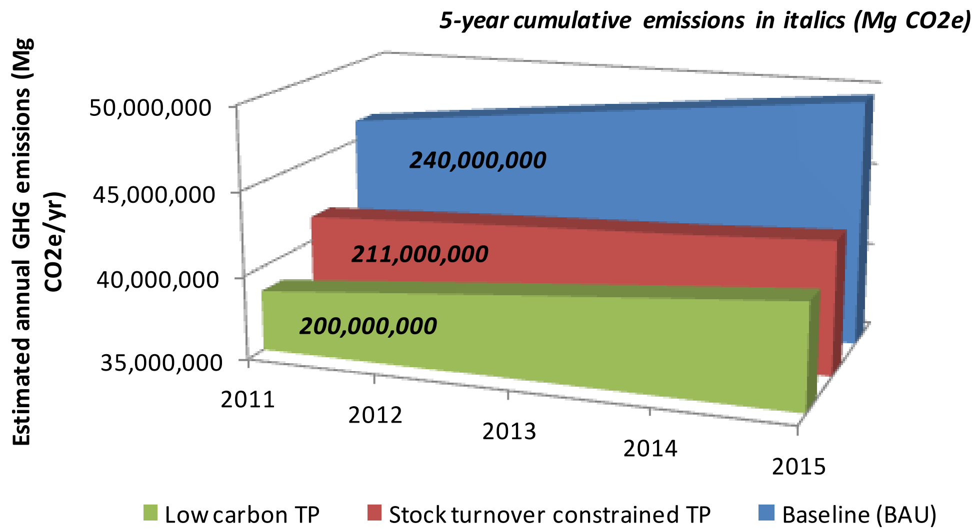
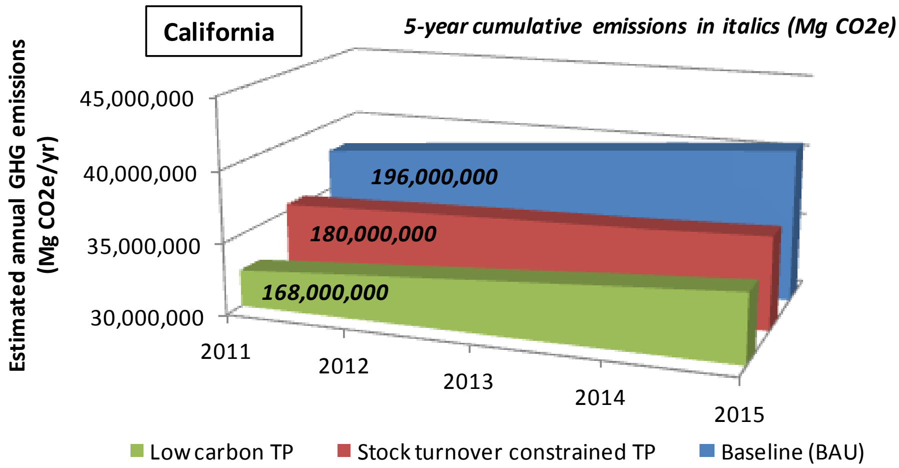


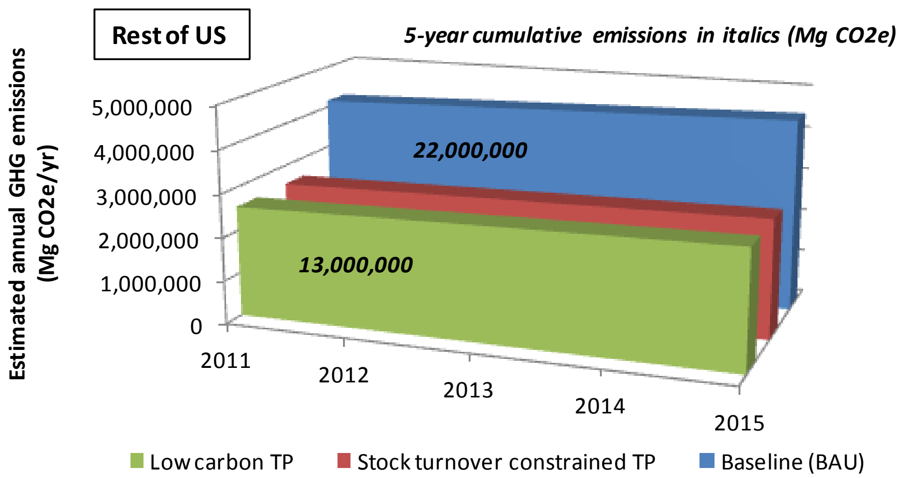
Figure 9: Results for scenarios 1-3 with cumulative emissions

Figure 9 shows the results of the projections for the BAU, low carbon TP, and stock turnover constrained TP scenarios. Included in the figure are the estimated cumulative emissions associated with the life-cycles of all 22 products over the projection period in each scenario. The cumulative emissions estimate for the BAU scenario is 240 million metric tons of CO_2e (or 240 Tg CO_2e); for context, total California net GHG emissions in 2008 amounted to 474 Tg CO2e (CARB 2010).

As can be seen from Figure 9, the estimated lower bound on cumulative emissions (i.e., the upper bound on potential emissions reductions) in the low carbon TP scenario is 200 Tg CO2e, which represents a 40 Tg CO2e (or 20%) reduction from the BAU scenario over the projection period. However, the stock turnover constrained TP scenario results suggest that 11 of these 40 Tg CO2e would be unattainable due to the natural rate of stock turnover of the durable goods considered in this study. Conversely, the 11 Tg CO2e difference between the technical potential TP and stock constrained technical potential TP scenarios can be viewed as the limit on potential reductions that might be achieved through accelerated stock turnover initiatives.

Figure 10 breaks the results in Figure 9 into the projected emissions that were estimated to occur in the California, Rest of US, and Rest of World regions. (Note the y-axis scale change between the graphs in this figure; and that the totals in Figure 10 might not add up to the total in Figure 9 due to rounding.) Given that the stock turnover constrained reductions affect only the product use phase—the emissions of which occur solely in California—the technical potential TP and stock constrained technical potential TP scenario results are identical in the Rest of US and Rest of World regions. As discussed earlier in this section, the California MRIO LCA model estimates that the bulk of projected emissions in each scenario occur within the state. However, it can also be seen that of the 40 Tg CO2e reduction between the BAU and technical potential TP scenarios in Figure 9, only 28 Tg are expected to occur in California.





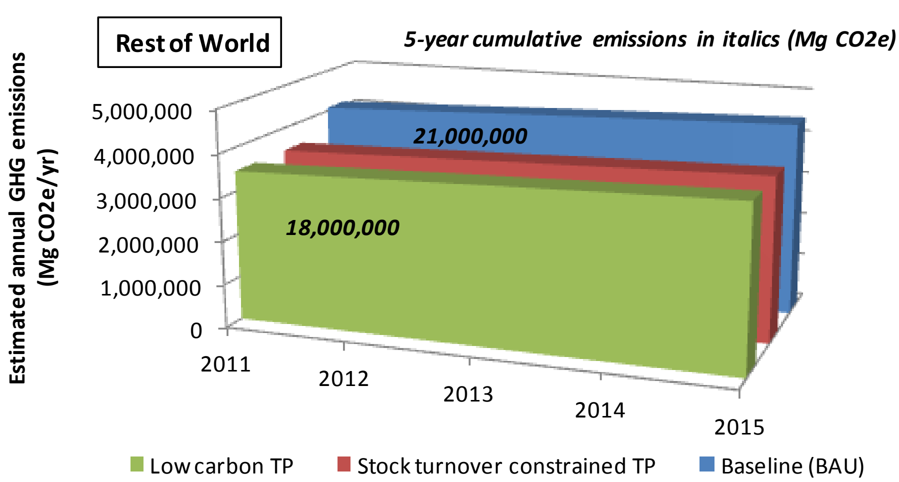


Figure 10: Results for scenarios 1-3 by region

Nine Tg of the 40 Tg difference are estimated to occur in the Rest of US region, and 3 Tg are estimated to occur in the Rest of World region. The results in Figure 9 and Figure 10 underscore two important points. First, given California's large population, installed stocks of durable goods, and annual purchases of retail products, the potential for Californians to drive significant reductions in life-cycle GHG emissions for the 22 products considered in this study is substantial. Second, given the multi-regional nature of product life-cycle GHG emissions, only a fraction of the potential reduction is likely to occur within the state. These two points suggest that while life-cycle GHG emission reduction policies might be a worthwhile policy pursuit toward California's long-term GHG emission reduction goals, such policies should ideally be focused on products that are likely to deliver the greatest in-state emissions reductions. The particular products that are expected to deliver the greatest life-cycle GHG emissions reductions in the state are discussed later in this section.

Figure 11 and Figure 12 summarize the projection results for all five scenarios considered in this study. As expected, the benchmark market update and life-cycle product standards scenarios show cumulative emissions that fall between the BAU and stock turnover constrained TP scenarios. And, similarly, the bulk of emissions in the benchmark market update and life-cycle product standards scenarios are estimated to occur in California.

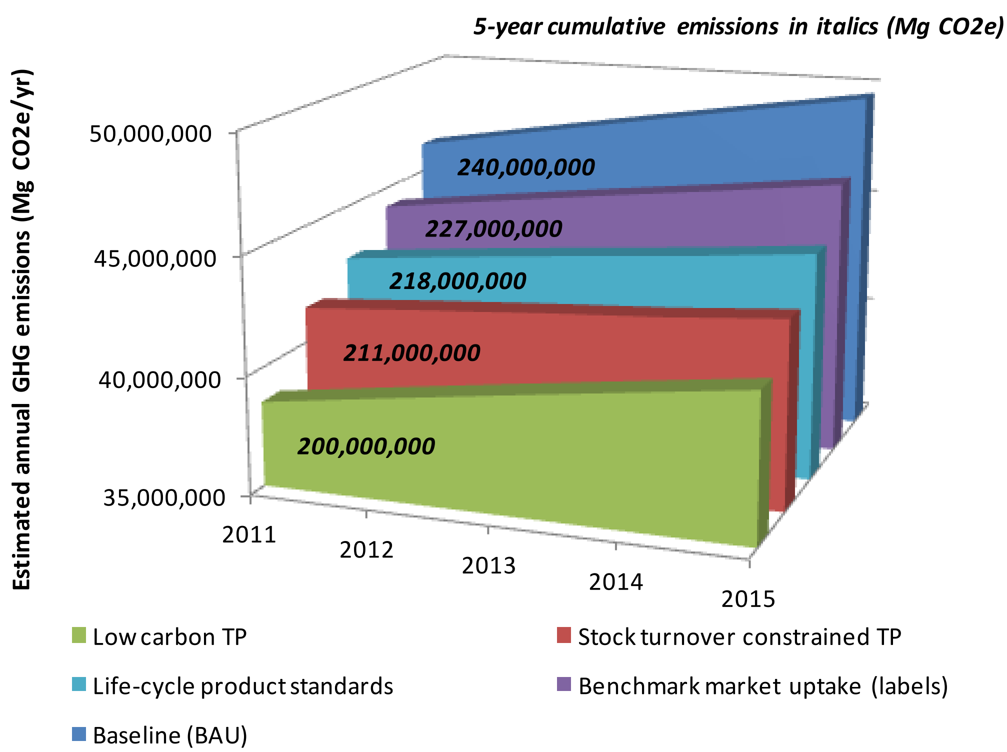


Figure 11: Results for all scenarios with cumulative emission reductions

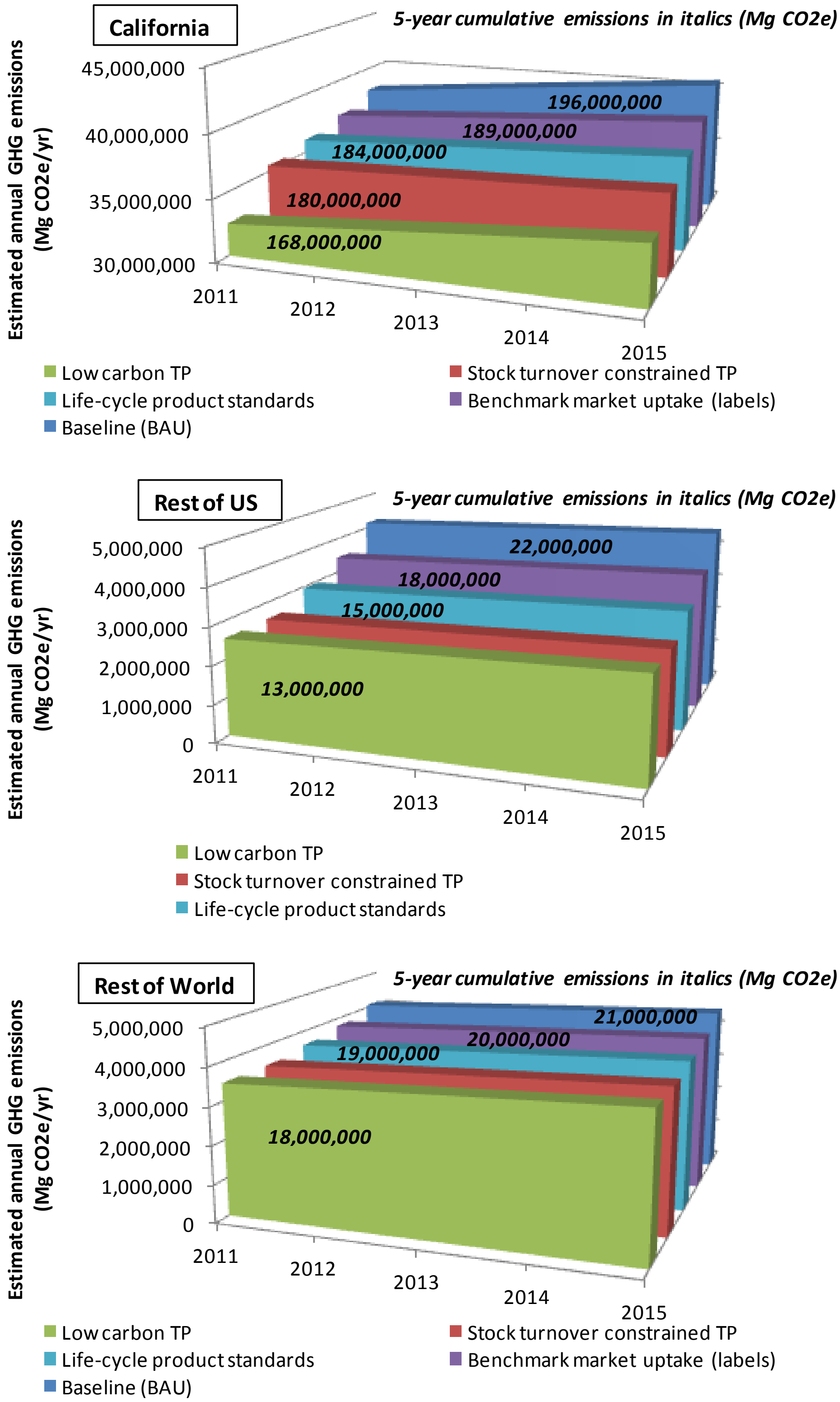


Figure 12: Results for all scenarios by region

The benchmark market uptake scenario results in Figure 11 suggest that, if consumers respond to carbon labels at the rates assumed in Table 21, emissions reductions over the projection period would amount to around 13 Tg CO2e (or roughly one-third of the 40 Tg upper bound on potential emissions reductions). However, an estimated 22 Tg CO2e of emissions reductions might be achieved if life-cycle GHG emissions standards were established, thereby removing the element of "low carbon" consumer choice from the purchasing decision. However, the total reductions achievable through such life-cycle product standards would be highly dependent on where the maximum emissions threshold is set for each product.

Interestingly, as seen in Figure 12, of the estimated 13 Tg of total emissions reductions in the benchmark market uptake scenario, only 7 Tg is expected to occur in California. Similarly, of the estimated 22 Tg of total reductions in the life-cycle product standards scenario, only 12 Tg are expected to occur in California. These results underscore again the importance of targeting products that are likely to deliver the greatest reductions in state if California were to pursue life-cycle GHG emission reduction policies of any type.

Figure 13 and Figure 14 present the estimated cumulative emissions reductions (compared to the BAU scenario) for each scenario by product for all regions combined, and for only the California region. As expected, the cumulative emissions reduction rank order of products in the Figure 13 follows closely the order that appears in Figure 8. The restaurant category shows the greatest potential emissions reductions for all regions, given that it was also (by far) the largest expenditure item included in this study. For all regions, the top reduction opportunities are dominated by restaurants, key energy using products, and livestock-intensive food items.

The difference in estimated reductions between scenarios sheds light on the importance of the assumed scenario – i.e., whether the technical potential is limited by stock turnover, consumer choice, or life-cycle standards policy—for the savings estimates associated with each product. For example, durable goods with longer average lives (e.g., refrigerators and hot water heaters) show much lower potential reductions due to stock turnover constraints than do durable goods with shorter average lives (e.g., the desktop control unit) or the LCD flat panel TV (which is rapidly replacing a much larger installed base of cathode ray tube televisions). Also, for products with low average market uptake rates estimated in Table 21, the estimated savings under the benchmark market uptake scenario are substantially lower than in the other scenarios (e.g., see the water heater and desktop PC results).

The results in Figure 14 show that the rank order of products by potential emissions reductions in California differs slightly from that of Figure 13, in that the estimated in-state savings for restaurants are much lower than the estimated savings for all regions (although still significant). Equally interesting is that the rank order and savings magnitudes for other key products (e.g., water heaters, beef, and milk) do not change significantly when considering only the California region. These results underscore the importance of considering the region of emissions savings for understanding when general national life-cycle data give reasonable regional estimates, and when they do not.

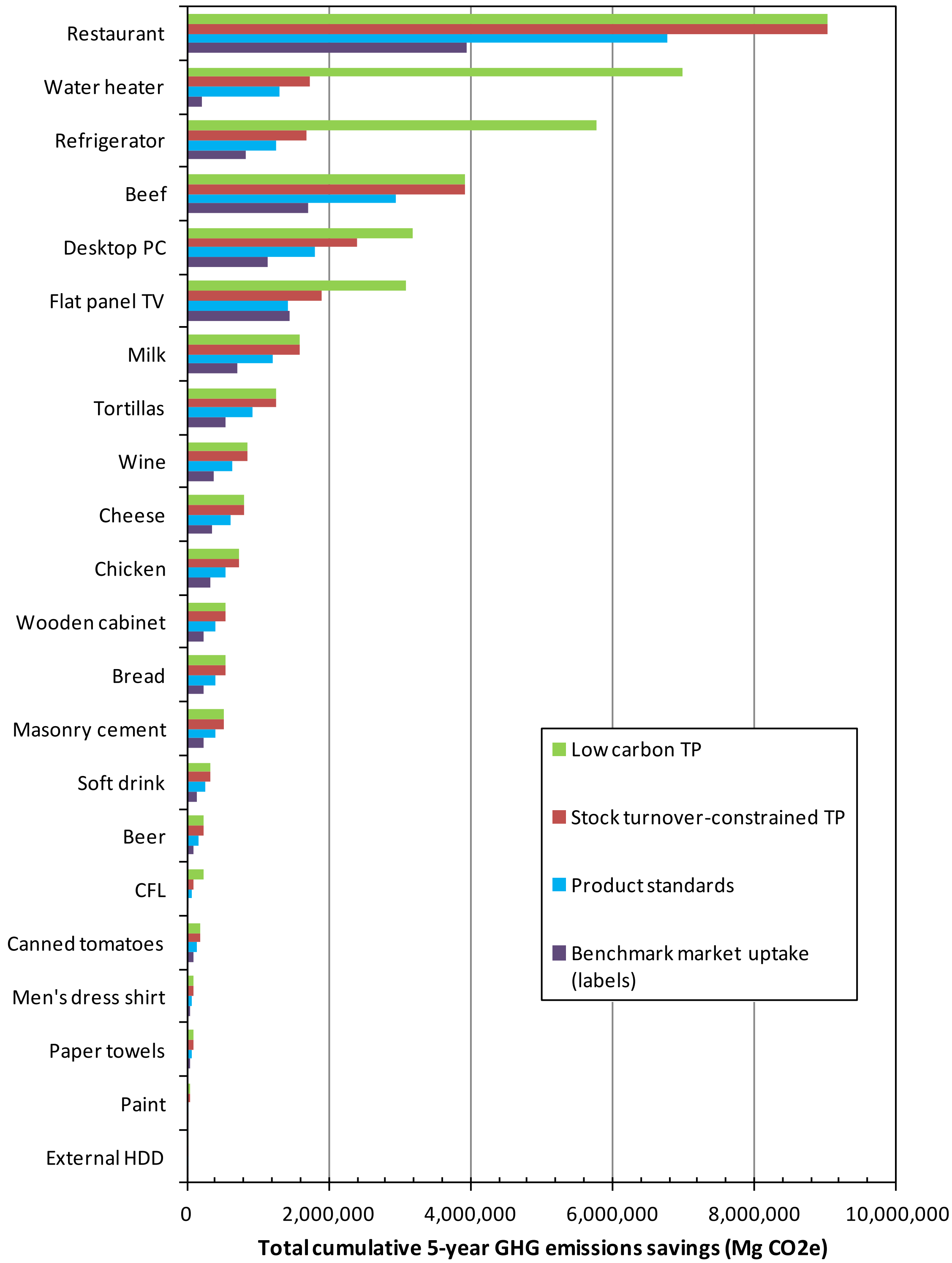


Figure 13: Cumulative emission reductions by scenario and product

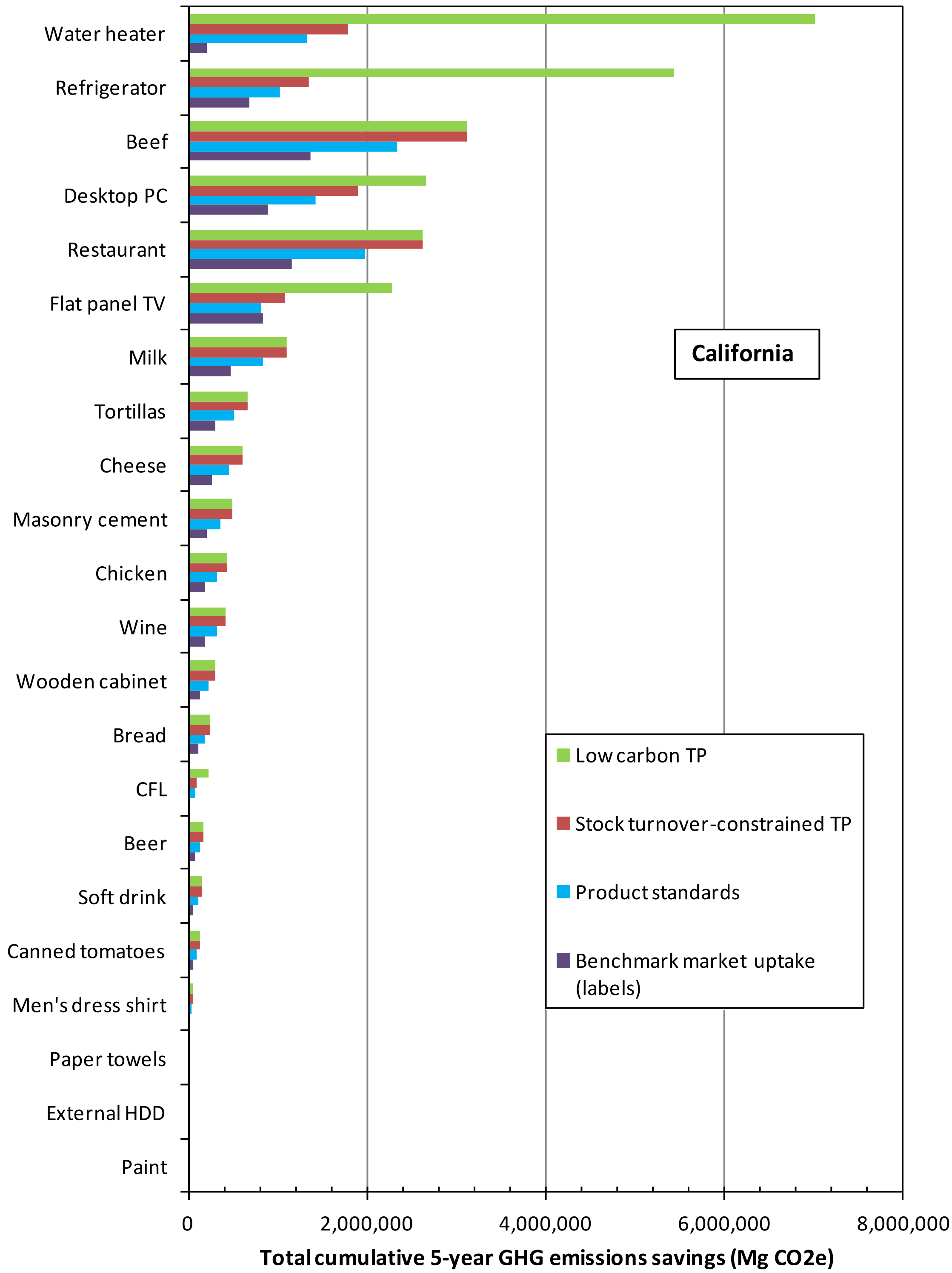


Figure 14: Cumulative emission reductions by scenario and product in California

Figure 15 presents the results of the uncertainty analysis for the 2011 baseline case. As discussed in the Methods section, these results include only the parameter uncertainties that the research team was able to characterize in its modeling efforts. As such, the uncertainty ranges in the figure should be interpreted as minimum uncertainty ranges, given that additional parameter uncertainty information might increase the magnitudes the confidence intervals (but would not decrease them). As expected, the greatest uncertainty was estimated to be associated with products that have significant non-energy GHG emissions in their life-cycle emissions footprints, given the large uncertainties that accompany the available non-energy GHG emissions inventory data that were used in the model. Also notable is that for products that have low annual life-cycle GHG emissions (i.e., the left side of the figure), the data uncertainties are not likely to matter when planning state-level policies (i.e., their life-cycle GHG emissions are not significant enough to warrant targeted policy efforts for GHG emissions).

The product-level uncertainty ranges in Figure 15 were applied in the projection for the low carbon TP scenario; the results of this analysis are shown in Figure 16. Given that the low carbon TP is associated with the greatest possible emission reductions, and all scenarios are calculated based on percent reductions compared to the BAU scenario, the results in Figure 16 show the largest range in results due to the identified parameter uncertainties.

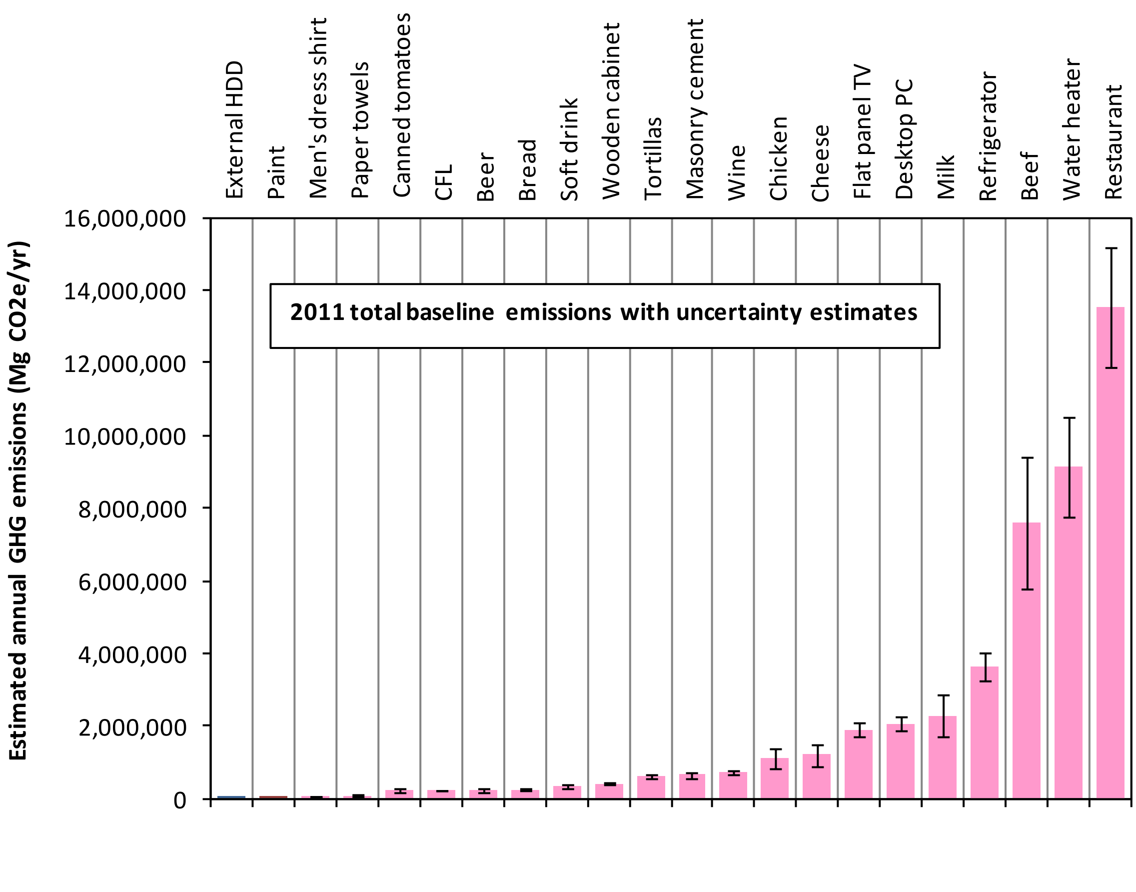


Figure 15: Uncertainty assessment results

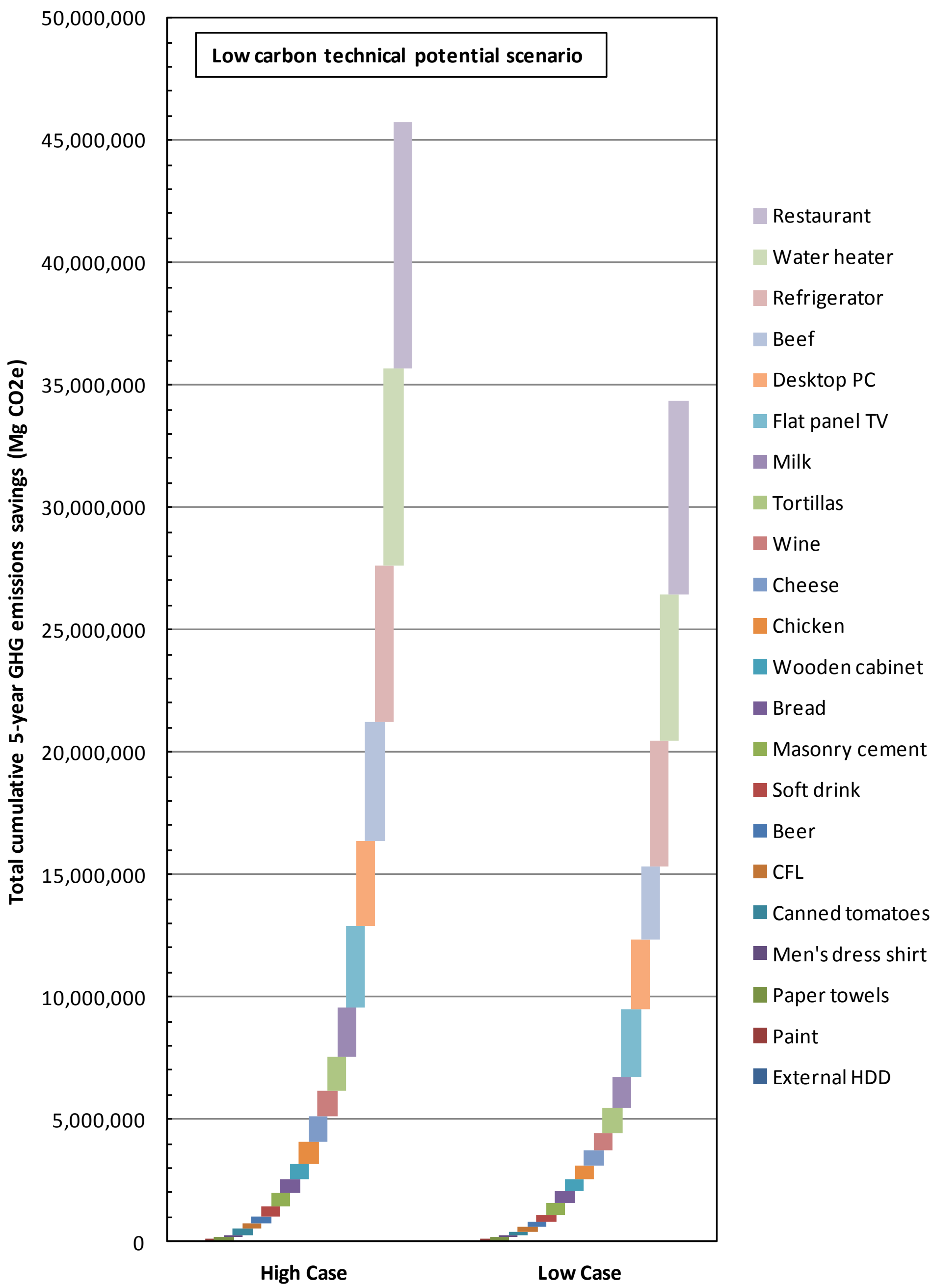


Figure 16: High and low emissions savings cases in the low carbon TP scenario

The results in Figure 16 are shown in cascade fashion so the results between products can be clearly seen. The high case shows the high end of savings predicted, and the low case shows the low end of savings predicted, considering the uncertainty ranges on the product-level results. A 25% difference between the high and low cases is evident, which suggests that even minimum parameter uncertainties have significant effects on the predicted emissions reductions in this study.

Summary and Conclusions

The goals of this project were to estimate the life-cycle GHG emissions associated with products purchased by Californians, to quantify the extent to which such emissions occur in California, and to assess by how much these GHG emissions might be reduced across product life-cycles if carbon labels and/or standards were successful in driving the market to best practice, low carbon products.

The research team first developed a comprehensive California LCA model to estimate the life-cycle GHG emissions of retail products, both inside and outside the state. This study considered 22 different product cases, which were selected by CARB from a preliminary short list of retail products and services prepared by the research team. The model consists of four major modules: (1) a multi-regional input-output (MRIO) model that is capable of estimating the full production (or value) chain energy use and GHG emissions of a wide variety of products; (2) a factory-to-retail transportation phase energy use and GHG emissions module; (3) a use phase energy use and GHG emissions of the 22 selected retail products; and (4) an end of life phase energy use and GHG module for product landfill, waste-to-energy recovery, recycling, and composting.

Next, the research team coupled outputs from the California MRIO model with analyses of "best practice" design features and life-cycle technology performance to estimate the life-cycle energy use and GHG emissions of hypothetical "low carbon" versions of the 22 selected products. Using results for current and "low carbon" versions of the 22 products, the research team then explored the GHG emissions reductions that might be achieved for the 22 selected products under different consumer adoption and policy scenarios over a five year projection period (2011-2015).

The results suggest that the California MRIO LCA model, when populated with product-specific information and coupled with product-level technical analyses of potential life-cycle design, operations, and technology improvements, could provide the state with valuable screening capabilities for identifying products and services that hold the greatest potential for in-state emissions reductions under life-cycle GHG emissions policy initiatives. Although this project only analyzed 22 products, the California MRIO LCA model could be applied to as many as 423 different product and service sectors.

Data uncertainties were found to be a key issue that should be addressed using the methodology moving forward. However, as a screening approach to identify products and services that are likely to hold large potential for life-cycle GHG emissions savings in California, the methods can be employed with reasonably approximate data as a means of illuminating areas of large savings for more detailed analysis in subsequent studies. As mentioned in the Introduction section, the California MRIO LCA model and study data were developed with such aggregate, state-level analyses in mind. For more precise estimates of product-level, life-cycle energy use and GHG emissions, more accurate data and methods than those applied here would be necessary for each life cycle stage.

For the 22 products considered in this study, it was found that energy-using devices and animal-based food items offered the greatest potential reductions among the products considered. In terms of the estimated potential GHG emissions reductions, 10 of the 22 products considered accounted for 90% of the estimated reduction potential. Of these 10, four were energy-using devices (refrigerator, water heater, flat panel TV, and desktop PC control unit) and four were animal-based food items (beef, milk, cheese, and chicken).

However, given that the technical potential GHG emissions reductions for some energy using products in this study (computers, TVs, and refrigerators) will be at least partially realized through existing energy state and federal efficiency standards, further study on the marginal emissions savings that might be realized through life-cycle carbon labels and/or standards for such products is warranted. Furthermore, the animal-related food products that rank high in the list also pose significant challenges for GHG emissions abatement, given the difficulties in measuring and managing emissions from land use, animal waste, and enteric fermentation (as opposed to energy efficiency at a food facility).

The total technical potential for life-cycle GHG emissions for the 22 products was estimated at 29 Tg CO2e over the period 2011-2015, when stock turnover is considered as a limiting factor for the achievable GHG emissions savings. Restaurants—the only commercial sector expenditure considered in this study—were estimated to be the number one emitter of life-cycle GHG emissions also ranked number one in terms of estimated GHG emissions reduction potential. The implication of these results is that the service sectors might hold large potential for GHG emissions reductions, but services have been largely overlooked to date in carbon labeling and standards initiatives (which have focused strictly on consumer products).

The study also found that only roughly two-thirds of the technical potential GHG emissions reductions was likely to occur within the state. When exploring how natural market uptake might affect the potential savings, the study found that even if consumers reacted with the same levels of uptake as products labeled with the ENERGY STAR—which is arguably the greatest labeling success story in the United States—that only roughly 40% of the technical potential might be realized. However, setting best practice standards for life-cycle GHG emissions for product purchases in the state might provide greater reductions—up to 75% of the estimated technical potential—since such standards would limit consumer choice to top performing products. However, the total reductions achievable through such life-cycle product standards would be highly dependent on where the maximum emissions threshold is set for each product, which is a process that must involve detailed technical analysis, stakeholder involvement, and market assessments.

The results also suggested that while significant savings might be achievable on a per-product basis for some products (e.g., over 40% savings were estimated for bread and tortillas), much larger potential emissions reductions might be achievable for products with lower per-unit potential (e.g., only 10% for beef) if they have large life-cycle GHG emissions and are purchased in large annual quantities in the state. The California MRIO LCA model and technical potentials analysis methods developed in this process were shown to have utility in identifying such potentials for large reductions.

However, one should consider carefully the timeline of this project (about three years) to generate estimates of life cycle energy and carbon emissions for a small number of products as a caveat.

Acquiring estimates for many more product sectors could be done, but would require similar amounts of time. That time though could be spent on the "back end" of the life cycle since the main product side of the model can be reused by others. Still, one could not expect to get such estimates for another 25 sectors in short order. Many barriers still exist to better automating this process.

While the input-output framework allows for rapid estimation of effects, various limitations prevent ideal application for policy analysis. One of the main limitations is the age of the data being used. Since the US Bureau of Economic Analysis (BEA) only releases benchmark input-output tables every 5 years (for years ending in 2 and 7, to match the Economic Census), and these tables take about 4 years to prepare, in the best case such models are already 5 years old, and inevitably will be 10 years old before updated with the next model (the 2002 benchmark will still be the newest in 2012-13 when the 2007 data is released). On the other hand, various studies have shown how "static" the US economy is, meaning that the resulting effects on supply chain production change relatively little between 5-year periods for which the IO models are updated.

Second, input-output methods can only estimate average fuel use and GHG emissions for a given IO sector as a whole. IO sectors are quite aggregated versions of underlying NAICS sectors. A typical IO sector in manufacturing is a direct map between a four, five, or six digit NAICS sector (which could in turn have many five or six digit NAICS sectors underneath). Likewise the IO sectors are in no way equivalent in size, economic output, or importance in the US economy. There is a single $200 billion electricity sector, as well as a small sector for tortilla manufacturing. This is the result of both the NAICS mapping (which is done by comparable production processes) as well as an underlying desire to create sectors representing commodity-like products. For IO sectors with heterogeneous product outputs (e.g., the frozen food sector which represents products from frozen pizzas to vegetables), the method provides fuel use and GHG emissions estimates that are averaged across all goods or services produced by that IO sector, the common process being "freezing". However, the method cannot estimate fuel use and GHG emissions specific to any product within that IO sector (e.g., frozen blueberries).

Third, the method presented here relies on many different data from a diversity of different sources to achieve the stated goals. In some cases single data sets are used for a particular process or life cycle mode that are distinct from the other modeling sources (e.g., the estimates of product transportation). Thus, the uncertainties associated with the method are significant. Data are used from appropriate national and California data sets that are popularly used by researchers in these domains, but the data are collected and compiled in completely different agencies, with different standards of testing and quality assurance. Data were generally mapped by SIC or NAICS sectors, which helps to link them, but could introduce additional uncertainties.

The preliminary parameter uncertainty estimates compiled in this project provide at least some idea of the minimum parameter uncertainty associated with the estimated averages for each IO sector. This project could not identify parameter uncertainty data for many of the model inputs, though, so the results should not be interpreted as comprehensive of all parameter uncertainties. Additionally, this project did not address modeling uncertainty, which is another key source of uncertainty inherent in the IO-based method as applied to multi-region models. It also did not address parameter uncertainty in the 2002 benchmark IO model itself. Several researchers have explored error propagation in IO tables in a theoretical fashion (see for example Hendrickson et al 2006, Nijkamp et al. 1992, or Bullard and Sebald 1977).

Overall all of these limitations are significant, but in the end the deliverable of a model able to represent reasonable ranges of estimates of production for California products at the sectoral level has been achieved. The effects differ for products produced in California, elsewhere in the US, or elsewhere in the world.

Recommendations

This study identified a number of areas for future work that would improve the robustness of the methodology, required data, and conclusions presented here. Additionally, some opportunities were identified for improving the state's capacity for conducting such analyses moving forward. For brevity and clarity, these recommendations are provided in bullet point fashion below.

- This study demonstrated how the California MRIO LCA model, coupled with design and technology improvement potentials analysis, can shed light on the products and supply chains that might provide the greatest potential GHG emissions reductions under best practices, and therefore help decision makers identify the most attractive candidates for life-cycle GHG emissions reduction policies. However, the study was limited in scope to only 22 products. Applying the method to a larger pool of products—even with preliminary estimates of improvement potential—would provide the state with a valuable screening method for identifying top opportunities among all viable options for further, more detailed analysis. This would ensure that future product-level potentials studies were focused on products and supply chains that were most likely to deliver large savings.

- While the state and/or its energy utilities have conducted detailed surveys on commercial (Itron 2006) and residential (KEMA 2010) buildings sector end uses and technologies in California, few survey or case study data are available in the public domain on the energy use, emissions, and end use technology practices of the state's manufacturing and agricultural industries. Together these industries account for almost 300 of the 423 sectors represented in the IO model. Comprehensive surveys and public use databases on these industries would vastly improve the accuracy of the sector-level assumptions in the MRIO model, and would provide valuable information to researchers and decision makers for assessment of state energy and climate policies.

- A key limitation of the LQ method employed in this study is that it assumes in-state production will meet consumer purchase demand before imports would; the limitations of this assumption have been discussed previously. However, additional study of, and perhaps better data collection from, key sectors (e.g., computer manufacturing) that have large economic presence in the state (e.g, for design and management) but small/no manufacturing presence would shed light on the products/sectors for which the LQ method is not likely to provide reasonable approximations of locally-sourced purchases.

- As no robust data are available on the annual purchases of products and services by Californians, the research team had to use national-level data as proxy information. Better data on the purchases of California households (e.g., modeled after the U.S. consumer expenditure survey) would provide more credible estimates from the California MRIO LCA model.

- This study only chose one commercial sector (restaurants) for detailed study, which was found to have the highest life-cycle GHG emissions footprints among the 22 products as well as the highest estimated GHG emissions reduction potential. Future analysis of other commercial sectors is recommended to assess how the supply chains of services might compare to the supply chains of manufactured products with respect to their potential for delivering GHG emissions reductions under carbon labeling and/or standards policies.

- The results for natural market uptake in this study were based on loose proxy data from the ENERGY STAR program; market studies (and ideally, empirical data) are needed to get a more realistic idea of potential consumer response to carbon labels (and therefore the likely savings potentials by product).

- The model and analyses required many different data, which were compiled from many different sources from different years and regions of study. Thus, uncertainties are likely to be significant (and well beyond the minimum parameter uncertainties quantified in this report). The state could consider conducting detailed life-cycle market, technology, and optimization studies for high emissions products (e.g., flat panel TVs, computers, animal-based foods) similar to Energy Using Products (EuP) preparatory studies conducted in Europe (e.g., ISIS 2007, VITO 2009, and IVF 2007). Such studies would draw in more stakeholders and provide a vehicle for more comprehensive data collection, analysis, and harmonization that could reduce uncertainties for key products.

- Given that the technical potential GHG emissions reductions for some energy using products in this study (computers, TVs, and refrigerators) will be at least partially realized through existing energy efficiency standards, further study on the marginal emissions savings that might be realized through life-cycle carbon labels and/or standards for such products is warranted.

- California agencies should consider working with the U.S. EPA and foreign entities like the Carbon Disclosure Project (CDP) and their emerging efforts to require GHG reports from relatively large facilities, and expand the scope of these reporting requirements at the unit process level (e.g., emissions per unit of product). Without this information, EPA and other agencies are simply developing databases of emissions that mostly replicate available knowledge. By acquiring data per unit of production, they could serve to validate existing LCI database values, such as those used in this study, and to begin to assess the variability in production emissions at the facility level. With better information on this variability, the feasibility of developing and implementing labels or standards could be better assessed.

- For all data collection and survey recommendations above, transparent and statistically consistent uncertainty information should be derived as part of the efforts so that analysts using these data in secondary fashion can better quantify uncertainties and take them into account when interpreting their results.

- While reducing energy use and GHG emissions are critical for mitigating climate change and improving energy security, more environmental indicators (e.g., water use, air pollution, water pollution, toxic emissions, and human health damages) should be incorporated into LCIs and analyses for carbon labels and standards. This information would allow for a better understanding of the full range of life-cycle environmental impacts, and allow decision makers to weigh tradeoffs and identify potential unintended consequences of life-cycle policies focused on GHG emissions reductions.

- While this study showed that there might be significant potential for life-cycle GHG emissions reductions for some products and services, life-cycle labels and standards are only one potential policy mechanism that might help the state realize such emissions reductions. The costs and benefits of potential carbon labels and standards policies should be compared to those of other possible policy avenues (e.g., minimum energy efficiency standards for industrial and agricultural facilities in the state).

- The improvement potentials analysis was limited to best practices that could be verified in the literature. Future studies might consider emerging technologies, next-generation processes and materials, or technologies that are still in the R&D stage. Such considerations would provide an indication of additional potential GHG emissions reductions to be tapped if manufacturers respond with more aggressive innovations that move beyond best practice.

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Appendix A: Derivation of energy and GHG emissions by IO sector

A.1 Details on Derivation of Energy and GHG Emissions Estimates by Sector[1]

A.1.1 Introduction

This appendix is separated into two parts, for derivation of energy and GHG emissions.

The energy data used in the 2002 US Benchmark Commodity by Industry EIO-LCA model are derived from several additional sources, generally for three aggregated sectors (minerals, manufacturing, transportation). The energy/fuel data are also the main required underlying data sources to estimate GHG emissions for the sectors.

A.1.2 Mineral Sectors

For the 11 mineral sectors (whose first 3 digits start with 211-213), the 2002 *Fuel and Electric Energy Report* published by the U.S. Census Bureau [Census 2002b] was used. This document reports fuel and electricity usage in physical units (e.g., short ton, barrel, cubic feet, gallon and kWh) as well as in some cases economic expenditures for the mineral sectors in 2002. Fuels presented in this report include electricity, coal, natural gas, and various petroleum-based fuels, which we again aggregate into the fuels listed above. Sectoral fuel use was calculated in terajoules (TJ) using the conversion factors shown in Table A-1 and A-2.

barrel (petroleum)=	42	gallons
barrel crude petroleum=	5800000	BTU
short ton anthracite coal=	25400000	BTU
short ton bit & lig=	26200000	BTU
1000 cu. ft. natural gas=	1035000	BTU
barrel distillate fuel oil=	5825000	BTU
barrel residual fuel oil=	6287000	BTU
barrel LPG=	4011000	BTU

[1] Information in this Appendix excerpted from “The 2002 US Benchmark Version of the Economic Input-Output Life Cycle Assessment (EIO-LCA) Model”, courtesy of the Green Design Institute, Carnegie Mellon University (with permission).

barrel gasoline=	5248000	BTU
barrel kerosene=	5670000	BTU
barrel natural gasoline=	4620000	BTU
BTU=	1055.1	Joules
TBTU=	1055100	GJ

Table A-1: Conversion factors [API 2005]

Finally, the use of fuels for each sector were divided by the industry outputs to obtain the fuel use factors in TJ/$million. The following economic assumptions were used to convert dollar-valued purchases of fuels into physical units.

Motor Gasoline	0.947	$/gal
Aviation Gasoline	1.288	$/gal
Kerosene-Type Jet Fuel	0.721	$/gal
Consumer Grade Propane and LPG	0.419	$/gal
Kerosene	0.99	$/gal
No. 1 Distillate	0.828	$/gal
No. 2 Distillate	0.759	$/gal
--No. 2 Diesel	0.762	$/gal
--No. 2 Fuel Oil	0.737	$/gal
No. 4 Distillate	0.657	$/gal
Residual Fuel Oil	0.569	$/gal

Table A-2: Conversions from Economic to Physical Unit Values – 2002 Refiner Prices of Petroleum Products to End Users Excluding Taxes (EIA 2008)

A.1.3 Agricultural Sectors

For the 14 agricultural sectors (sectors whose first 3 digits start with 111 and 112**)** the 2002 Census of Agriculture, specifically Table 59 was used [USDA 2002]. This document reports fuels as one category, “gasoline, fuels, and oils” and electricity usage in terms of expenditure by each of the NAICS codes included in the table. The 1997 Census of Agriculture included more detailed fuel expenditure information listing four fuel categories: gasoline and gasohol, diesel, natural gas, and LPG, fuel oil, kerosene, motor oil, grease, etc [USDA 1997]. The 1997 allocation of fuels within each sector was used to disaggregate the "gasoline, fuels, and oils" category within the 2002 Census. Expenditures were converted into physical units using values presented in Tables A-5 and A-6. Physical units were converted into terajoules (TJ) using the conversion factors shown in Table A-1.

Petroleum Prices Assumed for Agricultural Fuel Use, 2002			
Diesel	0.964	$/gal	[USDA 2005]
Gasoline, bulk delivery	1.374	$/gal	[USDA 2005]
LPG, bulk delivery	0.925	$/gal	[USDA 2005]
Residual Oil	0.561	$/gal	[EIA 2010]

Table A-3. Agriculture-specific Conversions from Economic to Physical Unit Values

A.1.4 Manufacturing sectors (all sectors from IO 311111 to IO 33999A)

The electricity and fuel use for manufacturing sectors (representing 279 of the 426 sectors in the model) were estimated using data from the 2002 *Manufacturing Energy Consumption Survey* (MECS) [EIA 2006]. This report presents fuel and electricity usage in trillion BTU, in 3 to 6 digit NAICS forms with physical units of BTU. Note that the specific MECS data required is for non-feedstock use of energy and fuels; as an example we do not consider feedstock use of petroleum for making plastics to be a use of petroleum in our data.

For sake of explanation, Table A-3 presents an *excerpt* of data reported in MECS. Since the MECS and IO data were from the same year, no further adjustments were made to the data.

NAICS Code	Major Group and Industry	Total	Net Electricity	Residual Fuel Oil	Distillate Fuel Oil	Natural Gas	LPG and NGL	Coal	Coke and Breeze	Other
311	Food	870	212	12	19	528	5	26	0	34
311221	Wet corn milling	228	23	0	0	61	0	121	0	11
313	Textile Mill Products	220	86	4	2	74	2	22	0	15
315	Apparel and Other Textile Products	30	12	0	1	16	0	0	0	0

Table A-3: Excerpt of data reported in 2002 MECS (Trillion BTU) [EIA 2006]

Two tables of the overall MECS data were used for building the EIO-LCA model: table A-2 (fuel consumption for energy purposes) and table 3.5 (selected byproducts for fuel consumption for energy purposes), which breaks up the "other" column of Table A-2 into 6 further categories. While the MECS data is a valuable single source of data on energy use for more than half of the sectors in the model, a significant shortcoming is that it is highly aggregated. As shown in excerpt Table A-3, the estimates provided are generally at the 3-digit NAICS level (e.g., NAICS 311). There are 29 sectors in this model that begin with 311. Thus the values from MECS for NAICS 311 need to be allocated to many sectors (except for sectors like 311221 which were explicitly provided by MECS).

In the case of the detailed fuel data estimates (row 1 of Table A-3), they were allocated from the 3-digit to 6-digit sector level by considering the dollar purchases of the fuels of each commodity sector in the model from the relevant industry sectors (i.e., from the 2002 US Benchmark IO Use Table). This assumption implicitly presumes that sectors within an aggregate industry sector have similar costs of energy. Table A-4 summarizes what data were used as proxies for this allocation. For example, in the 311 Food sector, if sector 311111 represented 90% of the dollar purchases of all the sectors beginning with 311 from power generation and supply in the use table, then 90% of the electricity use would be allocated to sector 311111.

Electricity	Purchases from 221100 Power Generation and Supply
Residual and Distillate Oil	Purchases from 324110 Petroleum Refineries
Natural Gas	Purchases from 221200 Natural Gas Distribution
Coal	Purchases from 212100 Coal Mining
LPG/NGLs	Purchases from 324110 Petroleum Refineries
Coke/Breeze and All Other	Purchases from 324110 Petroleum Refineries

Table A-4: Source of Allocation Factors for MECS Data

MECS also contains significant amounts of missing data for non-disclosure reasons. Wherever data was missing they were interpolated using the next-highest level of data.

A.1.5 Transportation sectors (IO 481000 – IO 4A0000)

Energy use of the 11 transportation sectors was estimated using data from the use table as well as *Transportation Energy Data Book* (edition 26), published by the U.S. Department of Energy [USDOE 2007, Table 2.5] which reports consumption of energy by fuel type and transportation mode in trillion BTUs for 2002. The modes include Highway (auto, motorcycle, bus, light truck, other truck) and Non-Highway modes (air, water, pipeline, and rail) of transportation. Fuels presented were gasoline, diesel fuel, liquefied petroleum gas, jet fuel, residual fuel oil, natural gas, and electricity. Energy use by automobiles, motorcycles, and light trucks (in orange below) were assumed to be out of scope and excluded since these vehicles are not generally used for production of goods and services (with the exception of corporate fleets used in service sectors, see "all other sectors" below). Table A-5 presents an excerpt of data included in the Transportation Energy Data Book.

	Gasoline	Diesel fuel	LPG	Jet fuel	fuel oil	gas	Electricity	Total
HIGHWAY	**16,447.50**	**4,922.70**	**26.9**	**0**	**0**	**11.6**	**0.9**	**21,409.60**
Light vehicles	**15,871.1**	**310.6**	**10**	**0**	**0**	**0**	**0**	**16191.7**
Automobiles	9,273.9	52.0	0	0	0	0	0	9325.9
Light Trucks	6,573.3	258.6	10	0	0	0	0	6841.9
Motorcycles	23.9	0.0	0.0	0.0	0.0	0.0	0.0	23.9
Buses	**6.7**	**171.7**	**0.2**	**0**	**0**	**11.6**	**0.9**	**191.1**
Transit	0.2	77.5	0.2	0	0	11.6	0.9	90.4
Intercity - c	0.0	29.2	0	0	0	0	0	29.2
School - d	6.5	65.0	0	0	0	0	0	71.5
Medium/heavy trucks	**569.7**	**4,440.4**	**16.7**	**0**	**0**	**0**	**0**	**5026.8**

Table A-5: Excerpt of data from Transportation Energy Data Book [DOE 1999], all values in trillion BTU

All energy usage from medium/heavy trucks was scaled down to avoid double counting energy use associated with own account transportation, using data from the BEA's *Transportation Satellite Accounts for 1996 (TSA)*, published by the Bureau of Economic Analysis [BEA 2000]. The TSA provide the estimated use of different transportation commodities incorporated in the regular input-output use table and the use of one additional commodity, the own-account transportation activities for 101 aggregated industries [BEA 2000]. Own-account transportation includes all transportation activities within a non-transportation industry that support the production processes, e.g., the trucks owned and used by a company as opposed to that company paying a trucking company for the same services. We assumed that trucks provided all own-account transportation. The use of "Motor freight transportation and warehousing" and "Own-account transportation" commodities were summed for the sectors and the ratio of own account transport was determined as (use of own account transportation)/(use of own account transportation + use of motor freight), and these ratios were applied to each sector to estimate gasoline and diesel usage for own-account trucking in each sector. The sum of the gasoline and diesel usage for own-account was then subtracted from total figures for medium and heavy trucks to yield the estimated petroleum usage by sector 484000, Truck transportation.

Energy usage for pipelines was mapped to the sectors 'Natural gas distribution' and 'Pipeline transportation' because the latter does not include the transmission and distribution of natural gas to final consumers, which also involves use of pipelines [Census 2005b]. Since the majority of freight-rails are powered by diesel fuel the electricity usage from rail travel was mapped to the 'Transit and ground passenger transportation' sector and all diesel fuel usage went to 'Rail transportation' sector [DoT 2004; AAR 2004]. All energy usage for buses was mapped to the 'Transit and ground passenger transportation' and 'Scenic and sightseeing transportation and support activities for transportation' sectors using the ratio of sectoral outputs as weighting factor.

All sectoral consumption data were converted into TJ. Finally, the sectoral use of fuels were divided by the corresponding industry outputs to obtain the fuel use factors in TJ/$million.

A.1.6 All other sectors

The sectoral economic values of consumption of coal, electricity, and natural gas of the roughly 100 sectors not covered by the sources above were estimated from the purchases of electricity and fuels from the 2002 CxI Use table at the Detailed level from the sectors listed in Table A-4 and then divided by the wholesale prices listed below to estimate the resource use in physical units.

As a result of the indirect estimates of energy use from this method, the estimates for these sectors are thus more uncertain than the other sectors. For example, the coal purchased by the wholesale trade sector is listed at $4 million, which is then adjusted by the average cost for coal paid by electric utilities (not a specific value for the wholesale trade sector), then converted to a value of 3.3 trillion BTU.

The following heat contents, provided in the *Transportation Energy Data Book* (edition 19), published by the U.S. Department of Energy [DOE 1999e, Table B.1], and the conversion factor of 947.8 million BTU/TJ was used to estimate the sectoral energy consumption in terajoules:

- Coal: 21.015×10^6 BTU/short ton
- Natural gas: 1,027 BTU/ft^3

Average Retail Price of Electricity [EIA] (cents per kWh)

	2002
Residential	8.44
Commercial	7.89
Industrial	4.88
Transportation	NA
Other	6.75
All Sectors	7.2

Natural Gas Prices [EIA] ($/1,000 cu. Ft.)		$/MBTU
City Gate Price	4.12	3.98
Residential Price	7.89	7.62
Commercial Price	6.63	6.41
Industrial Price	4.02	3.88
Electric Power Price	3.68	3.56

Petroleum Prices [EIA] - Sales to End Users - 2002

Motor Gasoline	0.947	$/gal
Aviation Gasoline	1.288	$/gal
Kerosene-Type Jet Fuel	0.721	$/gal
Consumer Grade Propane**Use this price for LPG (is	0.419	$/gal

that ok?)		
Kerosene	0.99	$/gal
No. 1 Distillate	0.828	$/gal
No. 2 Distillate	0.759	$/gal
--No. 2 Diesel	0.762	$/gal
--No. 2 Fuel Oil	0.737	$/gal
No. 4 Distillate	0.657	$/gal
Residual Fuel Oil	0.569	$/gal

Table A-6: Price data for extrapolating natural gas, electricity, and petroleum fuels usage for sectors where no better data exists

A.1.7 Summary and Validation of Energy Use Data

The total consumption of electricity and fuels were calculated after estimating energy use factors for all IO sectors and compared to EIA data. Table A-7 presents the results of the comparison.

Fuel	EIA Data	Total for all IO sectors	Percent Difference
Electricity, MkWh	2,070,000	2,054,220	0.99
Coal/coke, trillion BTU	21900	21500	1.7
Natural Gas, trillion BTU	18600	17200	7.6
Petroleum, trillion BTU*	120900	12200	5.8

*Note that the IO data does not include personal vehicle use that consumes approximately 95% of motor gasoline as well as some diesel, fuel oil, etc. Thus, households data were removed from the table

Table A-7: Comparison of the estimated total sectoral electricity and fuel use for 2002 to the EIA estimates [EIA 2004]

A.1.8 Special Notes on Estimates of Electricity Use

The electricity data represents the electricity consumption of each IO sector normalized by the total economic output of the sector and has the units of kWh/$. Both consumption and economic data were obtained from a number of different sources such as the Bureau of Economic Analysis (BEA), the United States Department of Agriculture (USDA), the 2002 Economic Census, and so on. The following section documents the development of the electricity vector along with the various public data sources that were used.

Economic output data:

The Bureau of Economic Analysis (BEA), US Department of Commerce, publishes Economic Input Output Accounts benchmark data every five years. The 2002 EIO LCA model uses economic data from the year 2002 to obtain the total economic output data in million $ from each of the IO sectors. The Standard Make tables[2] were used for this purpose, where the total output from any sector is the sum of the economic output of that sector across all other sectors that it might contribute to. This sum was used as the denominator value while determining the components of the electricity vector.

Electricity consumption data:

To estimate electricity consumption, the 428 IO sectors were grouped to include similar sectors based on the source of their consumption data. The various industry groups include agriculture, mining, utilities, manufacturing, transportation, and government agencies and households. For the sectors that did not have electricity consumption documented by any of the above data sources, the 2002 Benchmark Input-Output Standard Use table was used to estimate this information. The methodologies for estimating data for each of the industry groups are reported below.

Agriculture - USDA

The 2002 Economic Research Service of the United States Department of Agriculture (USDA) reports the total value of electricity purchased by14 IO sectors that are related to agricultural activities and include farming, milk production and animal production. According to this report, a total of $3900M of electricity was purchased by all the agricultural sectors[3]. Further, The EIA reports average retail prices of electricity to ultimate end users, and for agriculture, this was reported as $0.0488/kWh in the year 2002[4]. The total electricity consumed by all the agricultural sectors was calculated as the amount spent on electricity purchases ($3900M) divided by the average price of electricity ($0.0488/kWh). Thus, a total of 80,000 million kWh of electricity was estimated to be consumed by all the agriculture sectors. However, this data for 2002 was not available by sector.

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[2] BEA, 2008. 2002 Benchmark Input-Output Accounts. Bureau of Economic Analysis, Department of Commerce

[3] 2002 Economic Research Service of the United States Department of Agriculture (USDA)

[4] EIA Energy Power Monthly

The 1997 USDA Census of Agriculture[5], however, has electricity expenditure data by sector. Using this data, the percentages of electricity expenditures attributed to each of the individual sectors was determined. These percentages were assumed to be constant from 1997 to 2002 and were used to allocate the 80,000 million kWh of electricity among the 14 agriculture sectors for the year 2002.

Note: No data was available from the USDA for electricity consumed by 5 agricultural sectors that included activities such as logging, forests, fishing, hunting and trapping. For these sectors, consumption was estimated using the BEA Standard Use tables. Additionally, for the fishing sector, the Use table reports zero electricity purchases. It is assumed that the electricity purchases by the logging sector are negligible.

Mining - 2002 Economic Census

The 2002 Economic Census reports the total electricity consumption for each of the 29 NAICS sectors in the mining industry as well as the total electricity expenditure[6]. Since the EIO LCA model uses IO sectors, the Economic Census data was converted to represent the corresponding 11 IO sectors. The NAICS to IO bridge was used to implement this conversion where a many-to-one mapping between the NAICS sectors and the IO sectors was carried out. This mapping was used to estimate the total electricity purchased by each of the 11 IO mining sectors.

Utilities – Various

The IO sectors that correspond to utilities generation and distribution include the Power generation and Supply (IO sector code: 221100), the Water, sewage and other systems (IO sector code: 221300), Natural Gas distribution (IO sector code: 221200) and Pipeline transportation (IO sector code: 486000) and different data sources were used to estimate electricity consumed by each of these sectors.

Electricity consumed by the 'Power Generation and Supply' sector was estimated as 5% of gross electrical output consumed by power generators, as reported by the Annual Energy review, 2003[7]. Assuming that the same amount of power was consumed in 2002 as well, the total electricity consumption by the Power Generation and Supply sector was estimated as 202,000 million kWh.

Note: This is later compared to the value estimated through the BEA Use tables for verification.

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[5] 1997 United States Department of Agriculture (USDA) Census of Agriculture

[6] 2002 Economic Census, Sector 21: Mining: Industry Series: Detailed Statistics by Industry: 2002

[7] Annual Energy Review 2003, Energy Information Administration

Energy Star fact sheet[8] and American Water Works Association Research Foundation[9] survey of water and wastewater treatment plants report the total electricity consumption for this sector as 50,000 million kWh for the year 2002.

Note: This is later compared to the value estimated through the BEA Use tables for verification.

The Transportation Energy Data Book[10] estimates the electricity consumption by all pipelines as 72,600 million kWh for the year 2002. This includes natural gas distribution and transmission, crude petroleum and petroleum products, and coal slurry and water. Based on a two references from 1977 and 1981, the electricity consumption of petroleum, coal, and water pipelines is held constant at about 62,000 million kWh. The remainder of the electricity consumption by all pipelines is associated with natural gas distribution and is about 10,500 million kWh. This amount is allocated into two pools - Natural Gas Distribution (IO sector code: 221200) and Natural Gas Transmission (IO sector code: 486000). Based on discussions with experts in the pipeline industry on the relative energy needs of the two pipeline systems, distribution was assumed to consume 15% of the energy (1570 million kWh), and transmission 85% (8900 million kWh). Natural gas distribution is its own sector and therefore it was estimated to consume a total of 1570 million kWh. Natural gas transmission is included in the Pipeline Transportation sector along with petroleum, coal, and water pipelines and so the total electricity consumption for this sector was estimated as the sum of the individual sub-sectors and found to be approximately 71,000 million kWh.

Manufacturing - Economic Census Data

The 2002 Economic Census reports the total amount electricity purchased by each of the 473 NAICS sectors in the manufacturing industry as well as the total electricity expenditure[11]. Since the EIO LCA model uses IO sectors, the Economic Census data was converted to represent the corresponding 279 IO sectors. The NAICS to IO bridge was used to implement this conversion where a many-to-one mapping between the NAICS sectors and the IO sectors was carried out. This mapping was used to estimate the total electricity purchased by each of the manufacturing sectors.

Additionally, the 2002 Economic Census also reports the total amount electricity generated on-site and consumed by each of the NAICS manufacturing sectors. The NAICS to IO bridge was used estimate the electricity generated on site and consumed by each of the manufacturing IO sectors. The sectors for which this data was not available were assumed to have no electricity generation on site.

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[8] Energy Star

[9] American Water Works Association Research Foundation survey

[10] Transportation Energy Data Book, Edition X

[11] 2002 Economic Census, Sector 31: Manufacturing Industry Series Detailed Statistics by Industry: 2002

Transportation - Transportation Energy Data Book

The Transportation Energy Data Book provides estimates of electricity consumption of different transportation modes. This is the amount of energy consumed by the vehicles, rather than by the entire sector for overall operations. The electricity consumption data for buses and rail (transit and commuter) were used to represent the consumption of the Transit and ground passenger transportation (IO sector code: 485000) sector, and the total was found to be 19,000 million kWh. The intercity rail transit electricity consumption was assumed to represent the Rail Transportation sector (IO sector code: 482000) with a total of 2800 million kWh.

Air and water transportation are assumed to use no electricity directly to power the transport vehicles.

Note: For other transportation sectors, such as Air, Water, Truck transportation, where data was not explicitly available in the Transportation Energy Databook, the consumption was estimated using the BEA Standard Use tables.

Government agencies and Private Households - Annual Energy Review 2002

The EIA Annual Energy Review[12] for the year 2002 lists the total electricity consumption by government agencies and sources. The consumption by each of these sources was allocated to the different IO government IO sectors.

Electricity consumption by the postal service agency was allocated totally to the Postal Service sector (IO sector code: 491000).

Electricity consumption by the Defense group was allocated totally to the General Federal Defense government industry sector (IO sector code: S00500).

Electricity consumption by the Energy, Veterans Affairs, Transportation, General Services Administration, NASA, Agriculture, Justice, Interior, Health and Human Services and Others were allocated to the General Federal non-defense government industry sector (IO sector code: S00600).

Electricity consumption by the 6 other government sectors was assumed to be negligible.

The EIA Annual Energy Review also reports the total electricity consumption by residences as 1,270,000 million kWh[13], and this is allocated entirely to the Private Households sector (IO sector code: 814000).

[12] Annual Energy Review 2002, Energy Information Administration, Table 1.13 U.S. Government Energy Consumption by Agency and Source

[13] Annual Energy Review 2002, Energy Information Administration, Figure 8.1 Electricity Overview, Net-Generation-to-End-Use-Flow

Use table estimations

For sectors where actual electricity consumption is not documented, the Use tables are utilized to estimate these values. The 2002 Benchmark Input-Output Standard Use table indicates the money spent by each of the sectors on various other sectors. The Use column for all the sectors corresponding to the Power Generation and Supply sector is equivalent to the column of electricity purchases by all the sectors.

An average price of electricity is assumed for different sectors, as reported by the EIA, to estimate the electricity consumption as the ratio of the total expense ($) to the price of electricity ($/kWh) for each sector.

Average electricity price for different sectors

Sector	$/kWh*
Residential	0.0844
Commercial	0.0789
Industrial	0.0488
Transportation	na
Others	0.0675

*Source: EIA Energy Power Monthly: Average Retail Price of Electricity to Ultimate Customers

The average electricity prices for some sectors have been adjusted based on average price paid as calculated from Economic Census Data. For example, data for electricity consumption (kWh) and purchase ($) for the mining sectors is available from the census data. These values are used to estimate the average price of electricity paid by the mining industry and is found to be $0.053/kWh. Similarly, the average price of electricity for some of the manufacturing sectors such as Paper and Pulp, Aluminum, Petrochemicals and so on, was assumed to be approximately $0.036/kWh.

The 2002 Use table does not report any purchases for the Private Households sector and hence no electric consumption was estimated from this source.

The total electric consumption across all sectors excluding the Private Households sectors, estimated by the Use table, was found to be 2,340,000 million kWh for the year 2002. The total electric power generation in the US for the same year was reported to be 3,670,000 million kWh by the 2002 Annual Energy Review. The consumption by households as documented in the same report was 1,270,000 million kWh. Therefore, the total electricity consumption excluding private households as reported by the EIA was calculated to be 2,410,000 million kWh. The difference between the national consumption reported by the EIA and estimated from the use table (excluding private households) was approximately 2% and therefore compared well.

Note: For some of the actual sectors such as Power generation and Supply and Water, Sewage and other systems, the difference between the electric consumption estimated using data sources such as the EIA and Transportation Energy Databook, and the consumption estimated by the Use tables method, is very large. In such cases, the data sources were assumed more accurate and hence consumption data from the Use tables was ignored. However, the totals were found to compare very closely.

A.1.9 Electricity vector

The electricity purchased and consumed by the different sectors was then compiled from the different data sources into a single column. If no actual consumption data was available for any sector, then the Use table was used as a supplement. The total electricity consumption by all the sectors was about 3,680,000 million kWh and this was compared to the total documented by the EIA Annual Energy review. The difference was found to be less than 0.2% and therefore the values estimated seem reasonable.

The electricity generated on site by some of the manufacturing sectors is represented as a second column.

The economic output from each of the sectors was estimated as documented earlier.

The electricity vector (kWh/$) was finally calculated as the ratio of the electricity consumption (kWh) to the economic output ($) for each of the sectors. Another vector for the electricity produced on site is calculated is the ratio of electricity produced and consumed on site to the total economic output for each sector.

A.2 Greenhouse Gas Emissions Data In The Model

GHG emissions in general were estimated for the IO sectors based on either direct estimation of GHG emissions from fossil fuel combustion, or from other public EPA data on process GHG emissions for various GHG-intensive sectors where the emissions come from non-fossil combustion. The GHG emissions are separated into: CO2 emissions (fossil and process sources), methane (CH4), nitrous oxide (N2O), and hydrofluorocarbons (HFCs). Note that the latter three types of GHG emissions come largely from non-fossil combustion and thus are not separated into fossil and process emissions. Thus the two sections below discuss how the estimates for each category were made, and apply to all sectors.

Estimation of GHG emissions from fossil fuel combustion

Section 3 discussed how energy and fuel use was estimated for the sectors, resulting in intermediate estimates for each sector in physical units of BTU (before converting to TJ for display on the website). These BTU estimates by fuel for each sector were converted to trillion BTU, then multiplied by the GHG emissions factors in Table A-8 to estimate metric tons of CO2 from fuel combustion.

Carbon Intensities	Tg C/QBTU	mt CO2/TBTU
Utility Coal	25.76	94453
Industrial Coking	25.56	93720
Other Coal	26	95333
Nat Gas	14.47	53056
LPG	17.2	63066
Motor Gasoline	19.35	70950
Distillate Fuel	19.95	73150
Kerosene	19.72	72306
Jet Fuel	19.33	70876
Residual Fuel	21.49	78796
Coke Oven Gas		93997
Still Gas	15	64205
Coke		93997
Pet Coke		102132
Wastes/oils		73216

Table A-8: GHG Emissions Factors for Fuel Use

GHG emissions estimated by this method (in metric tons CO2 equivalents) are then normalized by the 2002 commodity sector outputs to be compatible with the economic input-output matrix from the BEA.

Estimation of GHG emissions from process or non-combustion emissions

Beyond fossil fuel combustion, there are GHG emissions throughout the economy from other sources. We separate our estimates of these sources between agricultural and non-agricultural sectors. For non-agricultural sources, EPA's ongoing GHG inventories track these other sources the EIO-LCA model associated them with specific input-output sectors as shown in Table A-9. EPA's summaries do not, however, distinguish agricultural emissions by crops, requiring a separate estimation method for agricultural emissions as described below. This Gas/Source table is taken directly from the EPA's Sources and Sinks of Greenhouse Gases in the U.S. from 2008. The most up-to-date document is used because the emissions values are constantly updated from year to year.

Gas/Source	Map	2002
CO2		5908.2
Non-Energy Use of Fuels	Unallocated	141.1
Iron and Steel Production & Metallurgical Coke Production	Iron+Steel	79.6
Cement Production	Cement	42.9
Natural Gas Systems	NG distribution, pipelines	29.6
Incineration of Waste	Unallocated	18.5
Lime Production	Lime and gypsum	13.1
Ammonia Production and Urea Consumption	Fertilizer	14.2
Cropland Remaining Cropland	Unallocated	8.6
Limestone and Dolomite Use	Unallocated	5.2
Aluminum Production	primary al	4.5
Soda Ash Production and Consumption	other basic inorganic	4.1
Petrochemical Production	petrochem	2.9
Titanium Dioxide Production	synthetic dye/pigment	1.8
Carbon Dioxide Consumption	Unallocated	1
Ferroalloy Production	iron and steel	1.4
Phosphoric Acid Production	fertilizer	1.3

Wetlands Remaining Wetlands	Unallocated	1
Zinc Production	nonferrous	0.9
Petroleum Systems	Refineries, pipelines, crude oil/gas	0.3
Lead Production	nonferrous	0.3
Silicon Carbide Production and Consumption	abrasives	0.2
CH4		580.9
Enteric Fermentation	See Ag below	134
Landfills	landfills	121.9
Natural Gas Systems	NG distribution, crude oil/gas	129
Coal Mining	coal mining	56.8
Manure Management	See Ag below	40.4
Forest Land Remaining Forest Land	Unallocated	18.1
Petroleum Systems	Refineries, pipelines, crude oil/gas	29.9
Wastewater Treatment	Water and sewer systems	24.7
Stationary Combustion	Power Generation	6.2
Rice Cultivation	Grain Farming	6.8
Abandoned Underground Coal Mines	coal mining	6.2
Mobile Combustion	Unallocated	3
Composting	Waste Management	1.3
Petrochemical Production	Petrochemicals	1.2
Field Burning of Agricultural Residues	Unallocated	0.7
Iron and Steel Production & Metallurgical Coke Production	Iron and steel	0.8

International Bunker Fuelsb		0.1
N2O		322
Agricultural Soil Management	See Ag below	207.6
Mobile Combustion	Unallocated	46.1
Nitric Acid Production	Fertilizer	19.3
Manure Management	See Ag below	14.2
Stationary Combustion	power gen	14
Adipic Acid Production	Other basic organic chemicals	6.1
Wastewater Treatment	Water and sewer systems	4.5
N2O from Product Uses	Unallocated	4.4
Forest Land Remaining Forest Land	Unallocated	2.2
Composting	Waste Management	1.4
Settlements Remaining Settlements	Unallocated	1.4
Field Burning of Agricultural Residues	Unallocated	0.4
Incineration of Waste	Unallocated	0.4
Wetlands Remaining Wetlands	Unallocated	+
International Bunker Fuelsb	Unallocated	1
HFCs		104.3
Substitution of Ozone Depleting Substances	Unallocated	83
HCFC-22 Production	Industrial gases	21.1
Semiconductor Manufacture	Semiconductor Mfg	0.2
PFCs		8.7
Aluminum Production	Primary Aluminum	5.3
Semiconductor Manufacture	Semiconductor Mfg	3.5
SF6		18.1

Electrical Transmission and Distribution	Power Generation	14.5
Magnesium Production and Processing	Primary Nonferrous	2.9
Semiconductor Manufacture	Semiconductor Mfg	0.7
Total		6942.3

Table A-9 Additional 2002 Emissions from EPA GHG Inventory

Agricultural Emissions

Our estimates of agricultural emissions are made starting with the 2009 EPA GHG Inventory values for Agricultural Emissions [EPA 2009]. Methane emissions occur due to the following activities: Enteric Fermentation, Manure Management, Rice Cultivation and Field Burning of Agricultural Residues. Emissions due to Enteric Fermentation were assigned to animal types by the EPA and subsequently assigned to the appropriate NAICS code based on animal. Similarly, methane emissions due to Field Burning were associated by the EPA with specific crops that were matched to NAICS sectors. Methane emissions due to rice cultivation were assigned to the Grain Farming Sector.

Nitrous Oxide emissions are organized by the EPA into the following source categories: Agricultural Soil Management, Manure Management, and Field Burning of Agricultural Residues. Agricultural Soil Management is by far the largest contributor of N2O emissions and is further subdivided into categories to reflect emissions due to synthetic and organic fertilizer application, manure application, release of nitrogen from crop residues (“residue N”, and indirect contributions from volatilization and leaching. Total fertilizer-related emissions reported by the EPA were assigned to NAICS sectors by creating ratios for each sector based on fertilizer consumption data reported by the USDA [USDA 2010, USDA 2002]. Residue N was assigned to NAICS sectors using harvested weight data given that these emissions are driven largely by materials remaining on the soil after harvest [USDA 2002]. Emissions associated with manure were assigned to NAICS sectors using the ratio of N2O emissions by sector to total, as calculated using the IPCC Tier1 method for calculating “N in urine and dung deposited by grazing animals on pasture, range and paddock” [IPCC 2006]. All other N20 emissions were allocated to NAICS sectors using acreage [USDA 2002].

Emissions from agriculture and soil management practices (sources) and are summarized in Table A-10. Note that the agricultural N2O emissions are the dominant sources of N2O emissions in the economy and within our dataset.

IO Code	IO Sector Description	CO2 (Tg CO2 Eq)	CH4 (Tg CO2 Eq)	N2O (Tg CO2 Eq)
1111A0	Oilseed farming		0.2	23.4
1111B0	Grain farming		7.3	68.9
111200	Vegetable and melon farming		0.0	5.2
111335	Tree nut farming		0.0	0.7
1113A0	Fruit farming		0.0	3.7
111400	Greenhouse and nursery production		0.0	2.1
111910	Tobacco farming		0.0	1.8
111920	Cotton farming		0.0	6.5
1119B0	Sugarcane and sugar beet farming		0.0	2.3
1119C0	All other crop farming		0.0	21.8
112120	Milk Production		46.6	8.3
1121A0	Cattle ranching and farming		100.7	64.8
112300	Poultry and egg production		2.7	1.9
112A00	Animal production, except cattle and poultry and eggs		24.6	9.9
113300	Logging			0.05
113A00	Forest nurseries, forest products, and timber tracts			0.05

Table A-10 GHG Emissions from Agriculture Sectors

GHG emissions estimated by the methods in section 4 (in metric tons CO_2 equivalents) are then normalized by the 2002 commodity sector outputs to be compatible with the economic input-output matrix from the BEA.

GHG Emissions Validation

The sectoral total GHG emission values were compared to the EPA's total US emissions inventory estimates for year 2002 as shown in Table A-11 [EPA 2008]. The largest error between the total in the model and the EPA reported total was for CO_2 from fossil fuels, which was overestimated by approximately 1.8%.

	CO2, process	CH4	N2O	HFCs, PFCs, & SF6	CO2 from fossil fuels
Total in Model	197	560	267	48	4556
Direct HH		3	46		1458
Unallocated	175	19	10	83	
Total accounted	373	582	323	131	6014
EPA Total	373	581	322	131	5908
Difference	0	1	1	0	106
Difference %	0.0%	0.2%	0.2%	0.1%	1.8%

Table A-11: Comparison of estimated sectoral nonfossil and process GHG emissions for 2002

Appendix B: Multi-Regional Input-Output (MRIO) model

Introduction

This section describes the construction of the MRIO model used in this study, which consists of three regions, each at a different geographical scale: the “California” region is at the sub-nation scale, the “rest of the US” (RUS) region is at the national scale, and the “rest of world” (ROW) region is at the global scale. Each region is mutually exclusive even though geographically one at a lower scale may lie within another at a higher scale.

The 2002 US input-output accounts include a standard use table for products as well as an import table representing use of imported commodities by domestic industries [1]. The first step in the creation of the model is thus to separate out imports, which are used both by US businesses for production (i.e., imported steel, auto parts, etc.) as well as sold to US households (i.e., final goods). This results in a generalization of the standard input-output model shown in *Equation B-3* to the 2 region (domestic and import) model shown in *Equation B-4* (see [2] for more details).

$$\mathbf{x} = \mathbf{A}\mathbf{x} + \mathbf{y} \qquad \textit{Eq. B-3}$$

$$\mathbf{x} = (\mathbf{A_d} + \mathbf{A_m})\mathbf{x} + \mathbf{y_d} + \mathbf{y_m} + \mathbf{y_{ex}} - \mathbf{m} \qquad \textit{Eq. B-4}$$

where $\mathbf{x}$ is the total implied production from a sector of the economy, $\mathbf{A}$ is the total, $\mathbf{A_d}$ is the domestic portion, and $\mathbf{A_m}$ is the import portion of the inter-industry transaction matrix, $\mathbf{y_d}$ is the final demand on domestic production, $\mathbf{y}$ is the final demand from a given sector, $\mathbf{y_m}$ is the final demand on imported production, $\mathbf{y_{ex}}$ is the final demand of exported goods, and $\mathbf{m}$ is the total import.

From *Equation B-4*, the next step is to separate production and consumption in California from the rest of the US. This is done using two methods: one is the simple location quotient (LQ), a standard method in regional economic analysis commonly used at the sub-national level [3]. The other method utilizes employment ratio multipliers (ERM) to estimate impacts occurred in the sub-national regions as the results of economic activities at the national level, a technique that has been used in Cicas 2005 [4] and Phares 2007 [5] to regionalize the US input-output model. The implementation of LQ and ERM in constructing the MRIO model is described in Sections B.2.1 through B.2.4.

Proxy for Economic Activities

Even though BEA has published estimates of industry output by aggregated sectors for individual states [6], detailed industry output data by detailed sectors and by states do not exist. In previous work by Carnegie Mellon researcher, Cicas [4] used employment data from the Occupation Safety and Health Administration as proxy for economic activities. Her regression analyses demonstrated that industry output and employment data are highly correlated and the proxy assumption is reasonable [4]. In the absence of better data, employment data by sectors and states are used as proxy for economic activities for creating the LQ and ERM vectors that will be used to separate California production and consumption from the US domestic inter-industry transaction matrix, $\mathbf{A_d}$.

To determine the most appropriate source of employment data, the Quarterly Census Employment and Wage (QCEW) data from the Bureau of Labor Statistics [7] and the County Business Pattern (CBP) data from the U.S. Census Bureau [8] were evaluated. Internal consistency check as well as comparison between the two data sets revealed that QCEW data set demonstrates high level of internal consistency. Furthermore, QCEW data did not require making assumptions that may unnecessarily increase the uncertainty of the estimates. Therefore, QCEW data set was chosen as the basis for LQ and ERM.

Location Quotient Method

Location quotient is a technique developed by IO researchers to estimate economic activities as proportion of total needs at regional level in the absence of survey data, which are not usually available at the sub-national level. Construction of MRIO model requires information on the inter-industry and inter-regional flow of goods and materials across state or regional boundaries. However, due to free-flow of materials and goods across state boundaries within the US, regional input-output data by detailed sectors are typically not tracked at sub-national level, except for a few existing survey-based studies. Collection and analysis of survey data on regional sectoral economic activities are very resource intensive and not frequently done by state and local governments. Alternative non-survey techniques have been developed by IO researchers to estimate economic activities as proportion of total needs at regional level [3, 9]. These non-survey techniques utilize publically available data by industry sector—such as employment, income, or output data— to estimate regionalized input coefficients by adjusting national technical input coefficients, which are readily available from BEA. Many of these non-survey approaches are variants of location quotient (LQ) technique. Although some authors compared regionalization results between LQ techniques and survey-based studies and have found that LQ may sometimes produce misleading results [10], LQ techniques remain popular for their simplicity in the absence of consistent regional data or resources to conduct surveys.

Among the many LQ techniques, the most straightforward and often used technique is Simple Location Quotient method (in this work, Simple Location Quotient is used synonymously as LQ). In LQ method, the proportion of industry output of a region is compared with the proportion of the total national output for that industry. It assumes that if an industry in the region produces enough supply to meet the demand of the region, it will supply the demand of the local region first before exporting to other regions. If the proportion of local industry's production is less than the national output for that industry, it is an indication that the demand in that region must be met with goods imported from other regions. Mathematically, LQ is defined as:

$$LQ = \frac{X_{industry\_i}^{region\_r} \Big/ X_{total}^{region\_r}}{X_{industry\_i}^{nation} \Big/ X_{total}^{nation}} \qquad \textit{Eq. B-5}$$

where X is either output or other proxies such as employment, income, or value added. If LQ is equal or greater than 1, i.e., the proportion of the industry output in that region is greater than the national proportion, it is assumed that the industry in the region is self-sustaining, and the technical input coefficients for that region are the same as the national coefficients. However, if LQ is less than 1, it is assumed that the local industry cannot fully supply the demand within the region, and the national technical input coefficient is scaled down by the respective LQ to obtain the estimated input coefficient of that industry in the region. The Bureau of Labor Statistics also provides a tool on its website to calculate LQs as well as brief discussion about LQ techniques. (Bureau of Labor Statistics, 2005).

For this study, the California portions of domestic production and consumption are separated from the rest of the US using LQ derived from the QCEW employment statistics for California and US [7]. To obtain the portion of **A** matrix representing California production that is supplying California demands, each column in the $\mathbf{A_d}$ matrix is multiplied by a vector of California LQ, element-by-element. To obtain the portion of the **A** matrix representing RUS production supplying California demands, each column in the $\mathbf{A_d}$ matrix is multiplied by a vector of complements of California LQ (i.e., one minus LQ_{ca} for each industry) element-by-element. These portions of California demands are incorporated into the final multi-regional **A** matrix, which is described further below.

Employment Ratio Multiplier

The ERM method is used to separate California and RUS supplies for meeting demands at the national level. ERM method is different from the LQ method in the regional designation of the "demanding" industries. Using the LQ method, the supplies from California and RUS are separated for meeting the demands of industries located in California; while using the ERM method, the supplies from these regions are separated for meeting the demands in the RUS region. In other words, the "demanding" industries in ERM method is in the RUS region, while in LQ method they are in the California region.

The ERM method assumes that demands at the national level are met by an "average" mix of supplies from sub-national regions based on each region's "shares" of production. It adjusts the national technical input coefficient by using ratio of industry output in a region as compared to national output for that industry. See Cicas [4] for a more detailed description of the approach.

The ERM is calculated as follows:

$$ERM = \frac{Employment_{industry\_i}^{region\_r}}{Employment_{industry\_i}^{nation}} \cong \frac{GSP_{industry\_i}^{region\_r}}{GDP_{industry\_i}^{nation}} \qquad \textit{Eq. B-6}$$

where GSP is Gross State Product and GDP is Gross Domestic Product for the region specified in the superscript and the industry specified in the subscript. To estimate the input coefficient of each industry sector in the sub-national region, the national technical input coefficient is scaled down by ERM. The result is a matrix representing the economic activities occurred in a region when a final demand is placed at the national level. An example of a scenario for applying the ERM approach may include estimation of coal supplies reduction in Pennsylvania as the result of a national policy to reduced coal-fired electricity production.

Similar to the LQ method, once an ERM vector is obtained, it can be incorporated into the domestic portion of the **A** matrix to separate California supplies from RUS supplies for meeting the demands by industries at the national scale. To obtain the portion of **A** matrix representing California production that is supplying industry demands in the RUS region, each column in the $\mathbf{A_d}$ matrix is multiplied by a vector of California ERM, element-by-element. To obtain the portion of the **A** matrix representing RUS production supplying RUS demands, each column in the $\mathbf{A_d}$ matrix is multiplied by a vector of complements of California RUS (i.e., one minus ERM_{ca} for each industry) element-by-element.

Construction of the Multi-Regional Model

Putting together California and RUS portions of the inter-industry transaction matrix, obtained from multiplying the domestic **A** matrix ($\mathbf{A_d}$) with a vector of LQ factors, a vector of ERM factors, a vector of 1–LQ, and a vector of 1–ERM, a multi-regional **A** matrix is created to represent the inter-regional transaction of the industries in the California, RUS, and ROW regions. The 3-region **A** matrix and its 9 compartments are illustrated in Figure B-1.

$$
\begin{array}{c|ccc}
 & \text{California} & \text{US} & \text{Rest of World} \\
\hline
\text{California} & \mathbf{A}_{\text{CA-CA}} \; \langle LQ_{CA} \rangle \times A_d & \mathbf{A}_{\text{CA-US}} \; \langle ERM_{CA} \rangle \times A_d & \mathbf{A}_{\text{CA-ROW}} \; 0 \\
\text{Rest of the US} & \mathbf{A}_{\text{RUS-CA}} \; \langle 1-LQ_{CA} \rangle \times A_d & \mathbf{A}_{\text{US-US}} \; \langle 1-ERM_{CA} \rangle \times A_d & \mathbf{A}_{\text{US-ROW}} \; 0 \\
\text{Rest of World} & \mathbf{A}_{\text{ROW-CA}} \; A_m & \mathbf{A}_{\text{ROW-US}} \; A_m & \mathbf{A}_{\text{ROW-ROW}} \; A
\end{array}
$$

Figure B-1. Three-region inter-industry transaction matrix, which consists of 9 blocks representing the transactions among the 3 regions.

This 3-region model assumes that only one national border is crossed in the production of any good. This means that goods produced abroad use no CA or RUS-made components and no goods are exported from the US only to be returned in goods imported to the US. Therefore, the ROW-CA and

ROW-US[14] compartments are filled with the US import portions of the **A** matrix (**A$_m$**), while the CA-ROW and US-ROW compartments are filled with zeros. Further, the rest of US and rest of world segments are assumed to produce goods similar to the US as a whole, such that the US EIO-LCA model can be used as a proxy for these production technologies. Therefore, the ROW-ROW compartment is filled with the original **A** matrix from the basic US IO model.

Using the 3-region **A** matrix, *Equation B-4* now becomes:

$$\begin{pmatrix} x_{CA} \\ x_{RUS} \\ x_{ROW} \end{pmatrix} = \begin{pmatrix} A_{CA-CA} & A_{CA-US} & 0 \\ A_{RUS-CA} & A_{RUS-US} & 0 \\ A_{ROW-CA} & A_{ROW-RUS} & A_{ROW} \end{pmatrix} \begin{pmatrix} x_{CA} \\ x_{RUS} \\ x_{ROW} \end{pmatrix} + \begin{pmatrix} y_{CA-CA} + \sum_{j \neq 1} y_{CA-j} \\ y_{RUS-CA} \\ y_{ROW-CA} \end{pmatrix}$$

Eq. B-7

Solving for the total production in each region, **x**, to express it in the Leontief inverse form, we get:

$$\begin{pmatrix} x_{CA} \\ x_{RUS} \\ x_{ROW} \end{pmatrix} = \left\{ I - \begin{pmatrix} A_{CA-CA} & A_{CA-US} & 0 \\ A_{RUS-CA} & A_{RUS-US} & 0 \\ A_{ROW-CA} & A_{ROW-RUS} & A_{ROW} \end{pmatrix} \right\}^{-1} \times \begin{pmatrix} y_{CA-CA} + \sum_{j \neq 1} y_{CA-j} \\ y_{RUS-CA} \\ y_{ROW-CA} \end{pmatrix}$$

Eq B-8

To assess the carbon footprint of California consumers, the consumption vector had to be split into final consumption of goods produced in California, goods produced in the RUS region, and goods produced in the ROW region. Similarly to the production part of the model, the final consumption vectors was first split into imported goods and domestically produced goods using the U.S. import matrix from 2002 [1]. The remaining consumption is split between goods and services produced in California and those produced in the RUS region using the LQ method, such that the availability of California-produced commodities would be related to the mix of commodities produced in California vs. the RUS region. Thus, the LQ method was used to split consumption of domestically produced goods in California into California-made goods and services ($y_{CA\text{-}CA}$) and RUS-made goods and services ($y_{RUS\text{-}CA}$) in *Equation B-7*.

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[14] The notation for the supplying and demanding regions is *i-j*, where *i* is the supplying region and *j* is the demanding region. For example, ROW-CA refers to the ROW production supplying the demands in the CA region.

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Appendix C: Product selection process

This appendix summarizes the process used to identify the 22 product cases analyzed in this study. The first step in the process was to choose key criteria for selecting interesting and relevant product case studies. Based on these selection criteria, the second step was to review existing data sources to identify a preliminary short list of products that met these criteria, and to present this list to CARB. The third step was for CARB to review the preliminary short list of products, and through discussions with internal and external staff, identify the final 22 product cases to be analyzed.

Selection criteria

The research team first identified a preliminary short list of candidate products based on five primary screening criteria. Each criterion is listed and discussed briefly below.

1. The product should have significant total life-cycle GHG emissions. Since the team can only focus on 22 products in detail, it gave priority to products that are likely to be major contributors to one's retail product carbon footprint (e.g., personal computers and beef and dairy products) or that come from product categories that are major factors in one's retail carbon footprint (e.g., food and eating out).

2. There must be credible data available for the product to facilitate the life-cycle modeling, improvement potentials assessments, and scenario projections objectives of our study. To perform the detailed assessments of 22 products, the team needs to use existing case studies, data, and models. So the team only chose products for which such detailed assessments are practical given existing resources.

3. The product should be from a sector for which there is a significant manufacturing and/or design/management presence in California. Priority is given to industries that are important from both economic and GHG emissions perspectives in California, and that will continue to be so over the near-term (i.e., through 2020). This criterion helped to ensure that the chosen products would be relevant to California's economy and GHG emissions reduction targets, both now and in the future.

4. The final list of products should represent a diverse mix of products and supply chain characteristics. This criterion helped ensure breadth of product coverage, so that the team could explore how GHG emissions footprints and reduction potentials might vary across product classes.

5. There should be a few products for which consumption by California residents is growing. Such products are likely to be of greater GHG emissions importance to California in the future (e.g., flat panel televisions).

Of these five criteria, the first two (significance of emissions and data availability) were given the greatest priority in the initial screening process. The final three criteria were then used to fine tune the preliminary short list.

Major contributors to one's retail product GHG emissions footprint

Figures C-1 to C-4 summarize the team's products and product categories presented to CARB that are expected to be major contributors to the retail product GHG emissions footprint of Californians. The research team explicitly excluded private transportation – the largest component of the typical household GHG emissions footprint – since much LCA work is already underway on this topic through CARB's low-carbon fuel standards initiatives. The team tried to choose a diversity of products representative of the major contributors to direct and indirect household emissions.

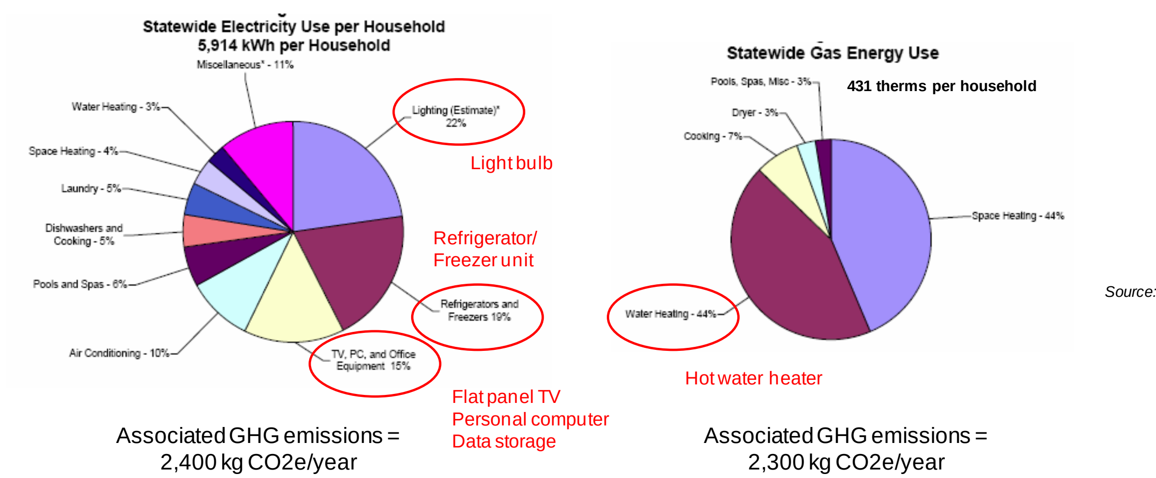


Figure C-1: CA Direct Household GHG Emissions

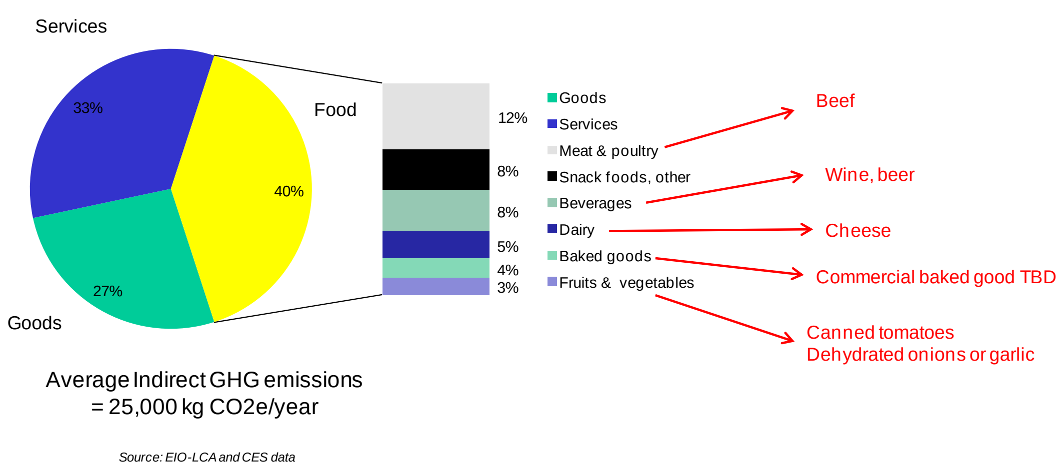


Figure C-2: California Household Indirect GHG Emissions: Food

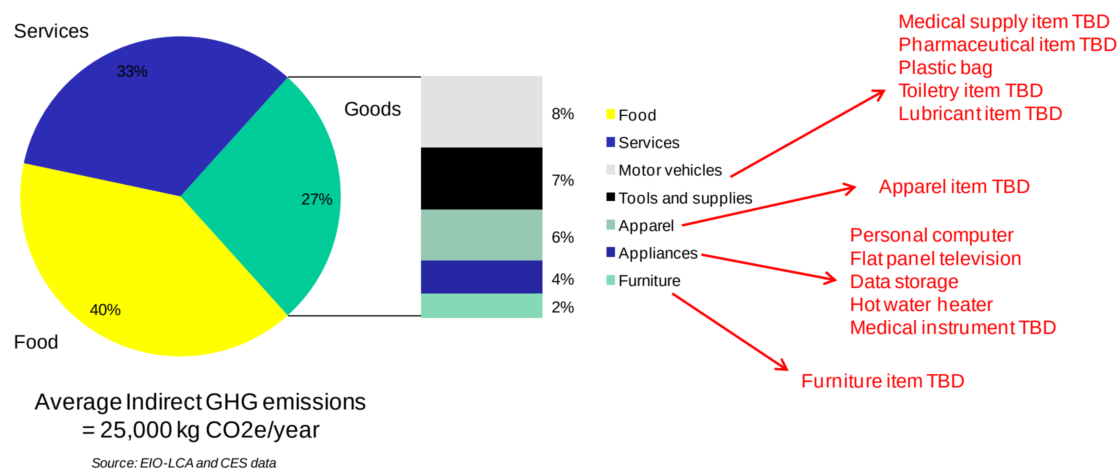


Figure C-3: California Household Indirect GHG Emissions: Goods

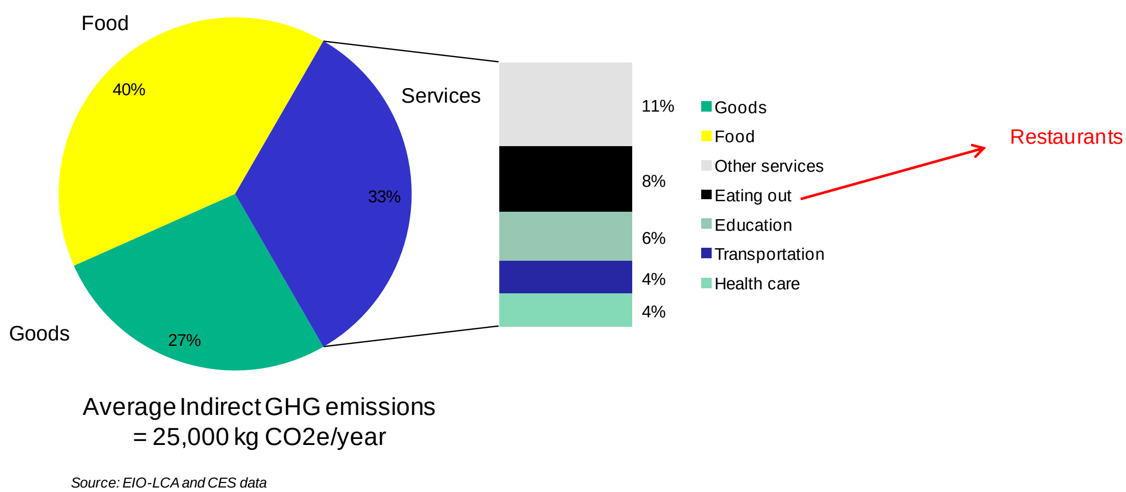


Figure C-4: California Household Indirect GHG Emissions: Services

Sectors of economic importance to California

Figure C-5 summarizes value of shipments projections for each NAICS code in California, which were derived using economic forecast data for California obtained from the California Energy Commission's Office of Demand Analysis. The 2020 growth projections are highly uncertain, but represent the best estimates of industrial sector growth in California over the state's GHG emissions reduction target period at the time of this study.

It should be noted that value added data are typically the best indicators of the importance of an industrial sub-sector in a given state. However, the only industrial forecast data available for California are on a value of shipments basis.

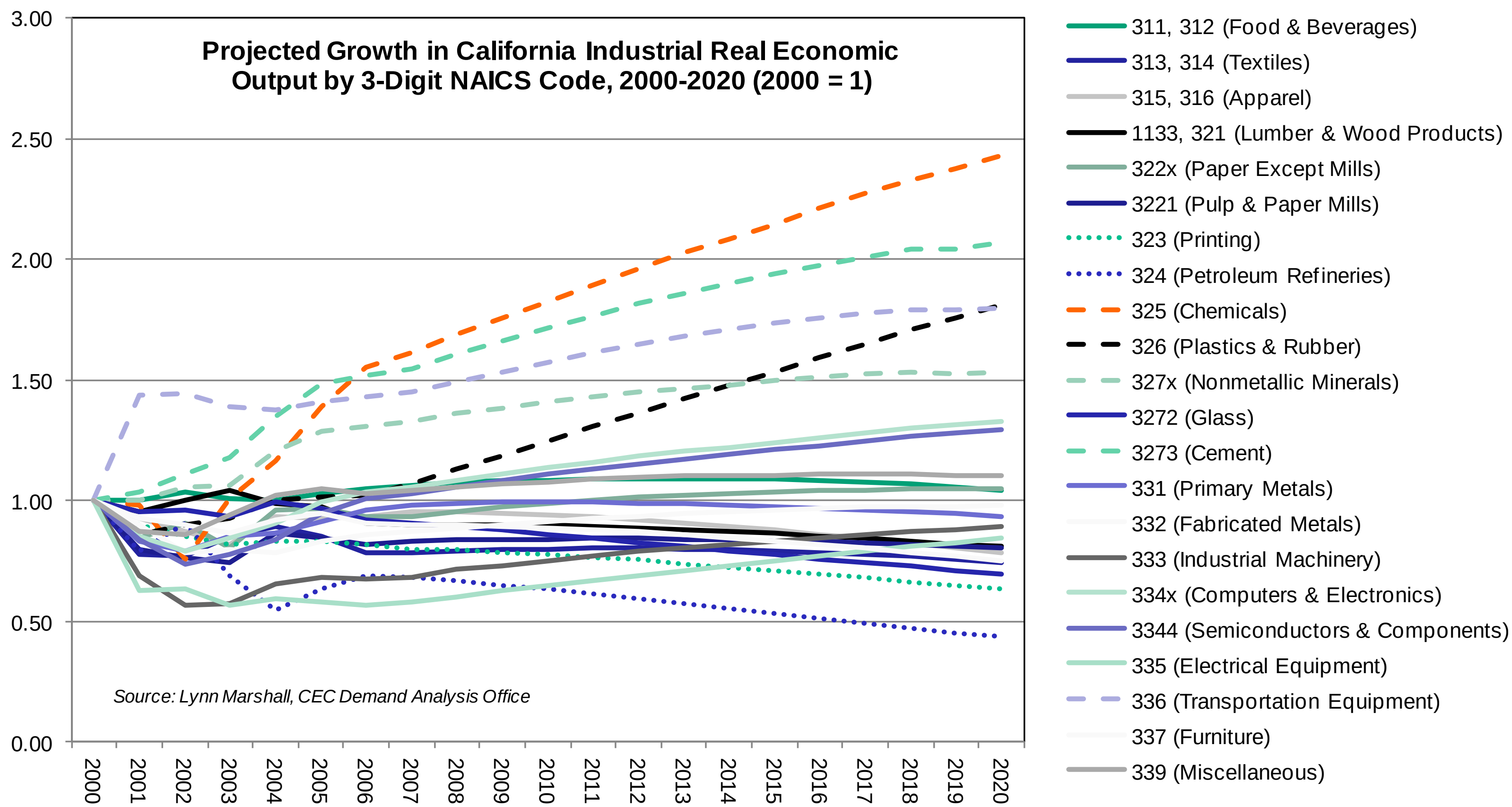


Figure C-5: Projected growth in California industrial output through 2020 by subsector

Sectors of GHG emissions importance to California

Table C-1 presents the data the team used to identify industrial sectors and related products of greatest GHG emissions importance to California supply chains. The table presents approximate 2000 GHG emissions associated with each major California industrial NAICS sector, based on data from the ARB California GHG emissions inventory and the California Energy Balances Database (CALEB). The 2000 emissions estimates include emissions arising from fuel combustion and electricity use by each NAICS sector.

Table C-1: California Industrial GHG Emissions (2000)

	2000		2020	
NAICS Sector (subsector name)	**Tg CO2e**	**% Total**	**Tg CO2e**	**% Total**
3273 (Cement)	10.0	12%	20.7	25%
324 (Petroleum Refineries)	36.5	44%	15.8	19%
325 (Chemicals)	3.6	4%	8.9	11%
339 (Miscellaneous)	7.7	9%	8.5	10%
311, 312 (Food & Beverages)	5.7	7%	6.0	7%
3344 (Semiconductors & Components)	2.9	3%	3.7	5%
336 (Transportation Equipment)	1.5	2%	2.7	3%
334x (Computers & Electronics)	1.9	2%	2.6	3%
326 (Plastics & Rubber)	1.3	2%	2.3	3%
327x (Nonmetallic Minerals)	1.3	2%	2.0	2%
331 (Primary Metals)	1.8	2%	1.6	2%
332 (Fabricated Metals)	1.4	2%	1.4	2%
3221 (Pulp & Paper Mills)	1.5	2%	1.2	1%
322x (Paper Except Mills)	0.8	1%	0.8	1%
3272 (Glass)	1.1	1%	0.7	1%
1133, 321 (Lumber & Wood Products)	0.8	1%	0.7	1%
333 (Industrial Machinery)	0.7	1%	0.6	1%
313, 314 (Textiles)	0.7	1%	0.5	1%
335 (Electrical Equipment)	0.5	1%	0.4	0%
323 (Printing)	0.6	1%	0.4	0%
315, 316 (Apparel)	0.2	0%	0.2	0%
337 (Furniture)	0.2	0%	0.1	0%

The table also presents projections of 2020 GHG emissions by NAICS sector, which are representative of a "frozen efficiency" scenario. In other words, in this scenario 2000 GHG emissions are simply extrapolated to 2020 based on projected real economic growth by NAICS sector through 2020. Although this is a very course approach, it highlights which NAICS sectors are likely to be important GHG emissions sources in the future in the absence of major shifts in operational efficiencies and practices. Moreover, no other projections of California industrial GHG emissions growth over the near-term are available, and the policy landscape that will affect state industrial GHG emissions is still evolving. Thus, the team felt that this coarse approach is justified for the purposes of this study given current forecasting constraints.

Preliminary short list and final product case selections

Based on the above information, the research team identified the short list that appears in Table C-2. This information was presented to CARB; the 22 final product cases chosen by CARB for this study are summarized in Table C-3.

Table C-2: Preliminary short list of products presented to CARB

#	Proposed product (*fast growing product category)	Significant Household GHGs		CA Significance		2020 CA Top 80% Sector	
		Direct	Indirect	Manufacturing	Management	Economic	GHGs
1	Cement (Portland and/or masonry)		x	x			x
2	Paraffin-based lubricants		x	x	x		x
3	Pharmaceutical item		x	x	x	x	x
4	Toiletry item		x	x		x	x
5	Medical instrument	x	x	x		x	x
6	Medical supply item		x	x		x	x
7	Canned tomatoes		x	x	x	x	x
8	Dried vegetable (onion or garlic)		x	x	x	x	x
9	Commercial baked item		x	x	x	x	x
10	Beer	x	x	x		x	x
11	Wine		x	x	x	x	x
12	Beef	x	x	x	x	x	x
13	Cheese	x	x	x	x	x	x
14	Personal computer*	x	x		x	x	x
15	Data storage device*	x	x		x	x	x

#	Proposed product (*fast growing product category)	Significant Household GHGs		CA Significance		2020 CA Top 80% Sector	
		Direct	Indirect	Manufacturing	Management	Economic	GHGs
16	Flat panel TV*	x	x				
17	Plastic bag		x	x		x	
18	Refrigerator/freezer	x	x				
19	Hot water heater	x	x				
20	Apparel item		x	x	x		
21	Light bulb*	x	x				
22	Restaurant		x		x		

Table C-3: Final 22 products selected by CARB

Industry/sector	Product	Industry/sector	Product
Apparel	Men’s dress shirt	Food	Beef
Appliances	CFL		Bread
	Refrigerator		Canned tomatoes
	Water heater		Cheese
Beverages	Beer		Milk
	Soft drink		Chicken
	Wine		Tortillas
Chemicals	Paint	Forestry	Paper towels
Commercial	Restaurant		Wooden cabinet
Electronics	Flat panel TV	Minerals	Masonry cement
	Hard disk drive		
	Personal computer		

Appendix D: Landfill methodology

Table D-1: Methodology for estimating the energy and GHG emissions associated with landfill disposal			
	All products (per ton waste)	**Value**	**Source/derivation**
A	AVG diesel consumption of household collection trucks	9.1 liter/t waste	McDougall et al. (2001); encompasses collection and transport to landfill.
B	AVG diesel consumption at landfill	5.8 liter/t waste	Franklin Associates (1994)
C	Energy content per liter diesel	38.14 MJ/liter	McDougall et al. (2001)
D	**Energy of collection and landfill equipment**	**570 MJ/t waste**	=C*(A+B)
E	Emitted CO2 per liter diesel combusted	3.12 kg CO2e/liter	McDougall et al. (2001)
F	**GHG emissions of collection and landfill equipment**	**46.64 kg CO2e/t waste**	=E*(A+B)
	Biodegradable products (per ton biogenic waste)	**Value**	**Source/derivation**
G	AVG landfill gas generation for biodegradable waste	250 Nm3/t waste	McDougall et al. (2001)
H	Methane % volume of landfill gas	53%	McDougall et al. (2001)
I	Density of methane (20C, 1atm)	.67 kg/m3	IPCC (1996)
J	AVG methane generation for biodegradable waste	88.44 kg/t waste	=G*H*I
K	CO2 % volume of landfill gas	44%	McDougall et al. (2001)
L	Density of CO2 (20C, 1atm)	1.80 kg/m3	IPCC (1996)
M	AVG CO2 generation for biodegradable waste	198.45 kg/t waste	=G*K*L
N	Landfill gas % recovery	75%	EPA (2011a)
O	Electricity production rate from recovered gas	1.5 kWh/Nm3	McDougall et al. (2001)
P	**Electricity generated from recovered gas**	**188 kWh/t waste or 677 MJ/ t waste**	=G*N*O*V; While some California landfills capture only methane, the majority also use it onsite to generate electricity (CEC 2002). For the purposes of this study, it is assumed that all California landfills that capture methane also produce electricity. 1 kWh = 3.6 MJ.
Q	CO2 emissions	342.55 kg/t waste	=M+J*N*(W+X)*44/12; Last term converts carbon in methane to post-combustion carbon dioxide. Note that the value of Q is set equal to zero for paper towels, whose post-combustion CO2

			emissions are assumed to be taken up by the next generation of trees in sustainable forestry practices.
R	Direct methane emissions	36.04 kg/t waste	=J*Y+J(1-N)*(W+X)
S	Total CO2e emitted from landfill	1171 kg CO2e/t waste	= Q+R*23; Last term converts direct methane emissions to 100-year CO2 equivalents
T	Average CO2 emission factor for CA grid electricity	0.4 kg CO2e/kWh	Marnay et al. (2002)
U	CO2 emissions avoided through landfill gas recovery	75 kg CO2e/t waste	=P*T
V	**Net CO2e emissions from landfill**	**1096 kg CO2e/ t waste**	=S-U
	CA Landfill Profile Summary	**Value**	**Source/derivation**
W	Mass % of California waste to landfills with gas-to-energy	67%	CEC (2002); CIWMB (2005)
X	Mass % of California waste to landfills where gas is flared	12%	
Y	Mass % of California waste to landfills where gas is vented	21%	

Appendix E: Product data derivations

Table E-1: Derivation of producer price to mass/product conversions		
Industry/sector	**Product**	**Derivation**
Apparel	Men's dress shirt	- U.S. Census (2009e) reports U.S. value of shipments of $74 million for 2.6 million men's and boy's woven dress and business shirts produced in 2009 (2002 data not reported). - Using the above data, the average 2009 producer price is estimated at $29 per men's dress shirt. - Using a 2002:2009 producer price index ratio of 126/129 for apparel from U.S. Census (2004c), the estimated average 2002 producer price for a men's dress shirt is $28.
Appliances	CFL	- The U.S. EPA (2009c) estimates that a 15w CFL cost $3.40 (U.S. retail) in 2008. - U.S. BLS (2010) estimates an average producer price to retail price ratio for electric lighting equipment of 50%, which suggests a 2008 CFL producer price of $1.70. - Using a 2002:2008 producer price index ratio of 110/109 for electric lamp bulb manufacturing from U.S. Census (2004c), the estimated average 2002 producer price for a 15w CFL is $1.72.
	Refrigerator	- U.S. Census (2009c) reports U.S. value of shipments of $5.9 billion for 10.4 million household refrigerators produced in 2008 (2002 data not reported). - Using the above data, the average 2008 producer price is estimated at $564 per household refrigerator. - Using a 2002:2008 producer price index ratio of 104/107 for household appliances from U.S. Census (2004c), the estimated average 2002 producer price for a household refrigerator is $548.
	Water heater	- U.S. Census (2009c) reports U.S. value of shipments of $756 million for 3.8 million storage type hot water heaters (34 to 54 gallon capacities) produced in 2008 (2002 data not reported). - Using the above data, the average 2008 producer price is estimated at $197 per storage type hot water heater. - Using a 2002:2008 producer price index ratio of 104/107 for household appliances from U.S. Census (2004c), the estimated average 2002 producer price for a storage type hot water heater is $191.
Beverages	Beer	- U.S. Census (2005a) reports U.S. value of shipments of $4.26 billion for 549 million cases of

Table E-1: Derivation of producer price to mass/product conversions		
Industry/sector	**Product**	**Derivation**
		non-returnable 12 ounce bottles of beer produced in 1997 (2002 data were withheld due to data disclosure issues). - Assuming 24 bottles per case, 1997 U.S. producer price per 12 ounce bottle is estimated at $0.33. - Using a 2002:1997 producer price index ratio of 147/135 for alcoholic beverages from U.S. Census (2004c), the estimated average 2002 producer price for a 12 ounce bottle is $0.36.
	Soft drink	- U.S. Census (1999a) reports U.S. value of shipments of $7.6 billion for 436 billion ounces of carbonated soft drinks in plastic bottles in 1997 (2002 data not reported). - Using the above data, the average 1997 producer price is estimated at $0.28 per 16 ounces of carbonated soft drinks in plastic bottles in 1997. - Using a 2002:1997 producer price index ratio of 151/133 for soft drinks from U.S. Census (2004c), the estimated average 2002 producer price for a 16 ounce bottle is $0.32.
	Wine	- U.S. Census (2004a) reports U.S. value of shipments of $8.3 billion for wines produced in 2002. - The Wine Institute (2010) reports total U.S. production of wines in 2002 of 552 million gallons. - Using the above data, the average 2002 producer price is estimated at $15.02 per gallon (or $2.98 per 750 ml bottle) of wine.
Chemicals	Paint	- U.S. Census (2009a) reports U.S. value of shipments of $18.5 billion for 1.2 billion gallons of paint, varnish, and lacquer produced in 2008 (2002 data not reported). - Using the above data, the average 2008 producer price is estimated at $15.36 per gallon of paint, varnish, and lacquer. - Using a 2002:2008 producer price index ratio of 161/222 for prepared paints from U.S. Census (2004c), the estimated average 2002 producer price for a gallon of paint, varnish, and lacquer is $11.14.
Commercial	Restaurant	n/a
Electronics	Flat panel TV	- IHS (2011) reports that second quarter 2011 global sales of flat panel televisions totaled 48 million units, with industry revenues of $31.5 billion. - Using the above data, the average 2011 producer price is estimated at $656 per flat panel television. - Using a 2002:2009 producer price index ratio of 68/54 (2011 data unavailable) for home

Table E-1: Derivation of producer price to mass/product conversions		
Industry/sector	**Product**	**Derivation**
		electronic equipment from U.S. Census (2004c), the estimated average 2002 producer price for a television is $826.
	Hard disk drive	- U.S. national level data on unit quantities produced are not available; thus, available hard disk drive manufacturer data are used to estimate the 2002 U.S. producer price per hard disk drive. - In 2011, Seagate Technology (a U.S. manufacturer of hard disk drives, and the world's largest) reported 52 million hard drives shipped at $2.9 billion in revenues (Seagate 2011). - Using the above data, the average 2011 producer price of is estimated at $56 per hard disk drive. - Using a 2002:2011 producer price index ratio of 138/65 for computer storage device manufacturing from U.S. BLS (2011a), the estimated average 2002 producer price for a hard disk drive is $119.
	Personal computer	- IT industry analysis firm IDC (2003) reported that fourth quarter 2002 global shipments of personal computers amounted to 38.4 million units at a value of $47 billion. Using these data as representative of the average 2002 PC, the average 2002 producer price for a personal computer is estimated at $1225.
Food	Beef	- The U.S. Department of Agriculture's Economic Research Service (2010a) reports that in 2002, U.S. beef production equaled 27.1 billion pounds. - U.S. Census (2004f) reports U.S. value of shipments of $35.2 billion for beef products produced in 2002. - Using the above data, the average 2002 producer price is estimated at $1.30 per pound of beef, or $2.87 per kg.
	Bread	- U.S. Census (2004h) reports U.S. value of shipments of $3.5 billion for 5.6 billion pounds of white pan bread produced in 1997 (2002 data not reported). - Using the above data, the average 1997 producer price is estimated at $0.63 per pound of white pan bread, or $1.38 per kg. - Using a 2002:1997 producer price index ratio of 190/174 for bakery products from U.S. Census (2004c), the average 2002 producer price is estimated at $0.68 per pound of white pan bread, or $1.51 per kg.
	Canned tomatoes	- U.S. Census (2004j) reports U.S. value of shipments of $619 million for 1.6 billion number 303 cans (0.5 liter) of canned tomatoes produced in 1997 (2002 data not reported).

Table E-1: Derivation of producer price to mass/product conversions		
Industry/sector	**Product**	**Derivation**
		- Using the above date, the average 1997 producer price is estimated at $0.38 per 0.5 liter can of canned tomatoes. - Using a 2002:1997 producer price index ratio of 133/126 for processed fruits and vegetables from U.S. Census (2004c), the average 2002 producer price is estimated at $0.40 per 0.5 liter can.
	Cheese	- U.S. Census (2004d) reports U.S. value of shipments of $12 billion for 8 billion pounds of packaged natural cheeses produced in 1997 (2002 data not reported). - Using a 2002:1997 producer price index ratio of 136/128 for dairy products from U.S. Census (2004c), the estimated average 2002 producer price for a pound of packaged natural cheese is $1.50, or $3.31 per kg.
	Milk	- U.S. Census (2004i) reports U.S. value of shipments of $4.2 billion for 8.1 billion quarts of low fat packaged milk produced in 2002. - Using the above data, the average 2002 producer price is estimated at $0.51 per quart of packaged milk, or $2.04 per gallon of packaged milk.
	Chicken	- U.S. Census (2004g) reports U.S. value of shipments of $4.9 billion for 5.7 billion pounds of tray pack chicken (broilers and fryers) produced in 2002. - Using the above data, the average 2002 producer price is estimated at $0.86 per pound of tray pack chicken, or $1.90 per kg.
	Tortillas	- The Tortilla Industry Association (2001) reports that in 2000, U.S. tortilla industry sales totaled $3.2 billion for $7 billion pounds of flour and corn tortillas. - Using the above data, the average 2000 producer price is estimated at $0.46 per pound of tortillas, or $1.01 per kg. - Using a 2002:2000 producer price index ratio of 190/182 for bakery products from U.S. Census (2004c), the estimated average 2002 producer price for a kg of tortillas is $1.05.
Forestry products	Paper towels	- U.S. Census (2005b) reports U.S. value of shipments of $422 million for 265,000 short tons of packaged paper towels produced in 2002. - Using the above data, the average 2002 producer price is estimated at $1.76 per kg of packaged paper towels.
	Wooden cabinet	- U.S. Census (2004e) reports U.S. value of shipments of $3.9 billion for 29.7 million kitchen cabinets produced in 2002. - Using the above data, the average 2002 producer price is estimated at $131 per cabinet.

Table E-1: Derivation of producer price to mass/product conversions		
Industry/sector	**Product**	**Derivation**
Minerals	Masonry cement	- The U.S. Geological Survey reports that in 2002, the average value of shipments for masonry cement in the United States was $108 per metric ton (USGS 2010).

Table E-2: Derivation of total product purchases in $2002 producer prices		
Industry/sector	**Product**	**Derivation**
Apparel	Men's dress shirt	- It is estimated from U.S. BLS (2011b) survey data that $79 per U.S. household was spent on men's shirts in 2002. - Assuming that the producer price accounts for 48% of the retail price (U.S. BEA 2008) for this product, the estimated 2002 spending per household for men's shirts in producer prices is $38. - Using the above data, total California household spending for men's shirts in 2002 producer prices is estimated at $462 million. - It is estimated from U.S. Bureau of Census (2011c) and (2009e) that men's dress shirts account for one-third of U.S. production of all men's shirts. Using this fraction as a proxy for purchases, the estimated spending for men's dress shirts is $152 million.
Appliances	CFL	- The KEMA (2010) survey estimates that there are at least 200 million light bulbs installed in California households, and that roughly 50% of these are CFLs. - Based on California household lighting wattage data from RLW Analytics (2008), it is estimated that roughly one-third of fixtures are used for 60 watt incandescent or 15 watt CFL bulbs. - Assuming an average CFL lifetime of 7 years (U.S. EPA 2009c), and (based on the above) 33 million 15 watt CFLs installed, annual purchases are estimated at 4.7 million 15 watt CFLs per year. - Based on the average 2002 producer price of CFLs ($1.72 per unit, derived earlier) total California household spending for 15 watt CFLs is estimated at $8.1 million.
	Refrigerator	- U.S. EPA (2011) estimates an average U.S. refrigerator life of 12 years, or an annual turnover rate of 1/12 of the installed stock. - The KEMA (2010) survey estimates that the number of primary use refrigerators installed in California households is 11.5 million. - Based on the above data, it is estimated that 960,000 refrigerators are purchased each year. - Based on the average 2002 producer price of refrigerators ($548 per unit, derived earlier) total California household spending for refrigerators is estimated at $526 million.
	Water heater	- Lu et al. (2011) estimate an average California water heater life of 12 years, or an annual turnover rate of 1/12 of the installed stock. - The KEMA (2010) survey estimates that the number of gas fired storage tank water heaters installed in California households is 7.9 million.

Table E-2: Derivation of total product purchases in $2002 producer prices		
Industry/sector	**Product**	**Derivation**
		- Based on the above data, it is estimated that 660,000 gas fired storage tank water heaters are purchased each year. - Based on the average 2002 producer price of gas fired storage tank units ($191 per unit, derived earlier) total California household spending for tank heaters is estimated at $126 million.
Beverages	Beer	- It is estimated from U.S. BLS (2011b) survey data that $112 per U.S. household was spent on beer in 2002. - Assuming that the producer price accounts for 50% of the retail price (U.S. BEA 2008) for this product, the estimated 2002 spending per household for beer in producer prices is $56. - Using the above data, total California household spending for beer in 2002 producer prices is estimated at $682 million. - It is estimated from U.S. Census (2005a) that bottled beer accounts for 37% of U.S. brewery output. Using this fraction as a proxy for purchases, the estimated spending for beer is $259 million.
	Soft drink	- It is estimated from U.S. BLS (2011b) survey data that $125 per U.S. household was spent on cola and other carbonated beverages in 2002. - Assuming that the producer price accounts for 62% of the retail price (U.S. BEA 2008) for these products, the estimated 2002 spending per household in producer prices is $78. - Using the above data, total California household spending for these products in 2002 producer prices is estimated at $945 million. - It is estimated from U.S. Census (1999a) that carbonated soft drinks in 16 ounce plastic bottles account for 37% of U.S. carbonated beverage production. Using this fraction as a proxy for purchases, the estimated spending for 16 ounce plastic bottled carbonated soft drinks is $345 million.
	Wine	- Total retail bottled wine purchases in California each year equal roughly 36 bottles per household based on data from Wine Institute (2010). - Using an average 2002 producer price of $2.98 per bottle of wine (see earlier derivation), total California household spending for these products in 2002 producer prices is estimated at $1.3 billion.
Chemicals	Paint	- It is estimated from U.S. BLS (2011b) survey data that $14 per U.S. household was spent on paint in 2002.

Table E-2: Derivation of total product purchases in $2002 producer prices		
Industry/sector	**Product**	**Derivation**
		- Assuming that the producer price accounts for 36% of the retail price (U.S. BEA 2008) for this product, the estimated 2002 spending per household for paint in producer prices is $5. - Using the above data, total California household spending for paint in 2002 producer prices is estimated at $61 million.
Commercial	Restaurant	- It is estimated from U.S. BLS (2011b) survey data that $2,276 per U.S. household was spent on eating out at restaurants in 2002. - Using the above data, total California household spending for eating out at restaurants in 2002 producer prices is estimated at $27.7 billion.
Electronics	Flat panel TV	- The KEMA (2010) survey estimates that the number of televisions (all types) installed in California households is 26 million. - Fraunhofer et al. (2007) estimate an average television lifetime of 10 years; based on this assumption, an annual rate of turnover of 1/10 of the installed stock is estimated in California. - Using the above data, it is estimated that 2.6 million televisions are purchased in California each year; furthermore, given the near obsolescence of picture tube TVs, all of these purchases are assumed to be flat panel TVs. - Based on the average 2002 producer price of flat panel TVs ($826 per unit, derived earlier) total California household spending is estimated at $2.2 billion.
	Hard disk drive	- Masanet and Horvath (2006a, 2006b) estimated an average life of 4 years for personal computer equipment in California, or an annual turnover rate of 1/4 of the installed stock. In the absence of data for external hard drives, this turnover rate is used as a proxy. - No data could be found on the number of installed external hard drives in U.S. or California households; thus, the research team estimates 1 in 10 California PCs are equipped with external storage. - The KEMA (2010) survey estimates that the number of desktop control units and laptops installed in California households is 18.2 million. - Based on the above assumptions, it is estimated that 450,000 external hard drives are purchased each year. - Based on the average 2002 producer price of external hard drives ($119 per unit, derived earlier) total California household spending is estimated at $53 million.
	Personal computer	- Masanet and Horvath (2006a, 2006b) estimated an average life of 4 years for desktop control units in California, or an annual turnover rate of 1/4 of the installed stock.

Table E-2: Derivation of total product purchases in $2002 producer prices		
Industry/sector	**Product**	**Derivation**
		- The KEMA (2010) survey estimates that the number of desktop control units installed in California households is 9.6 million. - Based on the above data, it is estimated that 2.4 million desktop PCs are purchased each year. - Based on the average 2002 producer price of desktop control units ($1225 per unit, derived earlier) total California household spending for desktop PCs is estimated at $2.9 billion.
Food	Beef	- It is estimated from U.S. BLS (2011b) survey data that $231 per U.S. household was spent on beef in 2002. - Assuming that the producer price accounts for 64% of the retail price (U.S. BEA 2008) for this product, the estimated 2002 spending per household for beef in producer prices is $148. - Using the above data, total California household spending for beef in 2002 producer prices is estimated at $1.8 billion.
	Bread	- It is estimated from U.S. BLS (2011b) survey data that $35 per U.S. household was spent on white bread in 2002. - Assuming that the producer price accounts for 62% of the retail price (U.S. BEA 2008) for this product, the estimated 2002 spending per household for white bread in producer prices is $22. - Using the above data, total California household spending for white bread in 2002 producer prices is estimated at $264 million. - It is estimated from U.S. Census (2004h) that white pan bread accounts for 76% of U.S. white bread output. Using this fraction as a proxy for purchases, the estimated spending for white pan bread is $201 million.
	Canned tomatoes	- It is estimated from U.S. BLS (2011b) survey data that $83 per U.S. household was spent on canned vegetables in 2002. - Assuming that the producer price accounts for 63% of the retail price (U.S. BEA 2008) for this product, the estimated 2002 spending per household for canned vegetables in producer prices is $52. - Using the above data, total California household spending for canned vegetables in 2002 producer prices is estimated at $637 million. - It is estimated from U.S. Census (2004j) that 0.5 liter (number 303) canned tomatoes account for 27% of U.S. canned vegetable output. Using this fraction as a proxy for purchases, the estimated spending for canned tomatoes is $170 million.

Table E-2: Derivation of total product purchases in $2002 producer prices		
Industry/sector	**Product**	**Derivation**
	Cheese	- It is estimated from U.S. BLS (2011b) survey data that $96 per U.S. household was spent on cheese in 2002. - Assuming that the producer price accounts for 64% of the retail price (U.S. BEA 2008) for this product, the estimated 2002 spending per household for cheese in producer prices is $61. - Using the above data, total California household spending for cheese in 2002 producer prices is estimated at $749 million. - It is estimated from U.S. Census (2004d) that packaged natural cheeses account for 60% of U.S. cheese output. Using this fraction as a proxy for purchases, the estimated spending for packaged natural cheese is $449 million.
	Milk	- It is estimated from U.S. BLS (2011b) survey data that $115 per U.S. household was spent on fresh milk in 2002. - Assuming that the producer price accounts for 68% of the retail price (U.S. BEA 2008) for this product, the estimated 2002 spending per household for fresh milk in producer prices is $78. - Using the above data, total California household spending for fresh milk in 2002 producer prices is estimated at $953 million.
	Chicken	- It is estimated from U.S. BLS (2011b) survey data that $81 per U.S. household was spent on fresh and frozen tray packed chicken in 2002. - Assuming that the producer price accounts for 68% of the retail price (U.S. BEA 2008) for this product, the estimated 2002 spending per household for tray packed chicken in producer prices is $55. - Using the above data, total California household spending for tray packed chicken in 2002 producer prices is estimated at $671 million.
	Tortillas	- The Tortilla Industry Association (2001) reports that in 2000, U.S. tortilla industry sales totaled 7 billion pounds of flour and corn tortillas, or 62 pounds (28 kg) of tortillas per U.S. household per year. - Assuming this per-household consumption holds for current California households, it is estimated that 341 million kg of tortillas were purchased in the state. - Based on the average 2002 producer price of tortillas ($1.05 per kg, derived earlier) total California household spending for tortillas is estimated at $358 million.
Forestry products	Paper towels	- It is estimated from U.S. BLS (2011b) survey data that $76 per U.S. household was spent on sanitary paper products in 2002.

Table E-2: Derivation of total product purchases in $2002 producer prices		
Industry/sector	**Product**	**Derivation**
		- Assuming that the producer price accounts for 65% of the retail price (U.S. BEA 2008) for this product, the estimated 2002 spending per household for sanitary paper products in producer prices is $49. - Using the above data, total California household spending for sanitary paper products in 2002 producer prices is estimated at $602 million. - It is estimated from U.S. Census (2005b) that paper towels account for 13% of U.S. sanitary paper output. Using this fraction as a proxy for purchases, the estimated spending for paper towels is $77 million.
	Wooden cabinet	- Furniture industry data suggest that the average lifetime of a kitchen cabinet is 17.5 years (This Old House 2011). - Using this assumption, it is estimated that 1/17.5 (6%) of kitchen cabinets installed in California households is replaced each year. - Based on the mass conversion data derived in this study for kitchen cabinets (i.e., the average California household has 204 kg of kitchen cabinets installed), it is estimated that 149 million kg of kitchen cabinets are currently purchased each year. - Based on the average 2002 producer price of cabinets ($5.70 per kg, derived earlier) total California household spending for kitchen cabinets is estimated at $850 million.
Minerals	Masonry cement	- Data from USGS (2010) indicate that 517,000 metric tons of masonry cement were shipped to final customers in California in 2002. - Assuming a 2002 producer price of $108 per metric ton (derived earlier), total California retail spending for masonry cement is estimated at $56 million.

Table E-3: Product unit mass assumptions			
Industry/sector	**Product**	**Unit**	**Derivation**
Apparel	Men's dress shirt	1 dress shirt	- An average size large men's button down dress shirt is estimated at 0.6 pounds (0.3 kg) based on shipping weights for different shirts (cotton and cotton blends) obtained from online retailers.
Appliances	CFL	1 15w CFL	- An average 15w CFL is estimated to weigh 0.3 pounds (0.14 kg) based on shipping weights obtained for spiral, tube, and bulb type CFLs from online retailers.
	Refrigerator	1 refrigerator	- Based on data in RLW Analytics (2008), the average refrigerator size in California is estimated at 22 cubic feet. - An average 22 cubic foot refrigerator is estimated to weigh 250 pounds (133 kg) based on shipping weights obtained for top freezer, side by side, and bottom freezer models from online retailers.
	Water heater	Gas fired tank water heater	- According to RLW Analytics (2008), the average gas-fired tank water heater size in California is 40 gallons. - Lu et al. (2008) reports an average mass of 53 kg for a 40 gallon tank water heater.
Beverages	Beer	12 oz. bottle	- The average weight of a 12 ounce glass bottled beer was estimated by weighing several brands on a digital scale; the average mass per bottle was estimated at 0.55 kg. - The average weight of an empty 12 ounce glass bottle was estimated by weighing several brands on a digital scale; the average mass was estimated at 200 grams per empty glass bottle.
	Soft drink	16 oz. bottle	- The average mass of a 16 ounce bottle of carbonated soft drink was estimated by weighing both diet and sugar types (weight varies based on sugar content) on a digital scale; the average mass per bottle was estimated at 0.4 kg. - According to NAPCOR (2011), the average empty 16 oz plastic bottle weighs 0.9 ounces, or 25 grams.
	Wine	750 ml bottle	- An average glass bottle mass of 500 grams per 750 milliliters of wine (i.e., the standard wine bottle volume) was assumed based on recent wine industry packaging studies from the UK's Waste and Resources Action Programme (WRAP) program (WRAP 2008, 2010).

Table E-3: Product unit mass assumptions			
Industry/sector	**Product**	**Unit**	**Derivation**
			- The density of wine was assumed to be 975 g/liter (i.e., slightly less dense than water due to its alcohol content) which led to an estimated product mass (glass bottle plus wine) of 1.48 kg.
Chemicals	Paint	1 gallon	- One gallon of paint, packaged in steel container, was weighed on a digital scale; mass = 9.1 lb, or 4.1 kg. - An empty one gallon steel paint container was weighed on a digital scale; mass = 1.3 lb, or 0.6 kg.
Commercial	Restaurant	$ 2002	n/a
Electronics	Flat panel TV	1 LCD TV	- The KEMA (2010) survey estimates that the number of small LCD TVs installed in California homes less than 36 inches is 5.2 million, and the number of large LCD TVs larger than 36 inches is 4.7 million, for a total of 9.8 million LCD TVs. - Fraunhofer et al. (2007) reports a total mass of 7.2 kg for a 32 inch LCD TV and 11.8 kg for a 42 inch LCD TV. Using these values for the average California LCD TV less than 36 inches, and greater than 36 inches, respectively, a weighted average mass for a California LCD TV is estimated at 9.4 kg.
	Hard disk drive	1 external drive	- Based on product mass data provided for a number of external storage drive models from Seagate (2011), the average mass of an external storage drive (desktop model) was estimated at 1 kg.
	Personal computer	1 control unit	- Masanet and Horvath (2006a, 2006b) estimated an average unit mass of 9 kg for desktop control units in California.
Food	Beef	1 kg	n/a
	Bread	1 kg	n/a
	Canned tomatoes	1 0.5 liter can	- An average 0.5 liter can of canned tomatoes is estimated to weigh 0.95 pounds (0.43 kg) based on shipping weights obtained for canned tomatoes (stewed, diced, sauces, and pastes) from online grocers. - An empty number 303 steel can was weighed on a digital scale; mass = 60 grams, or about 13% of product weight.
	Cheese	1 kg	n/a

Table E-3: Product unit mass assumptions			
Industry/sector	**Product**	**Unit**	**Derivation**
	Milk	1 gallon	- One gallon of milk, packaged in an HDPE bottle, was weighed on a digital scale; mass = 8.4 lb, or 3.8 kg. - The empty HDPE bottle weighed 60 grams.
	Chicken	1 kg	n/a
	Tortillas	1 kg	n/a
Forestry products	Paper towels	1 kg	n/a
	Wooden cabinet	1 cabinet	- The KEMA (2010) survey estimates that the average living area of all California dwellings is 1,591 square feet. - Using data from California MLS listings, the average kitchen size is estimated at 15% of floor area, or 240 square feet. - Assuming a 20 foot by 12 foot average kitchen, with 50% devoted to eating space, an estimated 10 foot by 6 foot area would have base and wall cabinetry on two walls. - Assuming an average cabinet depth of 12 inches, an average base cabinet height of 36 inches, and an average wall cabinet height of 30 inches (from online hardware store data), roughly 5 base cabinets (allowing for a range) and 4 wall cabinets (allowing for a range and a sink window) of 24 inch width might comprise the average kitchen cabinet installation. - Using shipping mass data for 24 inch width wooden cabinets from online hardware store data, it is estimated that the total kitchen cabinetry mass is 450 lbs, or 204 kg. - Using the above data, the average single cabinet mass is 23 kg.
Minerals	Masonry cement	Metric ton	n/a

Table E-4: Baseline scenario use phase assumptions			
Industry/sector	**Product**	**Unit**	**Derivation**
Apparel	Men's dress shirt	1 dress shirt	- KEMA (2010) estimates that the average California household will wash and dry around 3 loads of laundry per week; this source further estimates that the average combined UECs for laundry equipment (washing machine plus dryer) are 250 kWh/year and 11 therms/year. - Assuming an average load weight of 10 pounds (based on a review of washer and dryer capacities from online retailers), and that a 0.3 kg men's dress shirt (see product mass assumption derived earlier) will be washed once every two weeks, the per-shirt UECs are estimated at 2 kWh/yr and 0.1 therm/yr. - Based on data in DLI (2011), the typical lifespan of a dress shirt is estimated at 50 washings; based on these data, this study estimates an average lifetime UEC per shirt of 100 kWh and 5 therms.
Appliances	CFL	1 15w CFL	- U.S. EPA (2009c) estimates an average UEC for 15 watt CFLs in U.S. households of 16.4 kWh/yr
	Refrigerator	1 refrigerator	- KEMA (2010) estimates an average UEC for primary refrigerators in California households of 772 kWh/yr
	Water heater	Gas fired tank water heater	- KEMA (2010) estimates an average UEC for standard gas fired water heaters in California households of 195 therms/yr.
Beverages	Beer	12 oz. bottle	- KEMA (2010) estimates an average UEC for primary refrigerators in California households of 772 kWh/yr. - Based on data in RLW Analytics (2008), the average refrigerator size in California is estimated at 22 cubic feet (623 liters). - Assuming an average 12 ounce bottle will occupy roughly 0.5 liters of refrigerated space for two weeks, and that on average only 50% of refrigerated space is occupied, the average refrigeration energy use is estimated at 0.04 kWh/yr.
	Soft drink	16 oz. bottle	- See assumptions for refrigerator UEC and average volume for beer. - Assuming an average 16 ounce bottle will occupy roughly 0.75 liters of refrigerated space for two weeks, and that on average only 50% of refrigerated space is occupied, the average refrigeration energy use is estimated at 0.06 kWh/yr.
	Wine	750 ml bottle	- See assumptions for refrigerator UEC and average volume for beer.

Table E-4: Baseline scenario use phase assumptions			
Industry/sector	**Product**	**Unit**	**Derivation**
			- Assuming an average 750 ml bottle will occupy roughly 1 liter of refrigerated space for two weeks, and that on average only 50% of refrigerated space is occupied, the average refrigeration energy use is estimated at 0.1 kWh/yr.
Electronics	Flat panel TV	1 LCD TV	- Fraunhofer et al. (2007) estimates a UEC of 230 kWh/yr for a 32 inch LCD TV. - Based on online retailer data for 42 inch LCD power use, and the use pattern assumptions listed in Fraunhofer et al. (2007), an average UEC of 350 kWh/yr is estimated for a 42 inch LCD TV. - Using the same weighted averaging approach for these two LCD sizes as for average mass, the estimated UEC of flat panel TVs in California homes is 290 kWh/yr.
	Hard disk drive	1 external drive	- Data on the average energy use of home external hard drives could not be found in the public domain; as a proxy, the UEC is estimated based on external drive energy use information from data centers. - Masanet et al. (2011) report an average UEC per external data center hard drive of 237 kWh/yr for continuous operation. - Assuming that the average external hard drive in California homes will be operated for 25% of the time the average home PC is powered on (which is estimated at 1600 hours per year (IVF 2007)), the average external drive UEC is estimated at 11 kWh/yr.
	Personal computer	1 control unit	- The KEMA (2010) survey estimates that the number of desktop control units installed in California households is 9.6 million, and the number of laptops is 8.7 million. - The KEMA (2010) survey further estimates an average UEC for PCs in California households of around 600 kWh/yr. - Multiplying the average household UEC by the survey population in the KEMA (2010) study (11.5 million), and dividing by the total number of installed PCs (18.3 million), this study estimates an average desktop PC UEC of 380 kWh/yr.
Food	Beef	1 kg	- See assumptions for refrigerator UEC and average volume for beer. - Assuming two cups of raw beef weigh 1 pound (from online nutrition data), 1 kg of beef is estimated to occupy 1 liter of refrigerated space. - Assuming this space is occupied for two weeks, and that on average only 50%

Table E-4: Baseline scenario use phase assumptions			
Industry/sector	**Product**	**Unit**	**Derivation**
			of refrigerated space is occupied, the average refrigeration energy use is estimated at 0.05 kWh/yr.
	Cheese	1 kg	- See assumptions for refrigerator UEC and average volume for beer. - Assuming one cubic inch of cheese weighs 17 g (from online nutrition data), 1 kg of cheese is estimated to occupy 1 liter of refrigerated space. - Assuming this space is occupied for two weeks, and that on average only 50% of refrigerated space is occupied, the average refrigeration energy use is estimated at 0.05 kWh/yr.
	Milk	1 gallon	- See assumptions for refrigerator UEC and average volume for beer. - Assuming an average 1 gallon container will occupy roughly 3.8 liters of refrigerated space for one week, and that on average only 50% of refrigerated space is occupied, the average refrigeration energy use is estimated at 0.2 kWh/yr.
	Chicken	1 kg	- See assumptions for refrigerator UEC and average volume for beer. - Assuming two cups of raw chicken weigh 1 pound (from online nutrition data), 1 kg of chicken is estimated to occupy 1 liter of refrigerated space. - Assuming this space is occupied for two weeks, and that on average only 50% of refrigerated space is occupied, the average refrigeration energy use is estimated at 0.05 kWh/yr.

Table E-5: Baseline scenario end of life assumptions			
Industry/sector	**Product**	**Unit**	**Derivation**
Apparel	Men's dress shirt	1 dress shirt	- No data on final disposition of clothing from California households could be found; thus, it is assumed that all clothing will eventually end up in a landfill after one or more cycles of reuse. - In the absence of consistent data on the material makeup of men's dress shirts, it is assumed that 80% of shirt mass is comprised of cotton and 20% of shirt mass is comprised of non-biodegradable materials (polyester, plastic buttons, etc.). - As a further simplifying assumption, it is estimated that annual discards of men's dress shirts is equal to the annual purchased amounts (i.e., that a new shirt will replace an old shirt).
Appliances	CFL	1 15w CFL	- It is assumed that, under California law, all discarded CFLs will be recycled. - Based on bill of materials data for CFLs in VITO (2009) and the U.S. EPA's WARM model (U.S. EPA 2010a), an average recycling credit of 1.5 Mg CO2e per short ton of CFLs recycled is estimated.
	Refrigerator	1 refrigerator	- It is assumed that, under California appliance recycling law, all discarded refrigerators will be recycled. - Based on bill of materials data for refrigerators in ISIS (2007) – which estimate that the mass fractions of ferrous metals, nonferrous metals, and plastics in an average refrigerator are 48%, 5%, and 32%, respectively – and the recycling energy and GHG emissions factors associated with these materials from the U.S. EPA's WARM model (U.S. EPA 2010a), an average recycling credit of 2.45 Mg CO2e per short ton of refrigerators recycled is estimated.
	Water heater	Gas fired tank water heater	- It is assumed that, under California appliance recycling law, all discarded water heaters will be recycled. - Based on bill of materials data for gas fired tank water heaters in Lu et al. (2011) – which estimate that the mass fractions of ferrous metals, nonferrous metals, and plastics in an average refrigerator are 55%, 10%, and 35%, respectively – and the recycling energy and GHG emissions factors associated with these materials from the U.S. EPA's WARM model (U.S. EPA 2010a), an average recycling credit of 1.99 Mg CO2e per short ton of water

Table E-5: Baseline scenario end of life assumptions			
Industry/sector	**Product**	**Unit**	**Derivation**
			heaters recycled is estimated.
Beverages	Beer	12 oz. bottle	- According to CalRecycle (2011), the current rate of glass beverage container recycling is 85%. The remaining 15% is assumed to be sent to landfill.
	Soft drink	16 oz. bottle	- According to CalRecycle (2011), the current rate of PET beverage container recycling is 85%. The remaining 15% is assumed to be sent to landfill.
	Wine	750 ml bottle	- Unlike glass beer bottles, glass wine bottles in California are not subject to the California Redemption Value (CRV) fee at time of purchase. Consumers receive CRV refunds when they redeem the containers at a recycling center, which provides a strong incentive for recycling. - According to recent data from the U.S. EPA, in the United States roughly only 25% of glass is recovered for recycling while the remaining 75% is most likely sent to landfill (U.S. EPA 2008). - In the absence of California-specific data, the assumed landfill and recycled fractions of waste wine bottles are 75% and 25%, respectively.
Chemicals	Paint	1 gallon	- A steel can recycling rate of 63% is assumed for the paint can portion (600 grams, or 15%, see above) of the purchased product mass (CMI 2011). The remaining 37% of paint cans is assumed to be sent to landfill.
Commercial	Restaurant	$ 2002	n/a
Electronics	Flat panel TV	1 LCD TV	- It is assumed that, under California law, all discarded flat panel televisions will be recycled. - Based on bill of materials data for LCD TVs in IVF (2007) and the U.S. EPA's WARM model (U.S. EPA 2010a), an average recycling credit of 1.38 Mg CO2e per short ton of LCD TVs recycled is estimated.
	Hard disk drive	1 external drive	- It is assumed that, under California law, all discarded external drives will be recycled. - In the absence of mass composition data for typical external hard drives, as a proxy for hard drives this study adopts the U.S. EPA's WARM model's (U.S. EPA 2010a) average recycling credit of 2.26 Mg CO2e per short ton of personal computers recycled.
	Personal computer	1 control unit	- It is assumed that, under California law, all discarded personal computers will be recycled.

Table E-5: Baseline scenario end of life assumptions			
Industry/sector	**Product**	**Unit**	**Derivation**
			- This study adopts the U.S. EPA's WARM model's (U.S. EPA 2010a) average recycling credit of 2.26 Mg CO2e per short ton of personal computers recycled.
Special note on all food items below			- Hall et al. (2009) have estimated that 40% of U.S. food is wasted; in the absence of food waste data on specific food items, it is assumed that this 40% waste estimate applies to all food products in this study. - CalRecycle (2010) reports that only roughly 15% of California composting facilities handle food scraps as feedstocks, and suggests that food scrap composting outside the home is still quite limited in the state. - In the absence of more precise data, it is estimated that 15% of discarded food waste in California is composted, and that the remaining 85% is sent to landfill. - These assumptions apply to beef, bread, canned tomatoes, cheese, chicken, and tortillas; see data assumptions by food type.
Food	Beef	1 kg	- 32% of purchased mass sent to landfill as waste; 8% is composted
	Bread	1 kg	- 32% of purchased mass sent to landfill as waste; 8% is composted
	Canned tomatoes	1 0.5 liter can	- Of the purchased product mass, it is estimated that 13% is comprised of the metal can (see product mass assumptions above) and 87% is comprised of the tomato product. - Assuming 40% food waste, 35% of the purchased mass is generated as food scrap. - Based on the general assumptions for food items above, 30% of purchased mass is sent to landfill as waste and 5% is composted - A steel can recycling rate of 63% is assumed for the steel can portion (13%) of the purchased product mass (CMI 2011).
	Cheese	1 kg	- 32% of purchased mass sent to landfill as waste; 8% is composted
	Milk	1 gallon	- Plastic milk bottles in California are not subject to the California Redemption Value (CRV) fee at time of purchase. Consumers receive CRV refunds when they redeem the containers at a recycling center, which provides a strong incentive for recycling. - A recycling rate of 19% for HDPE bottles is assumed for the bottle portion

Table E-5: Baseline scenario end of life assumptions			
Industry/sector	**Product**	**Unit**	**Derivation**
			(2%) of the purchased product mass (Miller 2004).
	Chicken	1 kg	- 32% of purchased mass sent to landfill as waste; 8% is composted
	Tortillas	1 kg	- 32% of purchased mass sent to landfill as waste; 8% is composted
Forestry products	Paper towels	1 kg	- In the absence of composting data for paper towels, the same composting rate as for food scraps (15%, see above) is assumed; 85% of purchased mass is assumed to be sent to landfill.
	Wooden cabinet	1 cabinet	- In the absence of data on disposition paths for wooden cabinets in California, all cabinets are assumed to be sent to landfill when discarded.
Minerals	Masonry cement	Metric ton	- As a building material, it is likely that most applied masonry cement will stay in place over several decades. In the absence of data on the current installed amount of masonry cement in California, or on the average lifetime of masonry cement, as a simplifying assumption it is estimated that masonry cement disposal each year is equal to masonry cement purchases (i.e., that all purchases are to replace previous masonry cement applications).

Table E-6: Low carbon technical potential scenario assumptions: Final manufacturing		
Industry/sector	**Product**	**Approach synopsis**
Apparel	Men's dress shirt	- Given that the textile sector is the dominant source of energy use and greenhouse gas emissions in the value chain for men's dress shirts, best practice energy efficiency was considered for textiles (rather than cut and sew operations) in this study for improved final manufacturing performance. - The LBNL EAGER Textile tool and its supporting information were used to estimate the potential energy savings associated with best practice energy efficiency in textile mills; this tool contains dozens of energy efficiency measures using global best practice data (Hasanbeigi and Price 2011). - Based on the above information, it was estimated that best practice energy efficiency could save 30% in electricity usage (primarily in machine drive, pump, and compressed air systems) and 25% in fuel usage (primarily in steam, drying, and process heating systems) compared to an industry average textile mill (Hasanbeigi 2011). - In the absence of data on plant differences between the three regions in the MRIO model, electricity use is reduced by 30% and natural gas use by 25% in the textiles sector for all regions in the MRIO model in the low carbon scenario.
Appliances	CFL	- In the absence of comprehensive plant-level data or case studies for specific product manufacturing within the electronics and appliances manufacturing sectors, in this study general estimates are derived based on sector-level data and energy models from U.S. DOE (2011a) and Masanet et al. (2009b) for these three product classes. The available data and information focus primarily on energy savings through general cross-cutting efficiency measures such as machine drives, pumps, ventilation systems, compressed air systems, and steam systems. - Using the above data sources, it was estimated that best practice energy efficiency would lead to energy savings of 25% for electricity use (primarily in motor, ventilation, and lighting systems) and 20% for natural gas use (primarily for process heating and steam systems). - In the absence of data on plant differences between the three regions in the MRIO model, electricity use is reduced by 25% and natural gas use by 20% in the corresponding electronic, appliance, and electrical equipment sectors for all regions in the MRIO model in the low carbon scenario.
	Refrigerator	
	Water heater	
Beverages	Beer	- The energy savings associated with best practice breweries were estimated based on energy end use data for U.S. breweries in Galitsky et al. (2003), available U.S. plant fuel use data in (U.S. DOE 2011a), and a general energy efficiency potentials model for the beverages sector

Table E-6: Low carbon technical potential scenario assumptions: Final manufacturing		
Industry/sector	**Product**	**Approach synopsis**
		developed in Masanet et al. (2009b). - Based on the above data, it was estimated that a best practice brewery would save 25% in natural gas use and 15% in electricity use compared to the U.S. industry average for breweries. - In the absence of data on plant differences between the three regions in the MRIO model, electricity use is reduced by 15% and natural gas use by 25% in the breweries sector for all regions in the MRIO model in the low carbon scenario.
	Soft drink	- The energy savings associated with best practice soft drink manufacture were estimated based on available U.S. plant fuel use data in (U.S. DOE 2011a) and a general energy efficiency potentials model for the beverages sector developed in Masanet et al. (2009b). - Based on the above data, it was estimated that a best practice soft drink plant would save 20% in natural gas use and 15% in electricity use compared to the U.S. industry average for soft drink manufacture. - In the absence of data on plant differences between the three regions in the MRIO model, electricity use is reduced by 15% and natural gas use by 20% in the soft drink sector for all regions in the MRIO model in the low carbon scenario.
	Wine	- The LBNL BEST-Winery model was used to estimate the potential energy and GHG emissions savings associated with best practice winery efficiency; the model contains dozens of energy efficiency measures for wineries, was developed in coordination with wine industry personnel, and has been tested by California wineries (Galitsky et al. 2005). Based on average data for California wineries, the model suggests that if all available efficiency measures were deployed, natural gas and electricity use could be reduced by around 25%. - Given that the vast majority of U.S. wine production occurs in California (Galitsky et al. 2005), California wineries were used for examples of solar PV installations for (partially) meeting winery electricity demand. At least one winery – Fetzer Vineyards – has published data on a PV installation, which reportedly generates 1.1 GWh of electricity per year (Fetzer 2010). According to Galitsky et al. (2005), approximately 400 GWh of electricity was used in California to produce around 180 million cases of wine, or around 2.2 kWh per case. Based on Fetzer Vineyard's annual production of 2.2 million cases per year (Fetzer 2010), it was estimated that Fetzer generates around 0.5 kWh of electricity from solar PV per case produced. Assuming that this level of PV generation can be replicated across all wineries serving California consumers, it was estimated from the above data that around 25% of winery electricity use can be met by PV in the low carbon case. While this assumption is rough, it is representative of current best

Table E-6: Low carbon technical potential scenario assumptions: Final manufacturing		
Industry/sector	**Product**	**Approach synopsis**
		practices for the adoption of solar PV in wineries. - Based on the 25% PV adoption assumption, the direct electricity emissions coefficient for the wineries sector in the MRIO model was adjusted to reflect an assumed PV emissions intensity of 20 g CO_2/kWh (Fthenakis and Kim 2007) for 25% of its mix.
Chemicals	Paint	- The energy savings associated with best practice paint manufacture were estimated based on energy end use data for U.S. chemicals plants in Neelis et al. (2008), available U.S. plant fuel use data in (U.S. DOE 2011a), and a general energy efficiency potentials model for the chemicals sector developed in Masanet et al. (2009b). - Based on the above data, it was estimated that a best practice chemicals plant would save 20% in natural gas use and 20% in electricity use compared to the U.S. industry average plant. - In the absence of data on plant differences between the three regions in the MRIO model, electricity use is reduced by 20% and natural gas use by 20% in the paint and coatings sector for all regions in the MRIO model in the low carbon scenario.
Commercial	Restaurant	- Itron estimates that the adoption of best practice energy efficient technologies in California restaurants could reduce electricity and natural gas consumption by roughly 10% and 6%, respectively (Masanet et al. 2009a). - In the absence of data on restaurant differences between the three regions in the MRIO model, electricity use was reduced by 10% and natural gas use by 6% in the food services and drinking places sector for all three regions in the MRIO model in the low carbon scenario.
Electronics	Flat panel TV	- In the absence of comprehensive plant-level data or case studies for specific product manufacturing within the electronics and appliances manufacturing sectors, in this study general estimates are derived based on sector-level data and energy models from U.S. DOE (2011a) and Masanet et al. (2009b) for these three product classes. The available data and information focus primarily on energy savings through general cross-cutting efficiency measures such as machine drives, pumps, ventilation systems, compressed air systems, and steam systems. - Using the above data sources, it was estimated that best practice energy efficiency would lead to energy savings of 25% for electricity use (primarily in motor, ventilation, and lighting systems) and 20% for natural gas use (primarily for process heating and steam systems). - In the absence of data on plant differences between the three regions in the MRIO model, electricity use is reduced by 25% and natural gas use by 20% in the corresponding electronic,
	Hard disk drive	
	Personal computer	

Table E-6: Low carbon technical potential scenario assumptions: Final manufacturing		
Industry/sector	**Product**	**Approach synopsis**
		appliance, and electrical equipment sectors for all regions in the MRIO model in the low carbon scenario.
Food	Beef	- The energy savings associated with best practice beef processing were estimated based on energy and mass balance data meat packing from Brown et al. (1996), available U.S. plant fuel use data in (U.S. DOE 2011a), and a general energy efficiency potentials model for the food processing sectors developed in Masanet et al. (2009b). - Based on the above data, it was estimated that a best practice beef processing plant would save 30% in natural gas use and 20% in electricity use compared to the U.S. industry average plant. - In the absence of data on plant differences between the three regions in the MRIO model, electricity use is reduced by 20% and natural gas use by 30% in the beef processing sector for all regions in the MRIO model in the low carbon scenario.
	Bread	- The energy savings associated with best practice commercial baking were estimated based on available U.S. plant fuel use data in (U.S. DOE 2011a) and the U.S. EPA's Energy Performance Indicator (EPI) tool for U.S. cookie and cracker baking plants (U.S. EPA 2011b) (which uses data compiled from all U.S. plants to benchmark relative energy performance). - Based on the above data, it was estimated that a best practice commercial bakeries would save 30% in natural gas use and 20% in electricity use compared to the U.S. industry average. - In the absence of data on plant differences between the three regions in the MRIO model, electricity use is reduced by 20% and natural gas use by 30% in the bread and bakery product sector for all regions in the MRIO model in the low carbon scenario.
	Canned tomatoes	- The energy savings associated with best practice tomato processing were estimated based on energy end use data for tomato processing in Masanet et al. (2007), available U.S. plant fuel use data in (U.S. DOE 2011a), and the U.S. EPA's Energy Performance Indicator (EPI) tool for U.S. tomato processing plants (U.S. EPA 2011b) (which uses data compiled from all U.S. plants to benchmark relative energy performance). - Based on the above data, it was estimated that a best practice tomato processing plant would save 35% in natural gas use and 30% in electricity use compared to the U.S. industry average for tomato processing plants. - In the absence of data on plant differences between the three regions in the MRIO model, electricity use is reduced by 30% and natural gas use by 35% in the fruit and vegetable

Table E-6: Low carbon technical potential scenario assumptions: Final manufacturing		
Industry/sector	**Product**	**Approach synopsis**
		processing sector for all regions in the MRIO model in the low carbon scenario.
	Cheese	- Xu et al. (2009a, 2009b, 2010, 2011) have derived and published best practice specific energy consumption values for fluid milk and cheese manufacturing. Based on data in these sources, it is estimated that best practice cheese manufacture can save 20% in natural gas usage and 25% in electricity usage compared to an industry average plant. - In the absence of data on plant differences between the three regions in the MRIO model, electricity use is reduced by 25% and natural gas use by 20% in the cheese manufacturing sector for all regions in the MRIO model in the low carbon scenario.
	Milk	- Xu et al. (2009a, 2009b, 2010, 2011) have derived and published best practice specific energy consumption values for fluid milk and cheese manufacturing. Based on data in these sources, it is estimated that best practice cheese manufacture can save 30% -- in both electricity and thermal fuel usage – compared to an industry average plant. - In the absence of data on plant differences between the three regions in the MRIO model, electricity use is reduced by 30% and natural gas use by 30% in the fluid milk sector for all regions in the MRIO model in the low carbon scenario.
	Chicken	- The energy savings associated with best practice poultry processing were estimated based on energy and mass balance data meat packing from Brown et al. (1996), available U.S. plant fuel use data in (U.S. DOE 2011a), and a general energy efficiency potentials model for the food processing sectors developed in Masanet et al. (2009b). - Based on the above data, it was estimated that a best practice poultry processing plant would save 30% in natural gas use and 20% in electricity use compared to the U.S. industry average plant. - In the absence of data on plant differences between the three regions in the MRIO model, electricity use is reduced by 20% and natural gas use by 30% in the poultry processing sector for all regions in the MRIO model in the low carbon scenario.
	Tortillas	- Best practice electrical energy savings in tortilla manufacture are estimated at 20% through efficient lighting, variable speed drives, and refrigeration units based on a case study data from a Mission Foods tortilla plant (EDR 2005). - Best practice natural gas savings are estimated at 30% through more efficient steam systems and burners, improved process controls, and improved insulation (Masanet et al. 2009b). - In the absence of data on plant differences between the three regions in the MRIO model,

Table E-6: Low carbon technical potential scenario assumptions: Final manufacturing		
Industry/sector	**Product**	**Approach synopsis**
		electricity use is reduced by 20% and natural gas use by 30% in the tortilla sector for all regions in the MRIO model in the low carbon scenario.
Forestry	Paper towels	- Based on efficient technology data in Kramer et al. (2008), and a U.S. energy balance model for pulp and paper products in the United States from Jacobs and IPST (2006), best practice steam system savings were estimated at 20% (which were applied to direct fuel uses in the paper industry sectors) and best practice electricity savings were estimated at 15% through more efficient motor and lighting systems. - In the absence of data on plant differences between the three regions in the MRIO model, electricity use is reduced by 15% and direct fuel use by 12% in the paper industry sectors for all regions in the MRIO model in the low carbon scenario.
	Wooden cabinet	Brown et al. (1996), U.S. DOE (2011a), Masanet et al. (2009b)
Minerals	Masonry cement	- California cement plants are among the most efficient in the country; all use dry kilns, and all but one employ energy-efficient preheater/precalciners for waste heat recovery (CalTrans 2011). Thus, the efficiency improvement potential for California plants is mostly limited to electrical energy uses, such as the use of efficient roller mills (in raw materials preparation), variable speed drives, and other measures which can reduce plant electricity use by roughly 30% (Galitsky et al. 2008). For reduction of thermal fuels, the only major applicable measure is assumed to be the use of steel slag in the kiln, which is estimated to reduce kiln fuel use by around 5% (Galitsky et al. 2008). For the California cement sector, electricity use is reduced in the MRIO model by 30% and thermal fuel use in process heating is reduced by 5% to represent best practice for final manufacture in the low carbon scenario. - For the “rest of US” cement sector, ENERGY STAR for Industry data from the cement industry Energy Performance Indicator suggest that a best practice cement plant uses roughly 25% less energy than the U.S. average plant (U.S. EPA 2011b).

Table E-7: Low carbon technical potential scenario assumptions: Design assumptions			
Industry/sector	**Product**	**Unit**	**Derivation**
Apparel	Men's dress shirt	1 dress shirt	- This study considers two potential lower carbon practices for materials selection in the design of dress shirts. First, increased use of recycled fibers (e.g., polyesters based on recycled PET flakes) in fabric blends can offset the use of virgin materials in shirt manufacture. Second, is has been suggested that certain engineered fibers/blends with increased hydrophobic characteristics can reduce fabric drying time, thereby saving energy use in the product use phase. - To approximate the effect of increasing the use of recycled fibers by 30%, direct purchases of fabric by the men's and boy's cut and sew apparel sector were reduced by 30% in the MRIO model as an upper bound estimate on savings. - Savings in energy use of drying were estimated at 50% (for a highly hydrophobic blend) using data in Wallace (2002) as a rough guide.
Appliances	CFL	1 15w CFL	- Given that the majority of the energy use and GHG emissions associated with a CFL are in its use phase, the major design improvement opportunities identified in the literature are focused on reducing use phase energy consumption (VITO 2009). Thus, this study does not consider change to materials selections or design features separately from the use phase reductions identified in this study; VITO (2009) suggests that changes in bills of material for different designs have negligible impact compared to the energy savings achieved in the use phase.
	Refrigerator	1 refrigerator	- There are a number of best practice design features that have been recommended for refrigerators, nearly all of which are aimed at reducing operational energy use. Such features include improved controls, variable speed compressors, improved gaskets, high-efficiency heat exchangers, and materials for improved insulation. Given that the use phase dominates the energy and emissions footprints of a refrigerator, the effects of such design changes have typically been expressed in terms of their use phase energy savings (ISIS 2008) without analysis of changes to raw materials. In light of this lack of data on upstream and downstream implications of changes to bills of materials, this study does not consider change to materials selections or design features separately from the use phase reductions identified in this

Table E-7: Low carbon technical potential scenario assumptions: Design assumptions			
Industry/sector	**Product**	**Unit**	**Derivation**
			study.
	Water heater	Gas fired tank water heater	- A natural gas fired tankless water heater is considered as likely lower carbon design alternative to traditional tank storage heaters; based on data in Lu et al. (2011), gas fired tankless heaters may reduce operating energy by 15% compared to tank heaters. - Lu et al. (2011) also estimate that tankless water heaters will be comprised of 66% less steel, 32% less aluminum, 60% less brass, and 100% less rigid polyurethane (among other changes to the bill of materials) compared to a traditional storage tank design. - To approximate the effects of tankless materials reductions and changes to the production energy use and emissions associated with water heaters, the direct purchases of steel, aluminum, brass, polymers, and other materials by the "Other major household appliance manufacturing" sector were adjusted in the MRIO model by the amounts estimated in Lu et al. (2011).
Beverages	Beer	12 oz. bottle	- Similar to the design anaylsis of bottled wine (see below), this study considered two major design strategies for bottled beer. The first is the practice of "bottle light-weighting," which reduces the mass of the glass bottle while still meeting product standards. Based on case study data in WRAP (2011a), achievable mass reductions in glass beer bottles are estimated at 34%. The second practice is to utilize a high level of recycled glass in bottle, which leads to significant energy use reduction in the glass furnace process in bottle manufacture (Worrell et al. 2008). - To estimate savings from bottle light-weighting, the direct brewery purchases from the glass manufacturing sector were reduced by 34% (to estimate savings in glass manufacturing energy use and emissions) and the weight of the average bottle was reduced accordingly by 34% (to estimate savings in transportation energy use and emissions). - To estimate savings from savings from high use of recycled content, a two step approach was used. First, an average recycled content in baseline (i.e., current) bottles of around 25% (U.S. EPA 2008) was assumed; it was further assumed that this recycled content might be raised as high as 75% for best practice, low carbon bottle design (WRAP 2011d). The latter estimate is

Table E-7: Low carbon technical potential scenario assumptions: Design assumptions			
Industry/sector	**Product**	**Unit**	**Derivation**
			based on the achievable practical limit for wine bottles from WRAP (2010a). Second, the team estimated the reduced energy use and GHG emissions associated with high-recyclate glass in the MRIO glass container manufacturing sector using the U.S. EPA's WARM model (U.S. EPA 2006).
	Soft drink	16 oz. bottle	- Similar to glass bottles, major design opportunities for plastic soft drink bottles include the use of recycled PET and minimizing bottle mass through lightweight design techniques. In practice, these two strategies must be balanced, especially for carbonated soft drinks that exert pressure on the bottle. In the absence of performance and quality data on lightweight bottles that also use high levels of recyclate, this study considers the use of 100% recyclate as a singular strategy. - Using PepsiCo's "EcoGreen Bottle" as an example of successful use of 100% recyclate for carbonated beverages (PepsiCo 2011), the best practice design strategy for 16 oz. plastic soft drink bottles is assumed to be the use of 100% recycled plastic. - Reduced energy use and GHG emissions associated with manufacturing high-recyclate bottles in the MRIO plastics material and resin manufacturing sector were estimated using the U.S. EPA's WARM model (U.S. EPA 2006).
	Wine	750 ml bottle	- Given that the final form of bottled wine consists of few components (a glass bottle, wine, a cork, and a label), there are limited opportunities for design changes that will lead to significant life-cycle energy use and GHG emissions reductions. This study considered two major strategies that are being promoted in the global wine industry. The first is the practice of "bottle light-weighting," which reduces the mass of the glass bottle while still meeting product standards. According to WRAP (2008), reducing the mass of a wine bottle from around 500g to around 350g can reduce the life-cycle GHG emissions of a bottle of wine by around 15% due to decreased energy use in glass manufacture and product transport. Fetzer Vineyards in California reported similar life-cycle GHG emissions savings (14%) based on the use of lightweight glass in their bottles (Fetzer 2010). The second practice is to utilize a high level of recycled glass in bottle, which leads to significant energy use reduction in the glass furnace process in bottle

Table E-7: Low carbon technical potential scenario assumptions: Design assumptions			
Industry/sector	**Product**	**Unit**	**Derivation**
			manufacture (Worrell et al. 2008). - To estimate savings from bottle light-weighting, the direct winery purchases from the MRIO glass manufacturing sector were reduced by 30% (to estimate savings in glass manufacturing energy use and emissions) and the weight of the average bottle was reduced from 500g to 350g (to estimate savings in transportation energy use and emissions). - To estimate savings from savings from high use of recycled content, a two step approach was used. First, an average recycled content in baseline (i.e., current) wine bottles of around 25% (U.S. EPA 2008) was assumed; it was further assumed that this recycled content might be raised as high as 79% for best practice, low carbon bottle design. The latter estimate is based on the achievable practical limit for wine bottles from WRAP (2010a). Second, the team estimated the reduced energy use and GHG emissions associated with high-recyclate glass in the MRIO glass container manufacturing sector using the U.S. EPA's WARM model (U.S. EPA 2006).
Chemicals	Paint	1 gallon	- No design-related improvement opportunities could be identified for paint in this study.
Commercial	Restaurant	$ 2002	- The major design-oriented opportunities for restaurants identified in this study are: (1) to maximize the energy efficiency of the restaurant's operations, and (2) to purchase "low carbon" foods, beverages, and other items necessary for food service operations (e.g., tableware, furniture, and linens). Energy efficiency assumptions are considered in the "final manufacture" table, and "low carbon" purchases are estimated via the eSTEP model for all direct purchases as described in the "supply chain" table.
Electronics	Flat panel TV	1 LCD TV	- Fraunhofer (2007) and IVF (2007) provided detailed assessments of life-cycle improvement potentials for televisions and computers, respectively. Nearly all of the suggested improvements are related to improving the use-phase energy efficiency of these products, which is the dominant phase for energy use and emissions from a life cycle perspective. The main non use-phase opportunity identified by Fraunhofer (2007) was improved materials selection for plastic housings to increase the likelihood of plastics recycling
	Hard disk drive	1 external drive	
	Personal computer	1 control unit	

Table E-7: Low carbon technical potential scenario assumptions: Design assumptions			
Industry/sector	**Product**	**Unit**	**Derivation**
			(rather than landfill or energy recovery) at the end of life stage. Given that the recycling infrastructures for ferrous, nonferrous, and precious metals from electronic equipment are fairly well established, plastics remain one of the largest mass fractions that are not typically economically recyclable (Masanet and Horvath 2007). Therefore, this study considers improved recyclability as its main low-carbon design strategy (besides use phase energy efficiency, which is captured in the low carbon use phase assumptions above). - To estimate the effect of improved design for recycling for plastics—and resulting improved recycling rates—the mass fractions of plastics in each device were estimated based on data in Fraunhofer (2007) and IVF (2007), and these mass fractions were assumed to be 100% recycled by applying the appropriate plastics recycling credit from the WARM model.
Food	Beef	1 kg	- Given that most of the life-cycle emissions of packed meats are associated with livestock operations, agricultural operations for feed, and manure management, product design options for meaningful emissions reductions are somewhat limited. Two strategies identified for reduced packaging are the use of vacuum packed bags and lightweight trays. Case study data on tray lightweighting initiatives in the poultry packing sector suggest that through proper design plastic trays might be reduced in mass by 20% (WRAP 2007b). Vacuum packed bags are a complementary strategy for products with sufficient characteristics (e.g., size, shape, weight) for vacuum packing; this packaging strategy can also reduce packaging mass by anywhere from 10-60% (WRAP 2010b). - Assuming that these practices are mutually exclusive, yet applicable to the majority of packed meats, this study estimates that a 20% reduction in plastic packaging is achievable for all packed meats. - To estimate savings from reduced plastic packaging mass, direct meat packing purchases from the MRIO plastics and resins sector were reduced by 20% .
	Bread	1 kg	- According to WRAP (2011c), in the United Kingdom around 32% of purchased bread ends up as waste, and around 80% of this waste results

Table E-7: Low carbon technical potential scenario assumptions: Design assumptions			
Industry/sector	**Product**	**Unit**	**Derivation**
			from packages that have been opened and not fully eaten. One promising strategy suggested by WRAP (2011c) and piloted by a commercial bread baker is to introduce smaller loaves that still contain full-sized slices, thereby providing a full-sized slice alternative to those who won't typically eat a full loaf before it expires. The study estimated that a 15-slice loaf might be ideal for the typical consumer (as opposed to a typical 22-slice loaf). Another promising strategy is to make prominent a "best by" date on the packaging, so that consumers can more readily recognize when it is time to discard uneaten bread. - Assuming that the above design changes might reduce bread waste from the estimated 70% of consumers who routinely throw away unused bread slices (WRAP 2011c), this study estimates that "right sized" loaves and more prominent "best by" dates could reduce bread waste by up to 56%. - To estimate the effect of these two changes, the purchases of consumer purchases of bread were reduced by 30% (i.e., 15/22) for the 70% of consumers who routinely waste bread, and bread waste generation was reduced by 56%.
	Canned tomatoes	1 0.5 liter can	- For economic reasons, the mass of steel cans has been dropping steadily for many years due to continuous can design and manufacturing process improvements (CMI 2011); thus, the baseline steel can used in canned tomato manufacture is expected to already be fairly materials efficient. - However, a Heinz case study suggests that further can mass reductions might be achieved through cans that are designed with thinner ends (WRAP 2011b) on easy open products. Can mass reductions are estimated at 5% (WRAP 2007a). - Based on data in WRAP (2007a), it is estimated that around one-third of canned tomato products might be packaged with easy open ends; thus, steel savings for all canned tomato products is estimated at 2% (roughly 1/3 of 5%). - To estimate savings from can end light-weighting, direct steel purchases from the MRIO metal can, box, and other container manufacturing sector were reduced by 2% (to estimate savings in glass manufacturing energy use

Table E-7: Low carbon technical potential scenario assumptions: Design assumptions			
Industry/sector	**Product**	**Unit**	**Derivation**
			and emissions).
	Cheese	1 kg	- While several case studies of cheese packaging from WRAP (2011e) suggest that resealable pouches can prolong product life, and there are many examples of reduced plastic mass in food packaging, no data could be found in the public domain to credibly model either prolonged life or reduced packaging mass for cheese. Given that packaging represents less than one percent of the total life cycle emissions of cheese, no packaging changes were considered in this study for cheese.
	Milk	1 gallon	- Given that most of the life-cycle emissions of milk are associated with livestock operations, dairy operations, and refrigeration, product design options for meaningful emissions reductions are somewhat limited. One best practice identified was to reduce the mass of plastic used in the milk bottle through innovative designs, such as the elimination of the jug handle, which might reduce the mass of plastic by 10% (Packwire 2011). - To estimate the savings achievable through reduced mass milk jugs, the direct fluid milk manufacturing purchases from the MRIO plastics and resin sector were reduced by 10%, as was the amount of plastic discarded at the end of life stage.
	Chicken	1 kg	- See assumptions for beef; the same reduced packaging assumptions were used as best practice design for packaged chicken products.
	Tortillas	1 kg	- In the absence of data on lower-carbon design options for tortillas, this study applies the two identified practices for bread (i.e., right sized packages and more prominent “best by” dates) to tortillas as reasonable proxies.
Forestry products	Paper towels	1 kg	- Two major design improvement opportunities for paper towels were identified in this study. First, the use of 100% recycled fibers in paper towels can reduce the energy use and GHG emissions associated with paper production (Kramer et al. 2008) and is an option that is already available on the market. Second, the use of half-sheet designs (i.e., with twice as many perforated breaks in the roll leading to smaller sheet size if desired) might lead to longer lasting rolls, if consumers choose to use a half sheet when a full sheet is not really needed.

Table E-7: Low carbon technical potential scenario assumptions: Design assumptions			
Industry/sector	**Product**	**Unit**	**Derivation**
			- To estimate savings from high use of 100% recycled content, a two step approach was used. First, an average recycled content in baseline (i.e., current) paper towels of 25% was assumed. Second, the team estimated the reduced energy use and GHG emissions associated with the use of recycled fiber feedstocks in the MRIO sanitary paper manufacturing sector using paper industry energy data from Jacobs and IPST (2006) and Kramer et al. (2008), and relative savings associated with different levels of recycled content for different paper types from the Environmental Paper Network (2011) online paper calculator. - No credible data in the public domain could be found on the potential consumer paper towel savings associated with half-sheet rolls; thus, the team estimated that, on average, a half-sheet roll will last 10% longer than a full sheet roll. - To approximate the estimated effect of longer-lasting rolls, annual consumer purchases of paper towels were reduced by 10% in the low carbon scenario.
	Wooden cabinet	1 cabinet	- Many kitchen cabinets are replaced before their technical lifetime has been reached, due to reasons of style and aesthetics; for example, in a case study of cabinet manufacturer Kambium, Liedtke et al. (1998) state that many of its customers preferred to replace their cabinets every seven to ten years, when the technical life of a wooden cabinet is much longer. - This study considers the "product as service" concept as the major design improvement for kitchen cabinets, which changes the business model from one of selling cabinets to end users to one of leasing units that the cabinet maker can take back when the customer is no longer satisfied with the style or performance of the cabinets. A number of successful examples of this business model have been cited in the literature, for such products as carpets (Interface) and copiers (Xerox). The net result is that a manufacturer will design products that are more durable and suitable to refurbishment, since it is in their best interest to minimize the material costs of providing an extended service to the customer. - In this study, it is estimated that standardized, modular cabinets that are

Table E-7: Low carbon technical potential scenario assumptions: Design assumptions			
Industry/sector	**Product**	**Unit**	**Derivation**
			leased and can be “taken back” and refurbished by the manufacturer (e.g., through sanding, new doors and shelves, new veneers, and other refurbishment techniques) would reduce the virgin materials intensity of the cabinet life-cycle by 75% (assuming that refurbishment would require only 25% virgin materials). - To approximate the estimated effect of refurbished cabinets, annual direct purchases from the MRIO sawmills and reconstituted woods sectors were reduced by 75% in the low carbon scenario, as was the amount of generated waste at the end of life phase.
Minerals	Masonry cement	Metric ton	- No design-related improvement opportunities could be identified for masonry cement in this study.

Table E-8: Low carbon technical potential scenario: Use phase assumptions		
Industry/sector	**Product**	**Derivation**
Apparel	Men's dress shirt	- Savings in energy use of drying were estimated at 50% (for a highly hydrophobic blend) using data in Wallace (2002) as a rough guide.
Appliances	CFL	- The 15 watt CFL was chosen based on its equivalence to, and replacement potential for, standard 60 watt incandescent lamps. Equivalence is expressed on a lumens output basis, which is roughly 900 lumens for a 60 watt incandescent or an average 15 watt CFL. - Based on the above, the lumens per watt efficacy for an average 15 watt CFL is 60 (900 lumens/15 watts) - Using the September 2011 list of qualified ENERGY STAR CFLs (U.S. EPA 2011c), and a minimum lifetime of 10,000 hours, the best performing CFL with at least 900 lumens output has a lumens per watt efficacy of 74.1. - Using the lumen per watt efficacy of 74.1 as the current best practice technology, the technical potential for energy savings in 900 lumen CFLs is estimated at 20%.
	Refrigerator	- North (2008) and Itron and KEMA (2008) estimated an average savings of 15% across California primary refrigerators if they were upgraded to ENERGY STAR qualified refrigerators. - Using the September 2011 list of qualified ENERGY STAR refrigerators of 22 cubic feet capacity (the average size assumed in this study), the best practice model consumes roughly 15% less than the ENERGY STAR average energy consumption for this size class (U.S. EPA 2001d). - Based on the above, the technical potential for energy savings from adopting best practice technology for primary refrigerators is estimated at 30%.
	Water heater	- Lu et al. (2011) estimated the annual use phase energy savings associated with switching from a standard gas fired tank water heater to a whole house gas fired tankless heater in California homes. Annual energy savings – including savings in natural gas usage for water heating, and increased electricity use for tankless models – was estimated at 15% (average for Northern and Southern California) on a unit to unit basis. - Based on the findings of Lu et al. (2011), the technical potential for energy savings from switching from gas fired tank to gas fired tankless water heaters in California homes is estimated at 15%.
Electronics	Flat panel TV	- In 2009, the California Energy Commission enacted energy efficiency standards for flat panel televisions. These standards will set a minimum efficiency requirement for all flat panels sold in California starting in 2011. However, as is true for various ENERGY STAR qualified products, there is a range of energy performance among qualifying units for the new California standards. - Based on data for California qualifying flat panel televisions (CEC 2011), on average, the best practice

Table E-8: Low carbon technical potential scenario: Use phase assumptions		
Industry/sector	**Product**	**Derivation**
		technology for a given size class uses 30% less power in active mode than the average technology in that size class. - Based on the above, the technical potential for energy savings from adopting best practice technology (versus average qualified technology) for flat panel televisions is estimated at 30%.
	Hard disk drive	- In the absence of best practice power consumption data external hard drives, default enabled power management features are considered as best practice for energy efficiency in this study. - Using the savings of power management on PCs as a proxy (Masanet et al. 2005, Masanet and Horvath 2006b), the technical potential for energy savings from adopting best practice technology for external hard drives is estimated at 20%.
	Personal computer	- Based on data in Masanet et al. (2005, 2006a, 2006b, 2009) and U.S. EPA (2011e) it is estimated that switching to ENERGY STAR qualified computers would lead to roughly 20% energy savings for the average California desktop PC. - Using the September 2011 list of qualified ENERGY STAR desktop computers (U.S. EPA 2011d), and the usage pattern model in Masanet et al. (2005), the best performing ENERGY STAR qualified desktop PCs are estimated to consume, on average, around 15% less energy than the average ENERGY STAR qualified desktop. - Based on the above, the technical potential for energy savings from adopting best practice technology (versus average qualified technology) for desktop PCs is estimated at 35%.

Appendix F: Product case summaries

This appendix contains detailed results for the 2011 baseline and low-carbon technical potential estimates for each product case. To be concise, and for ease of access to specific results, each of the 22 product case summaries is organized in the same way. The organization scheme is described below.

The product definition is listed first, followed by a brief description of the product life cycle. Where available, the reader is referred to information in the public domain that describes the product life cycle in more detail.

Next, seven different figures are presented in the following order:

1. **Top 10 contributing sectors to the energy footprint of product manufacture (MJ/$).** This figure summarizes the outputs of the MRIO model for each product, in terms of the energy required by key value chain sectors to produce a dollar of output from the final manufacturing sector. Results are summarized by IO sector and fuel to indicate the major contributors to the "energy footprint" of product manufacture.

2. **Top 10 contributing sectors to the GHG emissions footprint of product manufacture (MJ/$).** This figure summarizes the outputs of the MRIO model for each product, in terms of the GHG emissions associated with key value chain sectors to produce a dollar of output from the final manufacturing sector. Results are summarized by IO sector and GHG emission type to indicate the major contributors to the "carbon footprint" of product manufacture.

3. **Top 15 estimated supply chain efficiency and GHG mitigation reductions by improvement opportunity and emissions source.** This figure summarizes the results of applying the eSTEP end use efficiency and GHG emissions abatement reduction factors across the entire supply chain of the product. The results show the top 15 emissions reduction opportunities across the supply chain in terms of estimated emissions saved by fuel and end use emissions reduction opportunities (for energy efficiency) and by GHG emissions type and abatement opportunity (for GHG emissions abatement measures).

4. **Top 10 estimated supply chain efficiency and GHG mitigation reductions by sector and emissions source**. This figure summarizes the top 10 supply chain IO sectors in which the eSTEP emissions reductions identified above would occur if fully implemented.

5. **Estimated 2011 GHG emissions by life-cycle phase and region in the baseline case.** This figure summarizes the results of the 2011 baseline case by life-cycle phase (production, transportation, use, end of life) and region of emission (California, Rest of US, Rest of World). Detailed baseline case assumptions are summarized for each product in Appendix E.

6. **2011 GHG emissions by life-cycle phase and region in the low carbon technical potential case.** This figure summarizes the results of the 2011 low carbon technical potential case by life-cycle phase (production, transportation, use, end of life) and region of emission (California, Rest of US, Rest of World). Detailed low carbon technical potential case assumptions are summarized for each product in Appendix E.

7. **Side by side comparison of baseline and low carbon technical potential scenarios.** This figure displays the baseline and low carbon technical potential results in side-by-side fashion to summarize the estimated achievable 2011 emissions reductions by life-cycle phase. The total estimated reduction across the product life cycle is listed in red text below the figure.

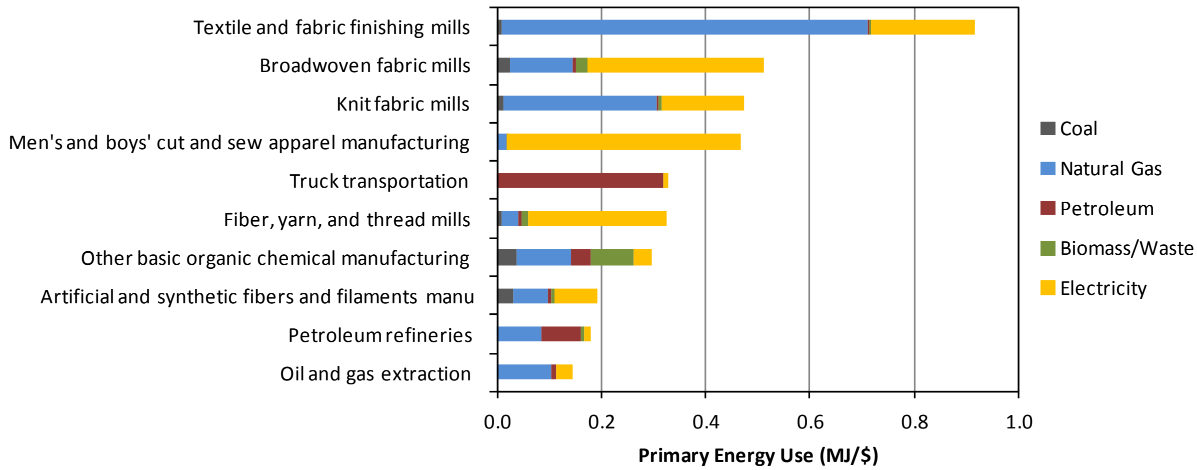
1: Apparel item (men's dress shirt)

Product

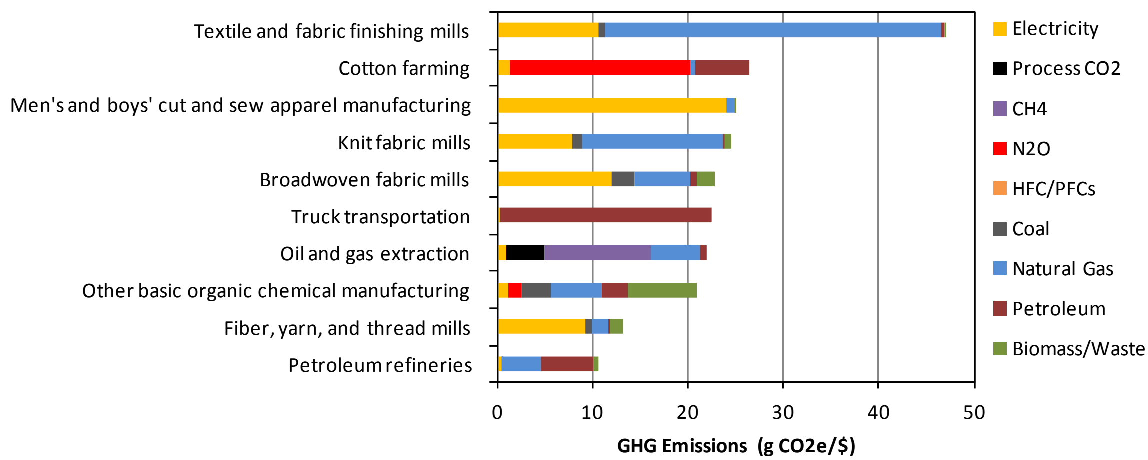
One men's dress shirt

Life-cycle system description

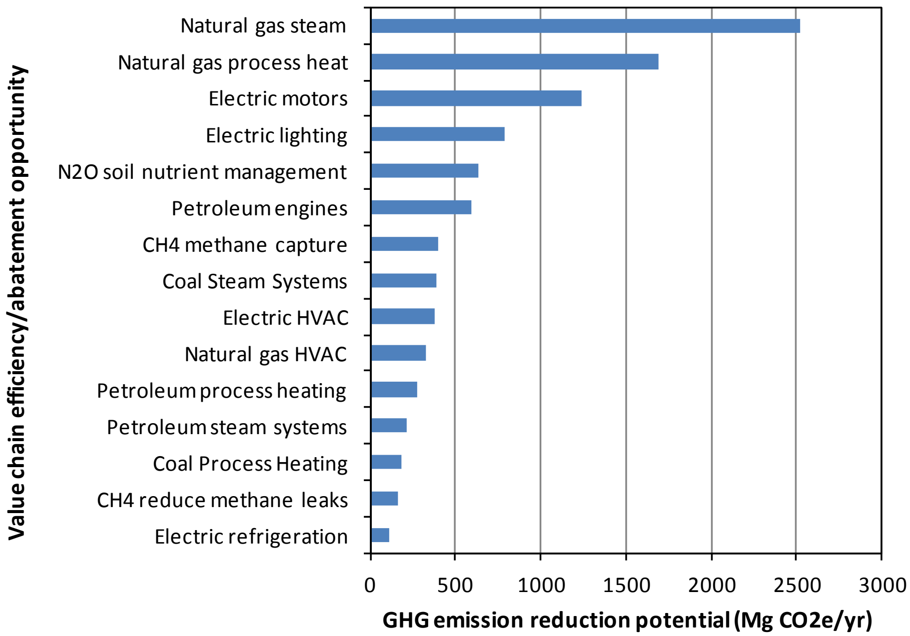
A men's dress shirt is typically made of fabrics that are comprised of cotton fibers or cotton fiber blends; most blends are also made of rayon or polyester fibers. The fabrics are made in a textile mill, which dyes, spins, and weaves the fibers into fabric, and then further dyes/prints/coats/finishes the fabric into a final textile product. Textile mills are typically large consumers of natural gas for generation of steam and process heat, and electricity for motor-driven equipment (Hasanbeigi 2010). Finished textiles are shipped to the cut and sew factory for manufacture into the final shirt. At the shirt factory, fabric is cut into various shapes and sewn into the final product, where it is packaged in a plastic film bag and shipped to the final retail outlet. During its life, a shirt is typically washed (and dried) dozens of times before reaching the end of its useful life; therefore, it is an indirect consumer of energy during its use phase. Ultimately most shirts are expected to be sent to landfill at the end of life phase.



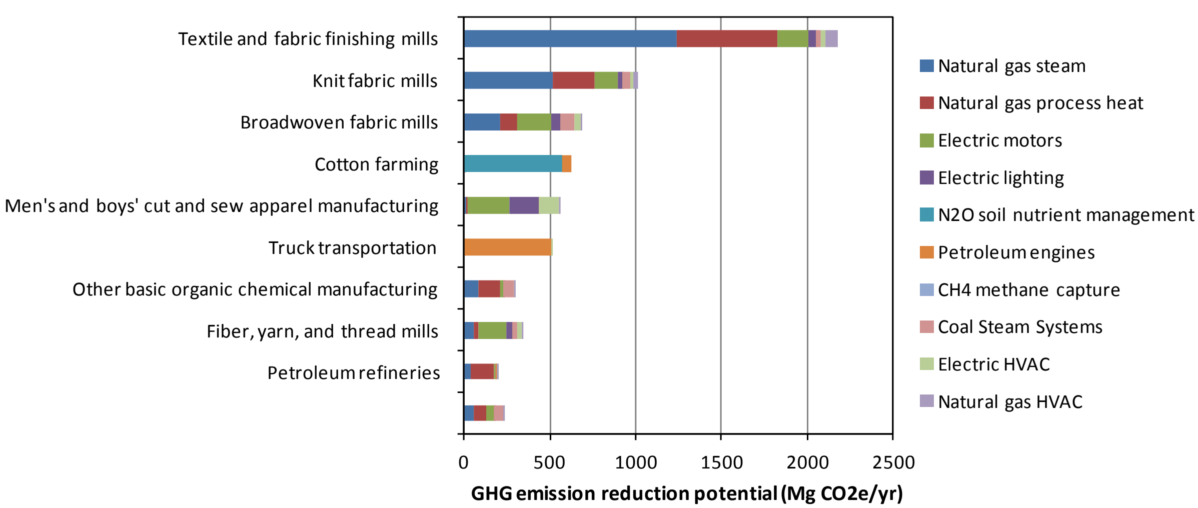
Top 10 contributing sectors to the energy footprint of product manufacture (MJ/$)



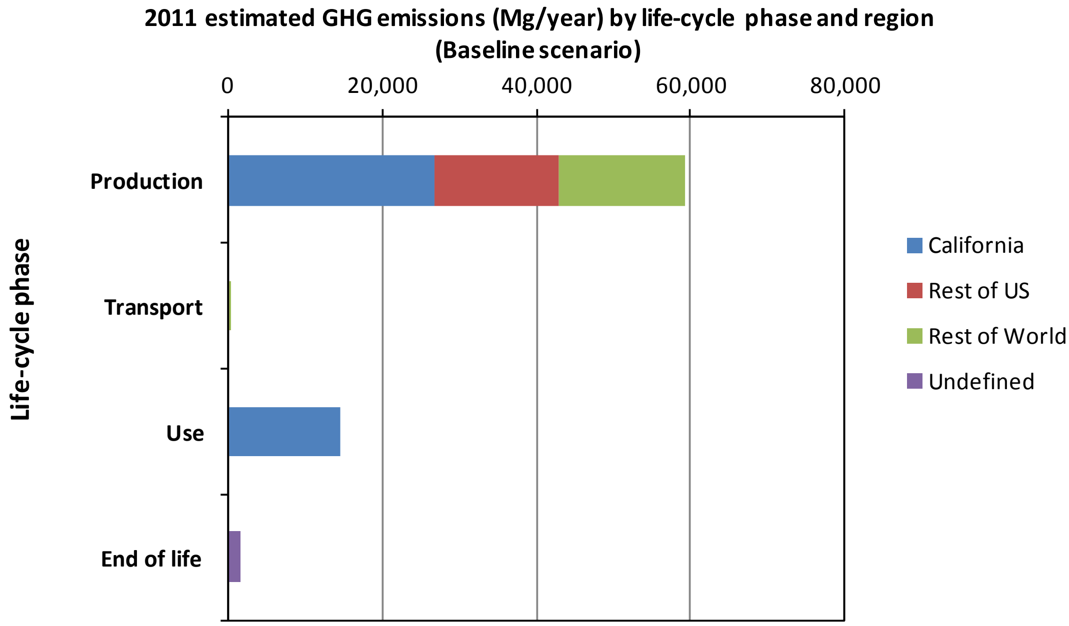
Top 10 contributing sectors to the GHG emissions footprint of product manufacture (MJ/$)



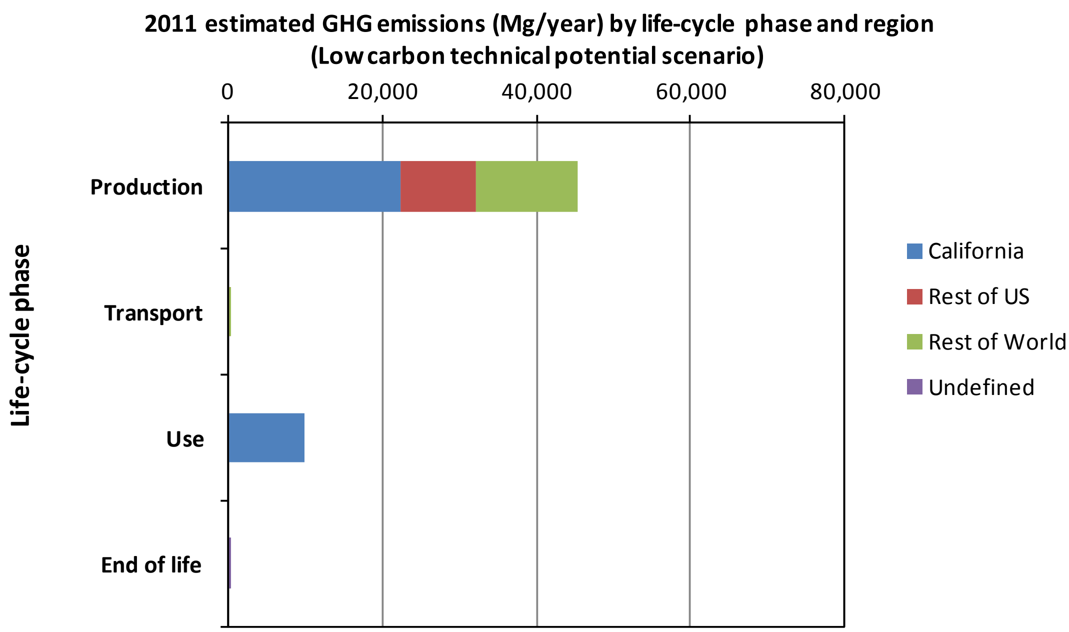
Top 15 estimated supply chain efficiency and GHG mitigation reductions by improvement opportunity and emissions source



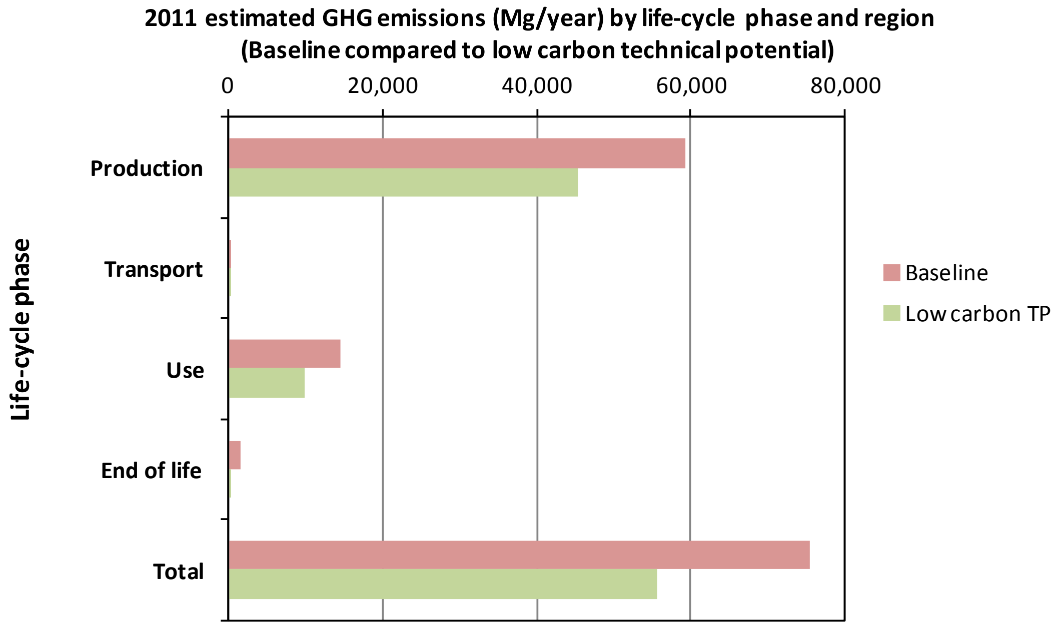
Top 10 estimated supply chain efficiency and GHG mitigation reductions by sector and emissions source



Estimated 2011 GHG emissions by life-cycle phase and region in the baseline scenario



2011 GHG emissions by life-cycle phase and region in the low carbon technical potential scenario



Side by side comparison of baseline and low carbon technical potential scenarios

Total savings = 26%

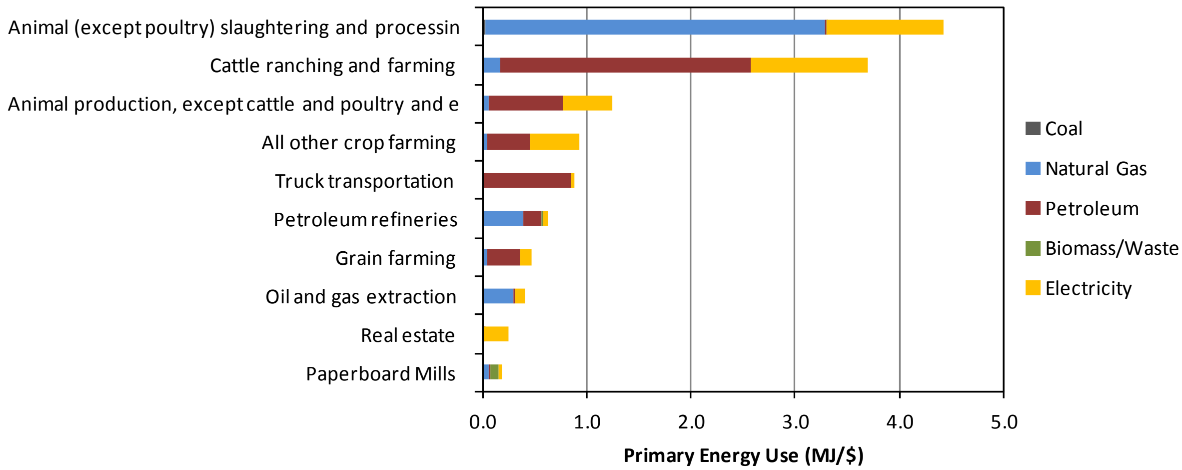
2: Beef

Product

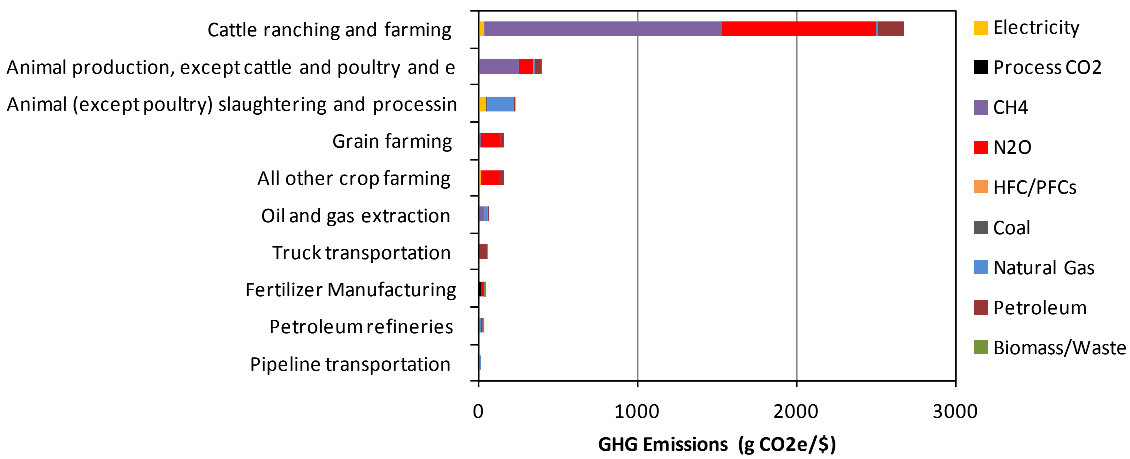
One kilogram of packaged beef

Life-cycle system description

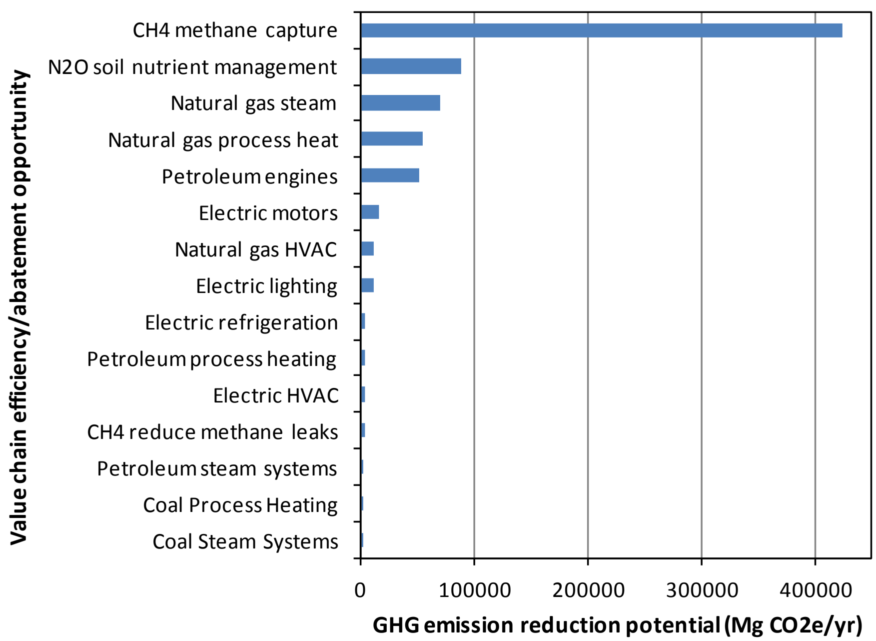
Cattle are first raised on a cattle ranch and fed a mostly grain and/or grass based diet. Ranches are significant generators of manure, which can be used as fertilizer or for methane recovery. In the processing plant, cattle are slaughtered and processed into different end use products such as cuts and ground beef. Beef is then packaged in plastic wrapping and trays and refrigerated prior to shipment to the retailer. After consumption, the plastic packaging is sent to landfill at end of life. Depending on the consumer, the product may be fully eaten or partially disposed as waste, which can either be landfilled or composted.



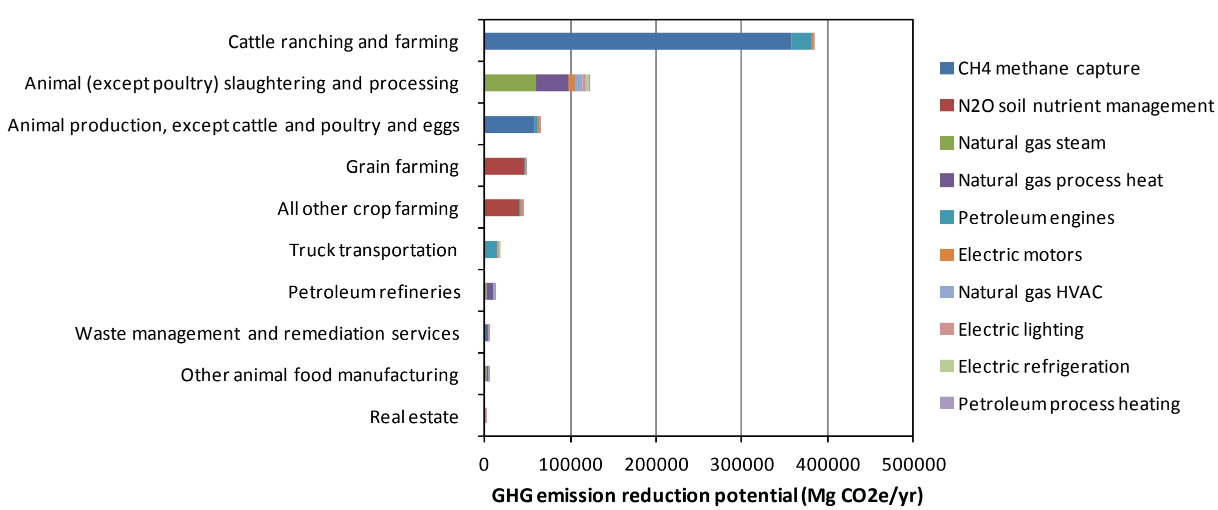
Top 10 contributing sectors to the energy footprint of product manufacture (MJ/$)



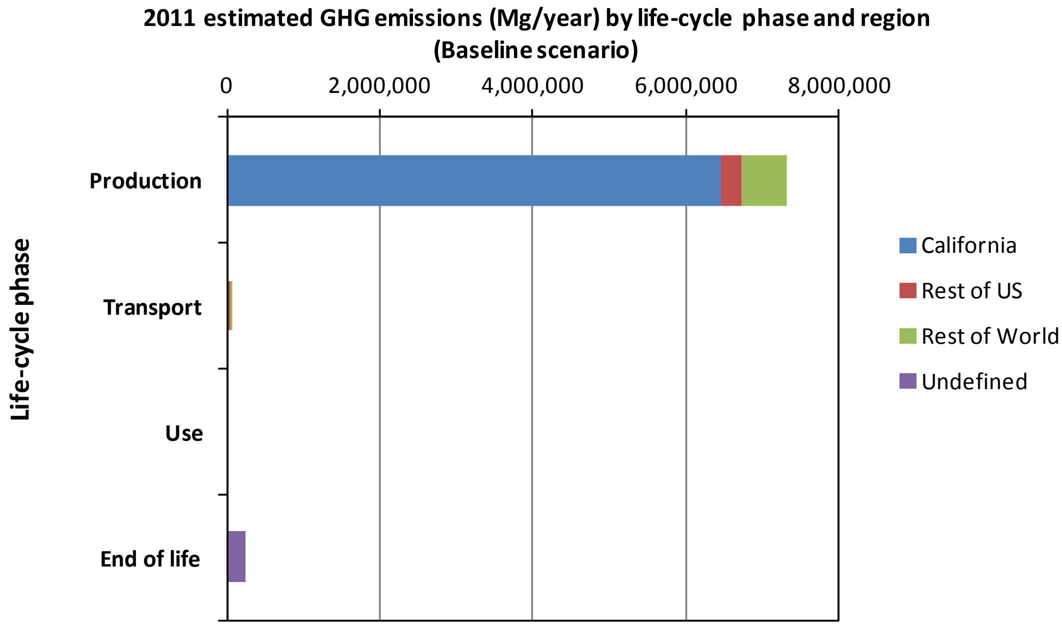
Top 10 contributing sectors to the GHG emissions footprint of product manufacture (MJ/$)



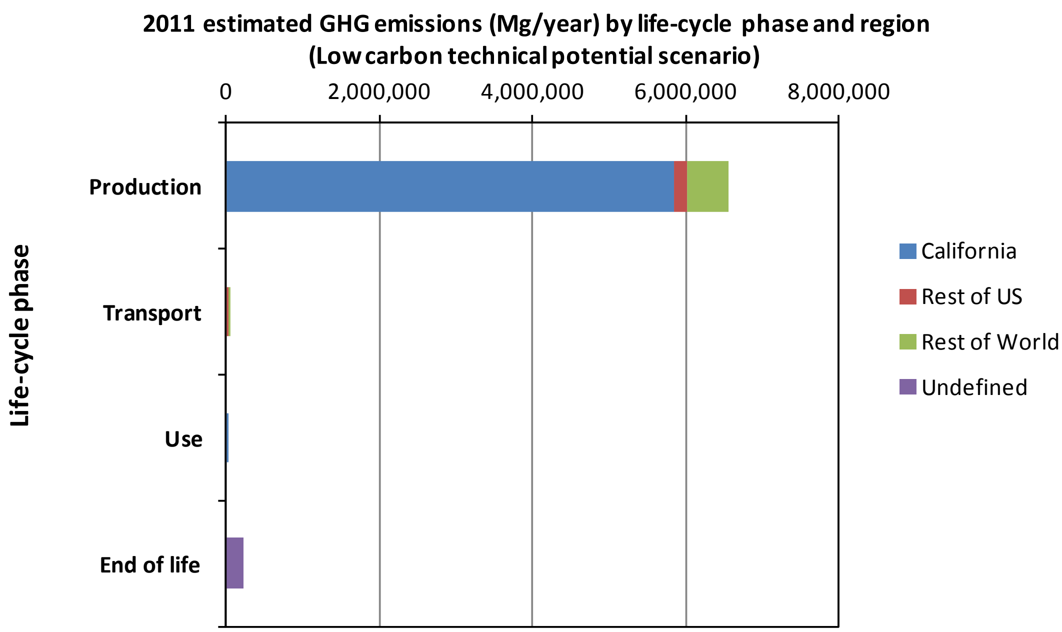
Estimated supply chain efficiency and GHG mitigation reductions by improvement opportunity and emissions source



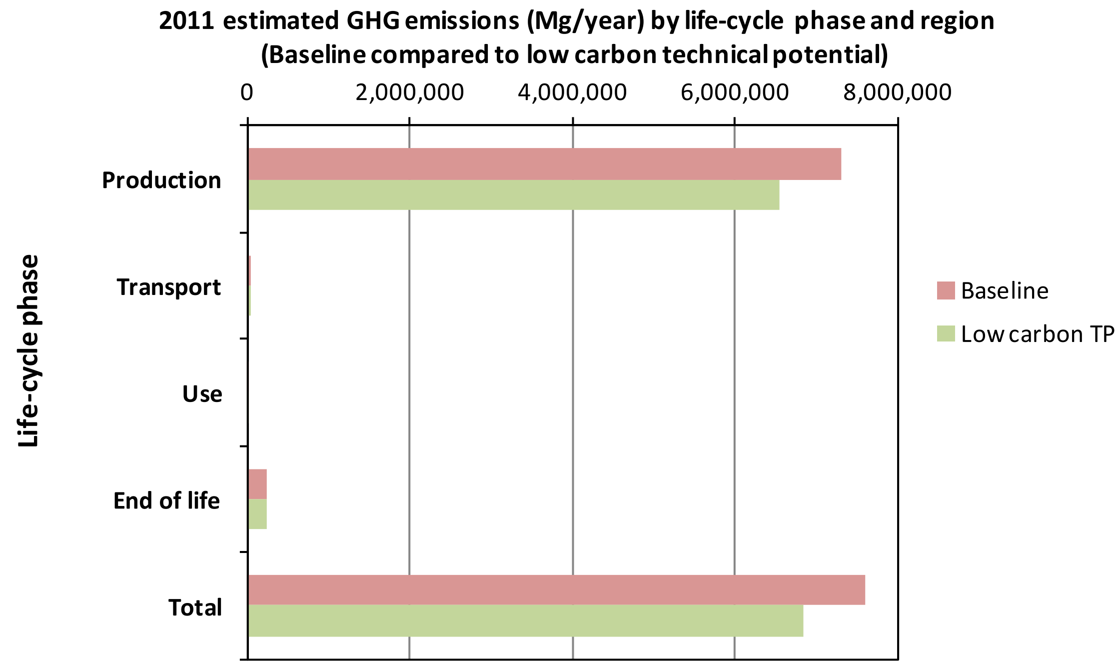
Estimated supply chain efficiency and GHG mitigation reductions by sector and emissions source



Estimated 2011 GHG emissions by life-cycle phase and region in the baseline scenario



2011 GHG emissions by life-cycle phase and region in the low carbon technical potential scenario



Side by side comparison of baseline and low carbon technical potential scenarios

Total savings = 10%

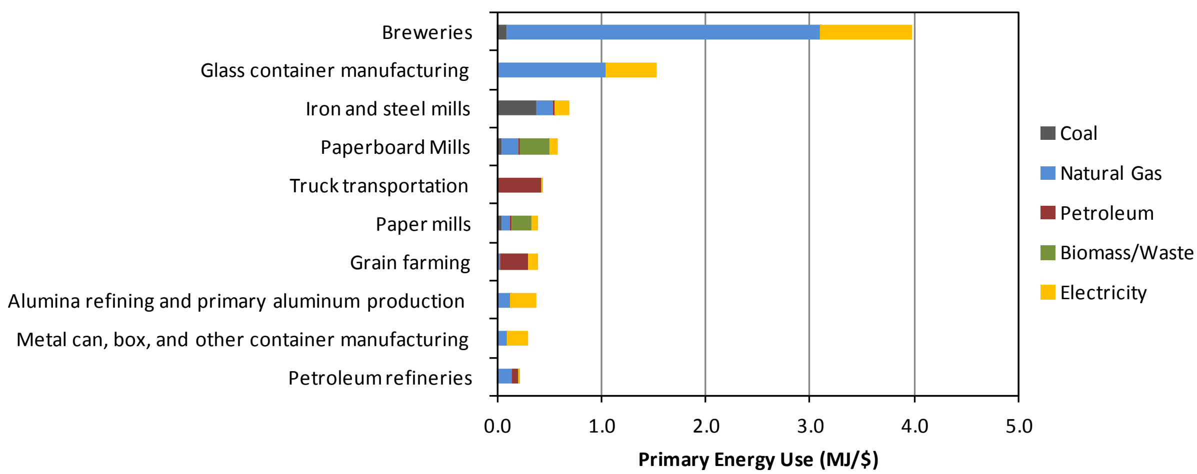
3: Beer

Product

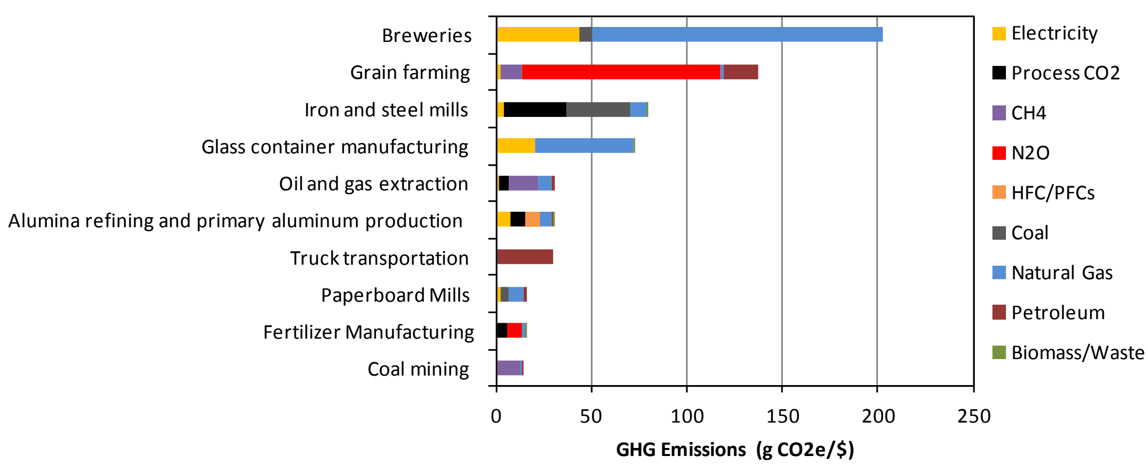
One 12 ounce bottle of beer

Life-cycle system description

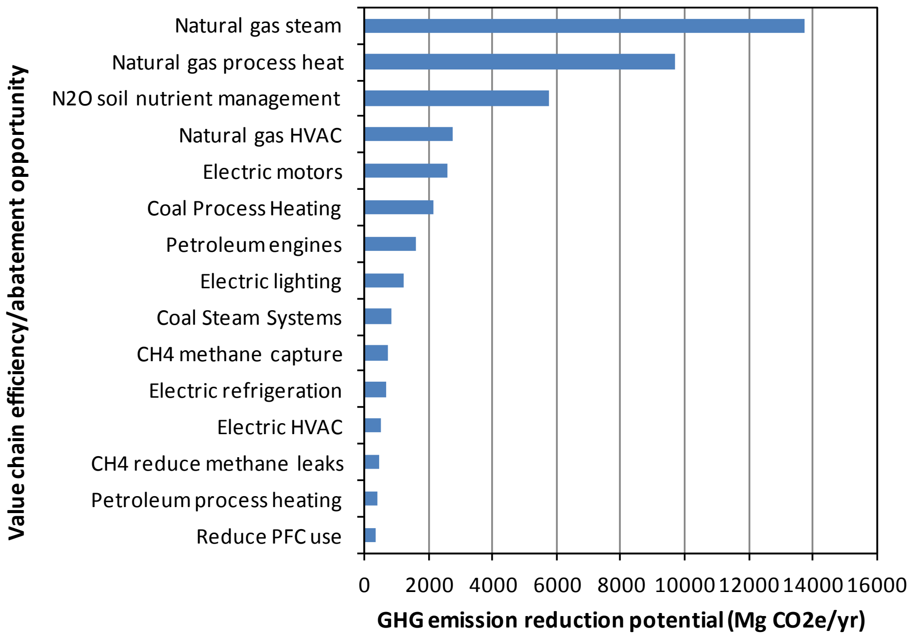
The brewing process uses malted barley and/or cereals, unmalted grains and/or sugar/corn syrups (adjuncts), hops, water, and yeast to produce beer. Brewing is a multi-step process, and can be divided into brewhouse operations to produce hopped wort, fermentation and carbonation, filtration and bottle filling, and finally pasteurization. The process consumes mostly natural gas and electricity, with brewing and pasteurization accounting for the greatest share of thermal energy and motors and refrigeration accounting for the greatest share of electricity (Galitsky et al. 2003). The production of glass bottles is also a major consumer of energy. The final product is typically packaged in cardboard containers and shipped by truck to the retail outlet. Beer is then refrigerated and consumed; around 85% of glass bottles are currently recycled thanks to the California Redemption Value (CRV) fee (CalRecycle 2011).



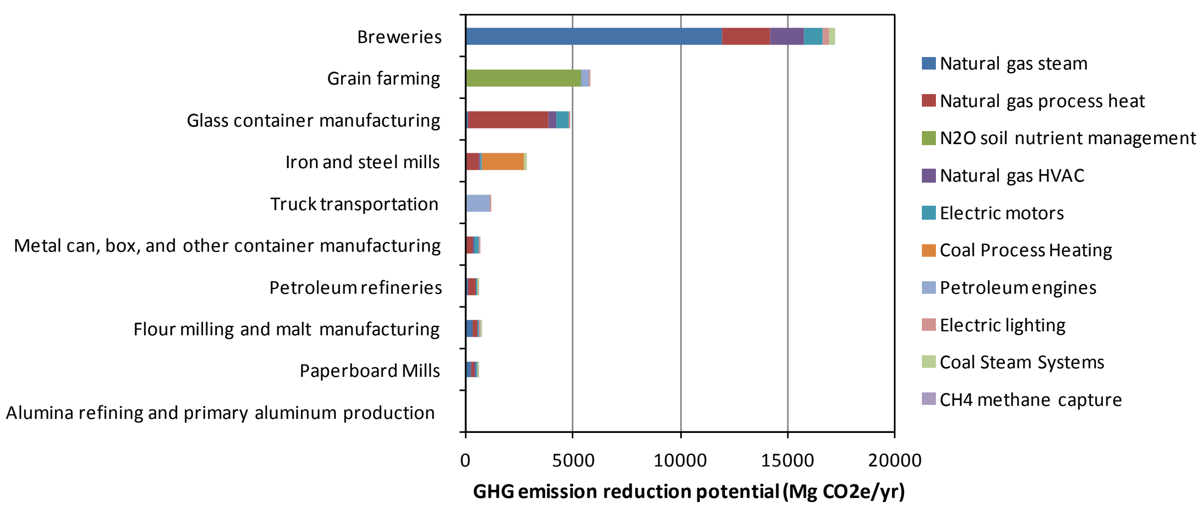
Top 10 contributing sectors to the energy footprint of product manufacture (MJ/$)



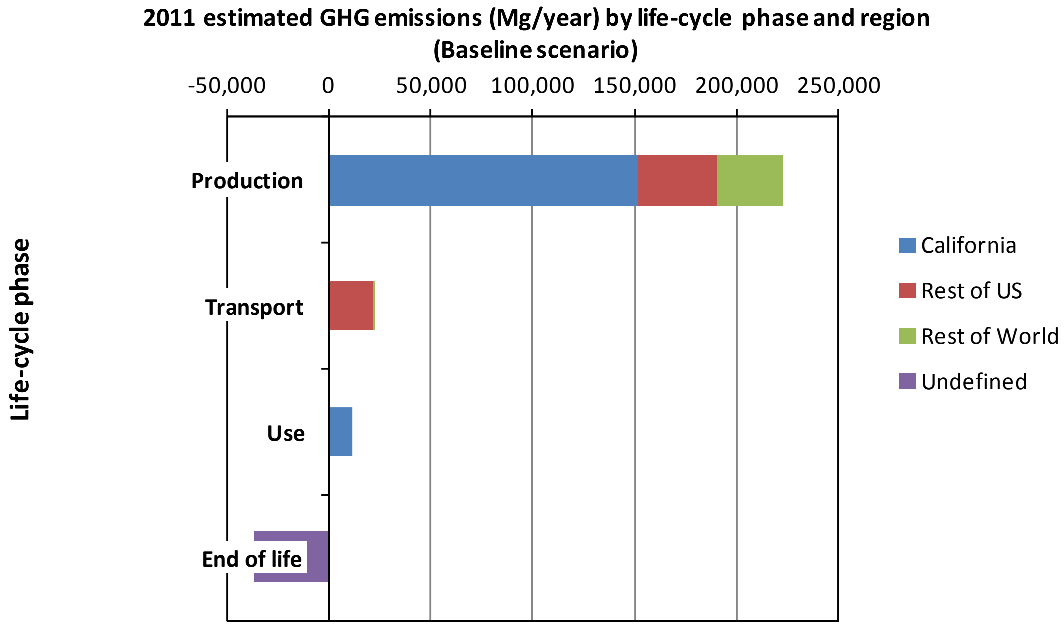
Top 10 contributing sectors to the GHG emissions footprint of product manufacture (MJ/$)



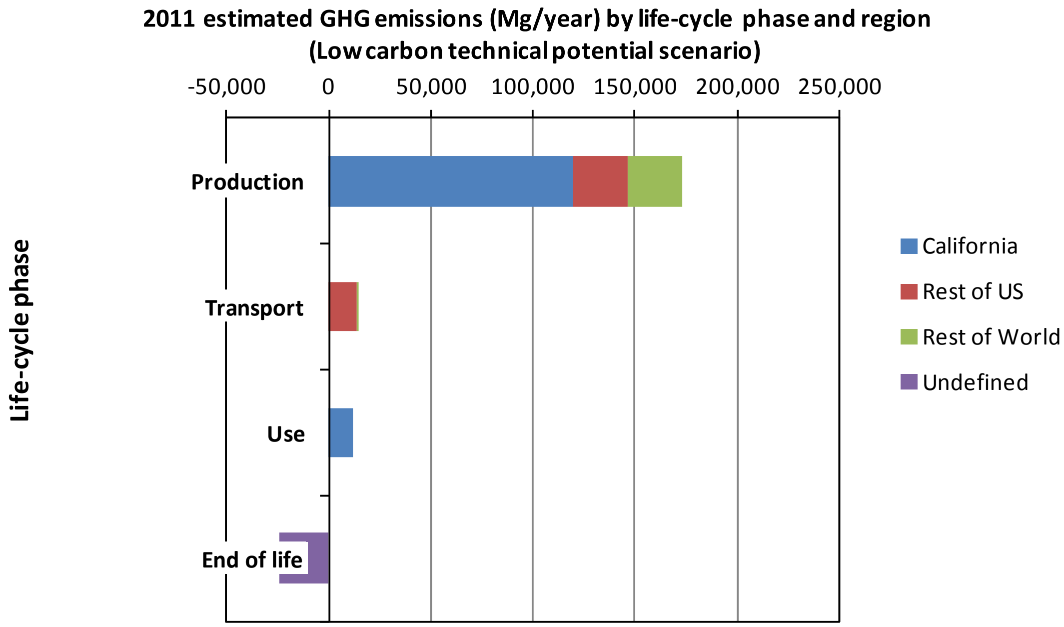
Estimated supply chain efficiency and GHG mitigation reductions by improvement opportunity and emissions source



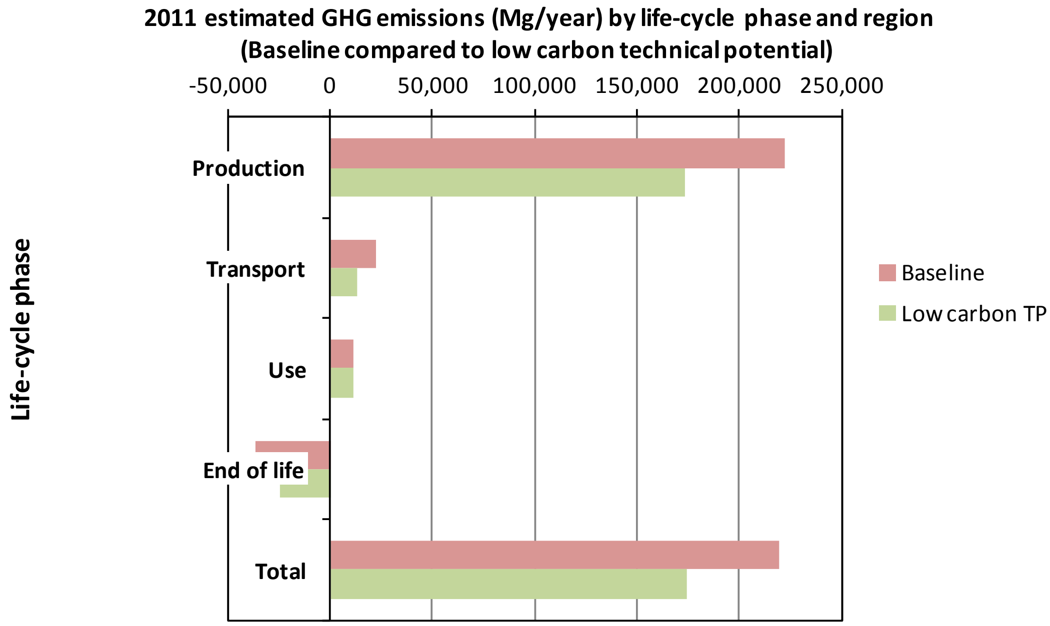
Estimated supply chain efficiency and GHG mitigation reductions by sector and emissions source



Estimated 2011 GHG emissions by life-cycle phase and region in the baseline scenario



2011 GHG emissions by life-cycle phase and region in the low carbon technical potential scenario



Side by side comparison of baseline and low carbon technical potential scenarios

Total savings = 20%

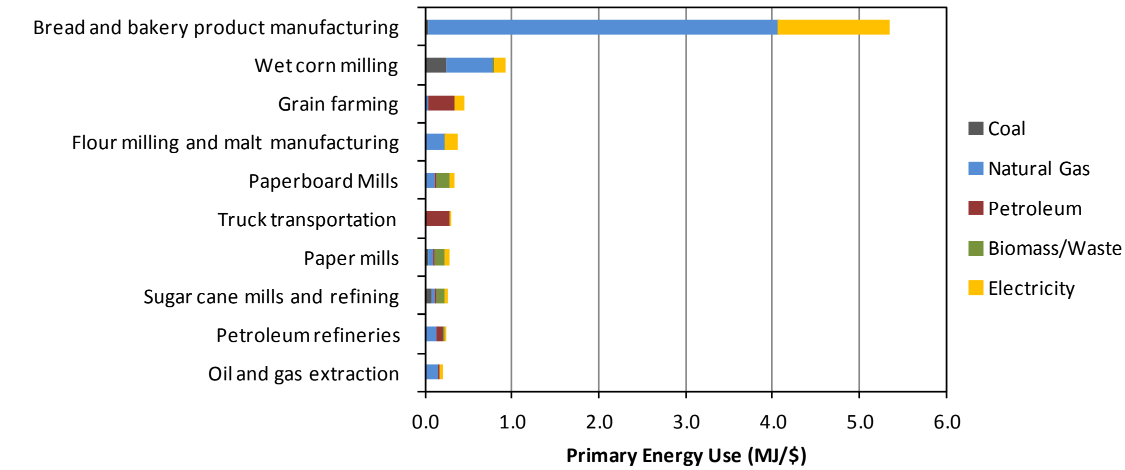
4: Bread

Product

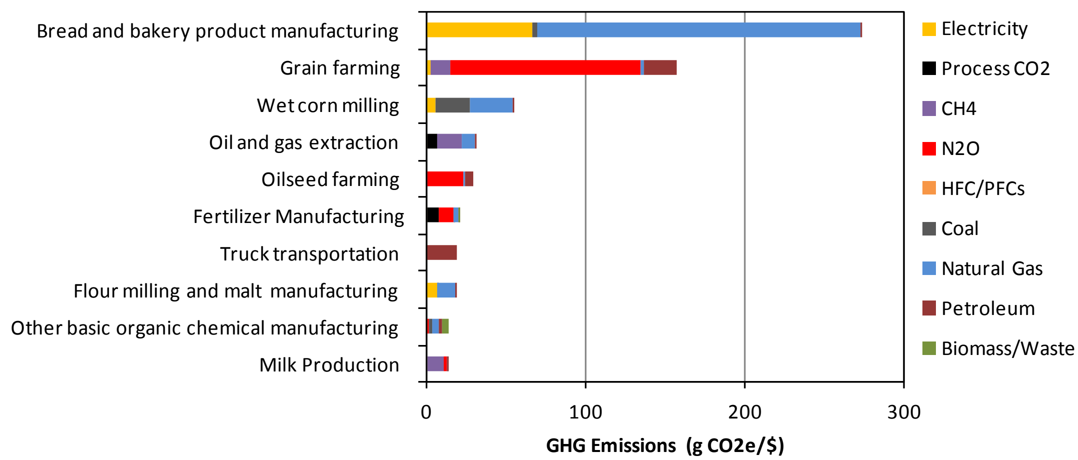
One kilogram of white bread

Life-cycle system description

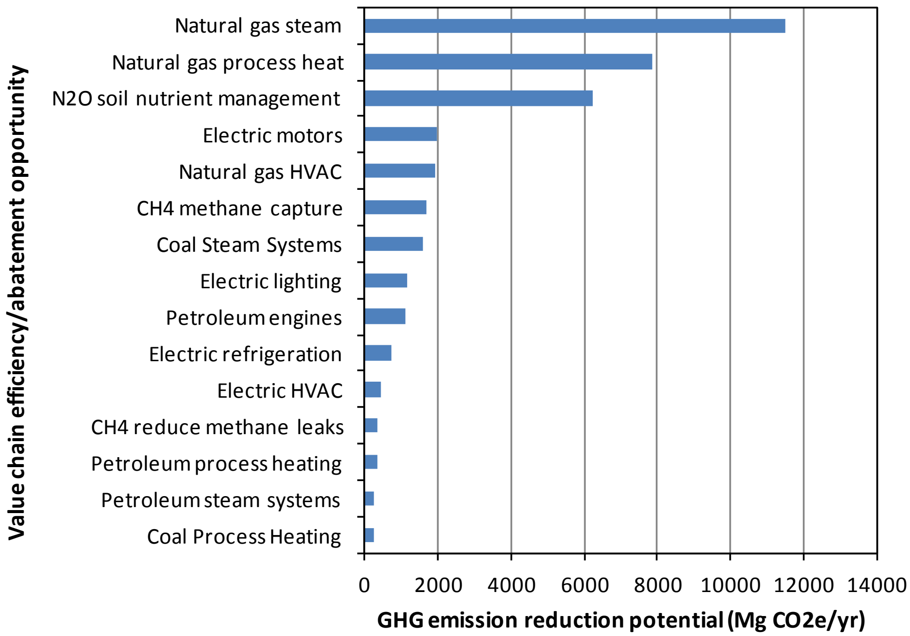
The primary ingredients in bread are wheat flour, water, yeast, and small amounts of fats (oil or butter), salt, sugar, and (sometimes) nutritional additives. Wheat flour and sugar are produced by crop harvesting and milling prior to shipment to the bread factory. At the factory, ingredients are mixed into dough and left to rise in the fermenting stage. The dough is then sent through an automated process that extrudes and cuts the dough into predefined sizes, flours and shapes it, and conveys it into baking trays. The trays are conveyed through a preheater for further rising, and then into a gas-fired baking oven. The finished bread is then packaged in plastic film bags, loaded onto trays, and shipped to the retailer. The plastic packaging is sent to landfill at end of life. Depending on the consumer, the bread may be fully eaten or partially disposed as waste, which can either be landfilled or composted.



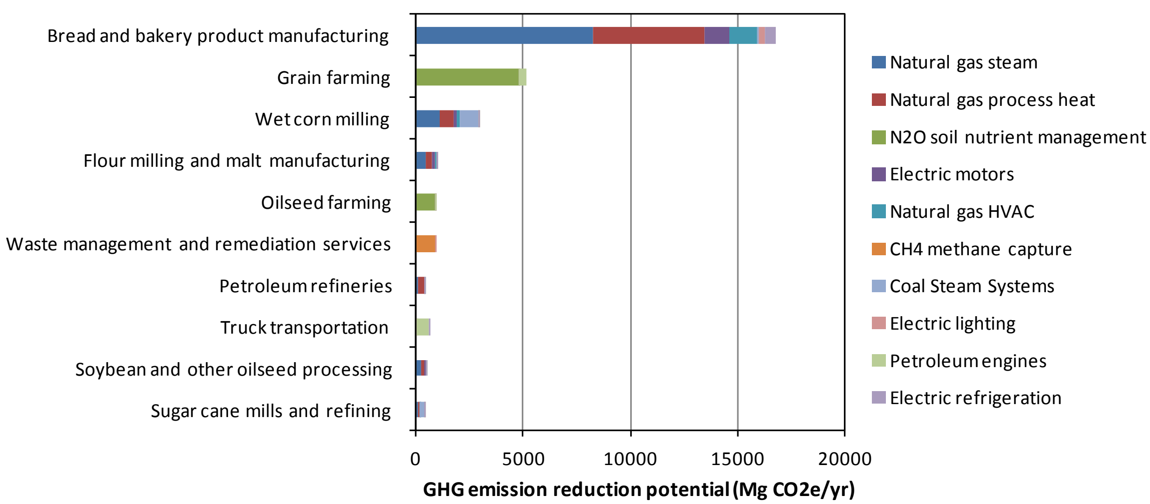
Top 10 contributing sectors to the energy footprint of product manufacture (MJ/$)



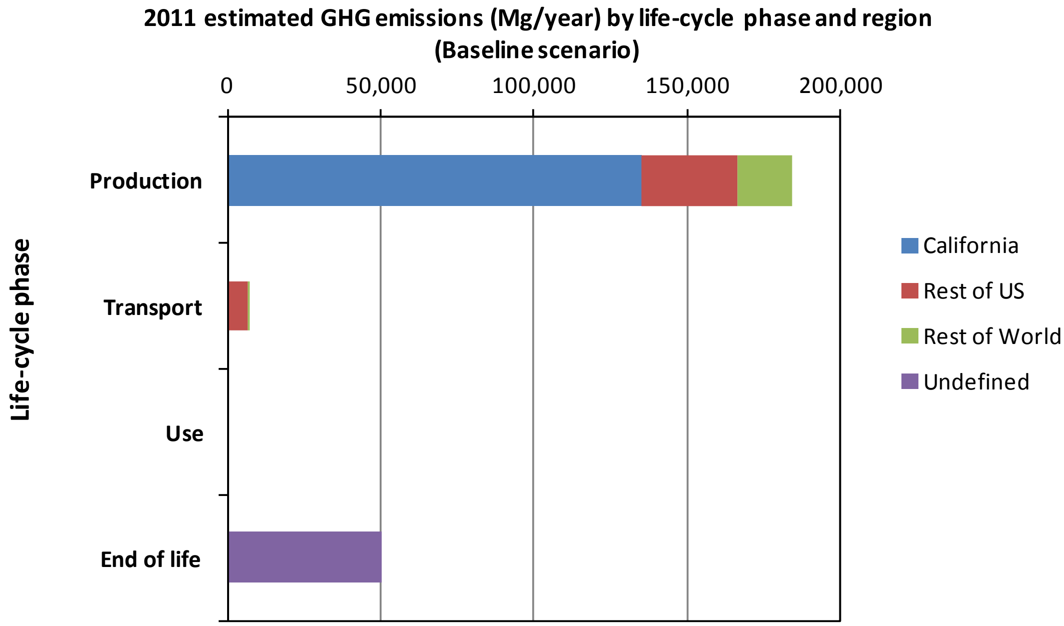
Top 10 contributing sectors to the GHG emissions footprint of product manufacture (MJ/$)



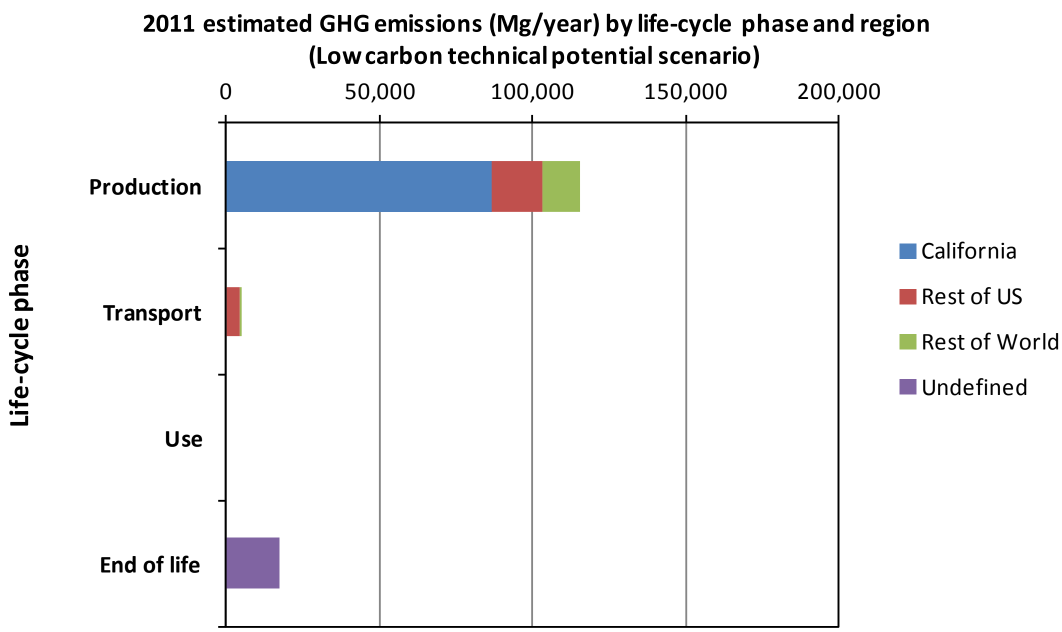
Estimated supply chain efficiency and GHG mitigation reductions by improvement opportunity and emissions source



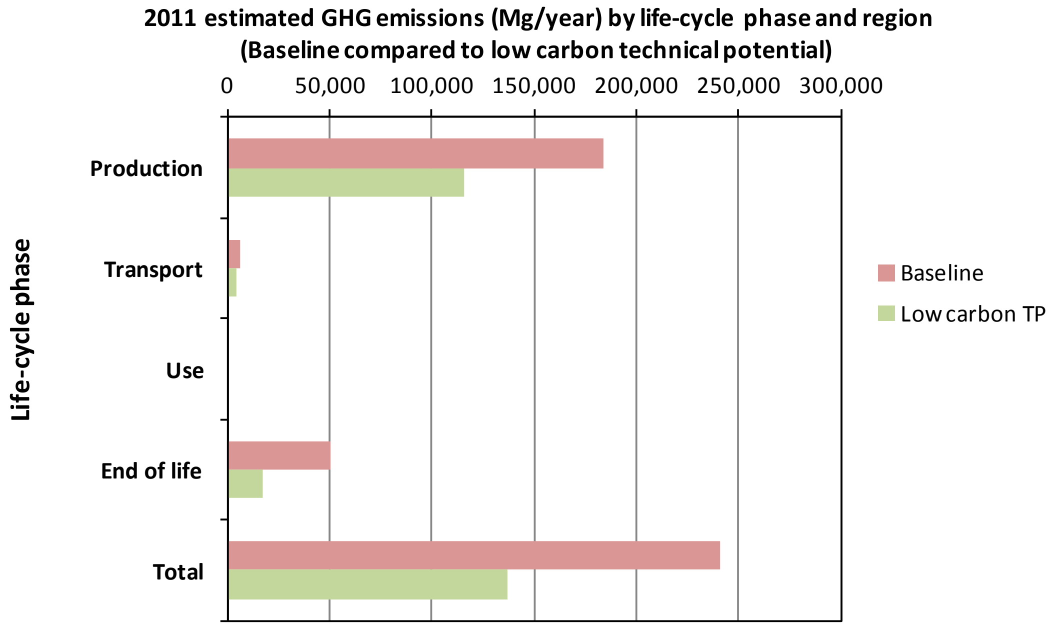
Estimated supply chain efficiency and GHG mitigation reductions by sector and emissions source



Estimated 2011 GHG emissions by life-cycle phase and region in the baseline scenario



2011 GHG emissions by life-cycle phase and region in the low carbon technical potential scenario



Side by side comparison of baseline and low carbon technical potential scenarios

Total savings = 43%

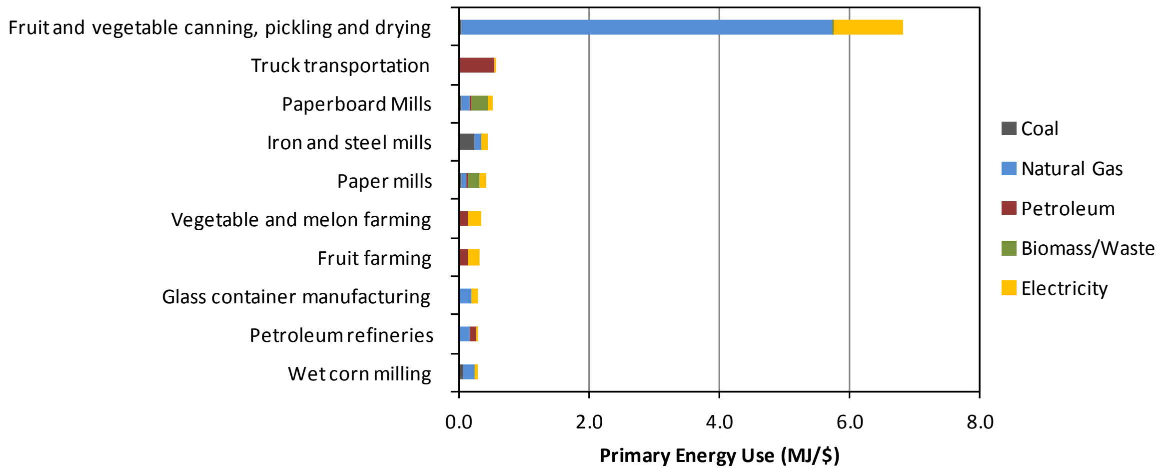
5: Canned tomatoes

Product

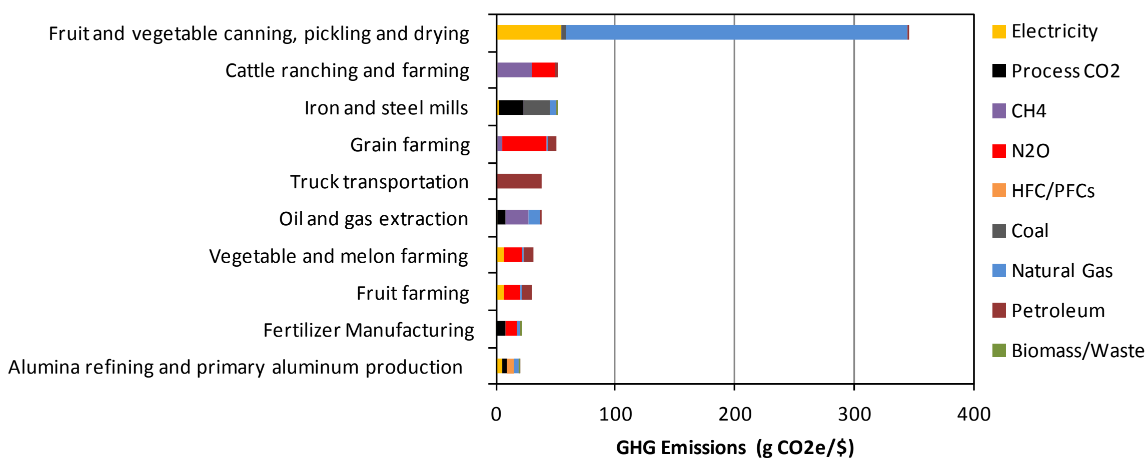
One 0.5 liter can of processed tomatoes

Life-cycle system description

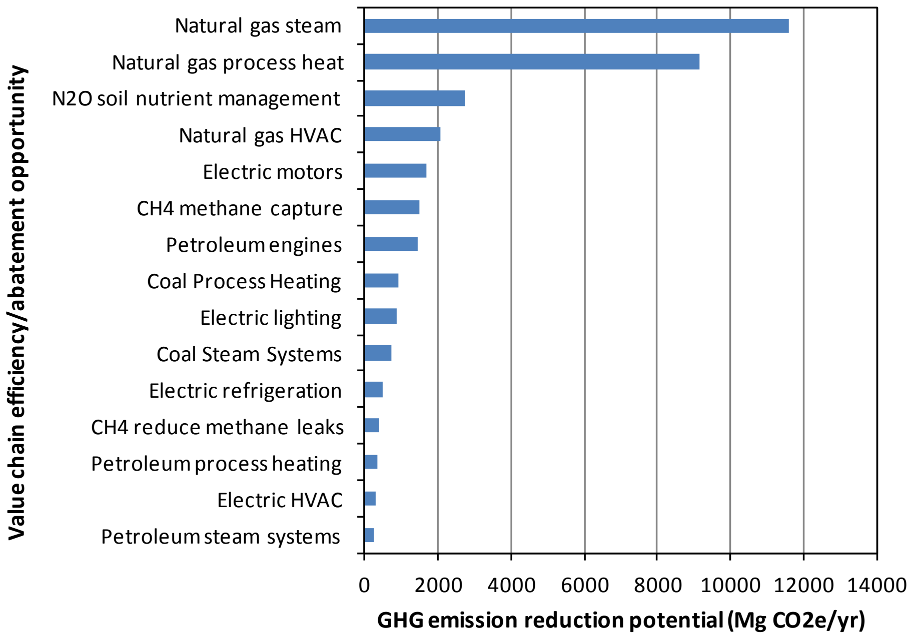
Tomatoes are harvested and shipped by truck to a processing facility. After inspection and grading, tomatoes are typically washed in a series of agitated water flumes. Next, color sorting is done either manually or automatically to remove green tomatoes, which are subsequently sent to pulping. The red tomatoes are then subjected to steam peeling, followed by manual sorting to remove tomatoes that have not been sufficiently peeled, which are also sent to pulping. Peeled red tomatoes are then diced and filled into cans using rotary brush fillers. The canned diced tomatoes are then exhausted, sealed, sterilized, and cooled before proceeding to final packaging operations. The pulper is used to crush green and unpeeled tomatoes as well as pulping waste from the dicer. After pulping, the tomato slurry proceeds to the evaporator for concentration into juice, puree, and paste (the final product is solely dependent on the remaining moisture content after evaporation). Tomato purees are then typically mixed with other ingredients to create tomato sauce. Prior to filling, evaporated tomato products undergo continuous sterilization. Once filled the canned tomato juices, pastes, and sauces are sent to final packaging operations (Masanet et al. 2007). Major consumers of energy in tomato processing are the washing, blanching, and sterilization stages. The canned tomatoes are then shipped to the retailer. After consumption, the steel can is either recycled or sent to landfill by the consumer. Depending on the consumer, the product may be fully eaten or partially disposed as waste, which can either be landfilled or composted.



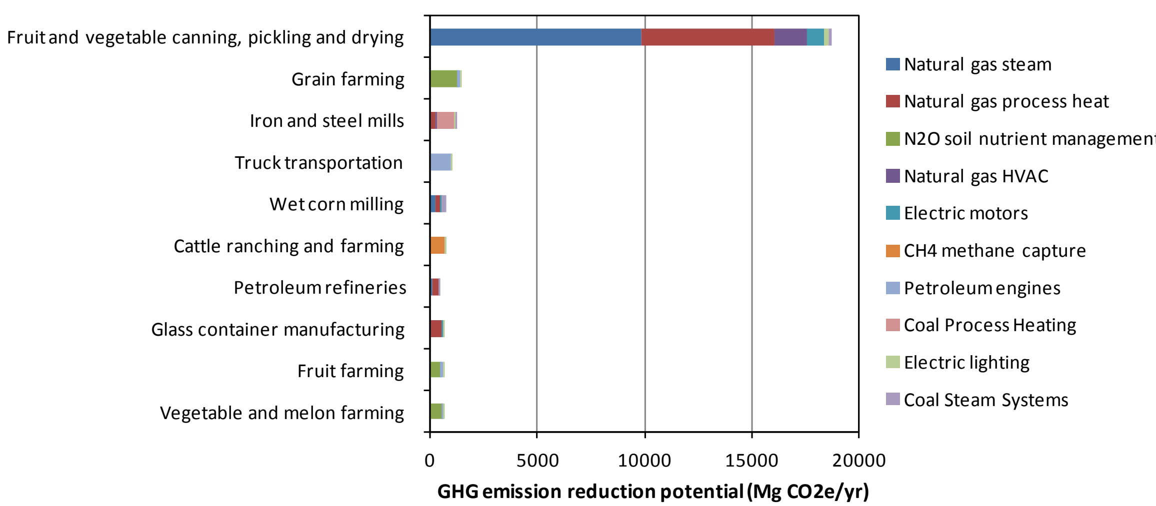
Top 10 contributing sectors to the energy footprint of product manufacture (MJ/$)



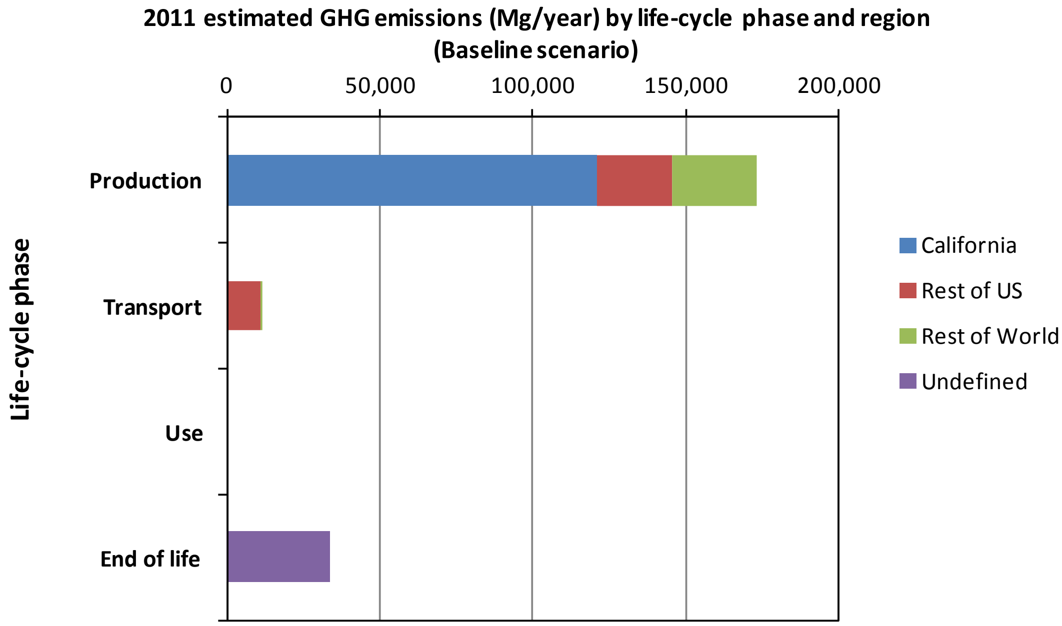
Top 10 contributing sectors to the GHG emissions footprint of product manufacture (MJ/$)



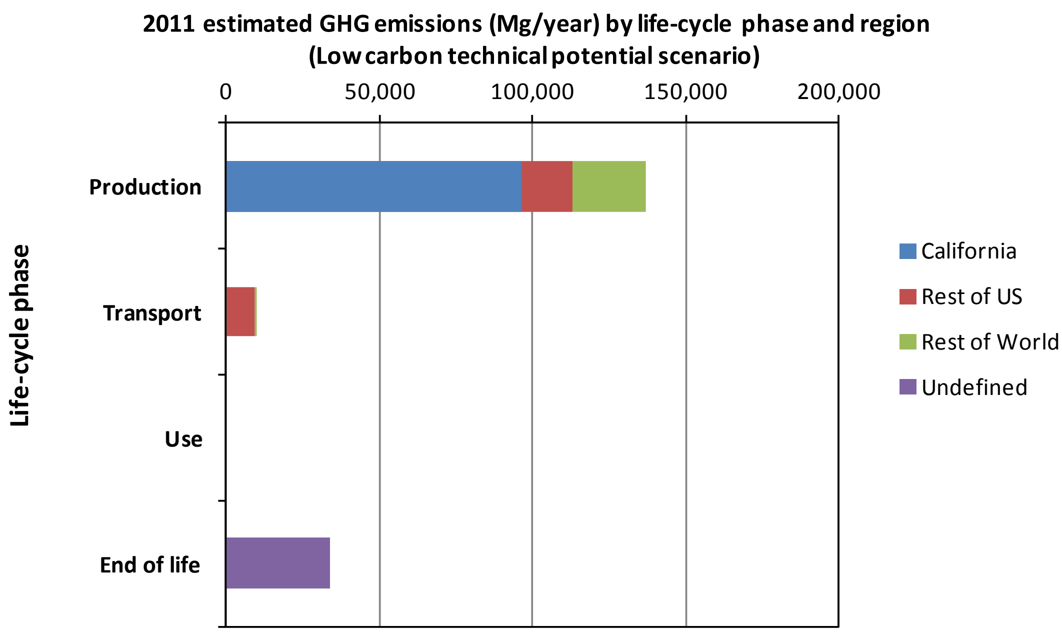
Estimated supply chain efficiency and GHG mitigation reductions by improvement opportunity and emissions source



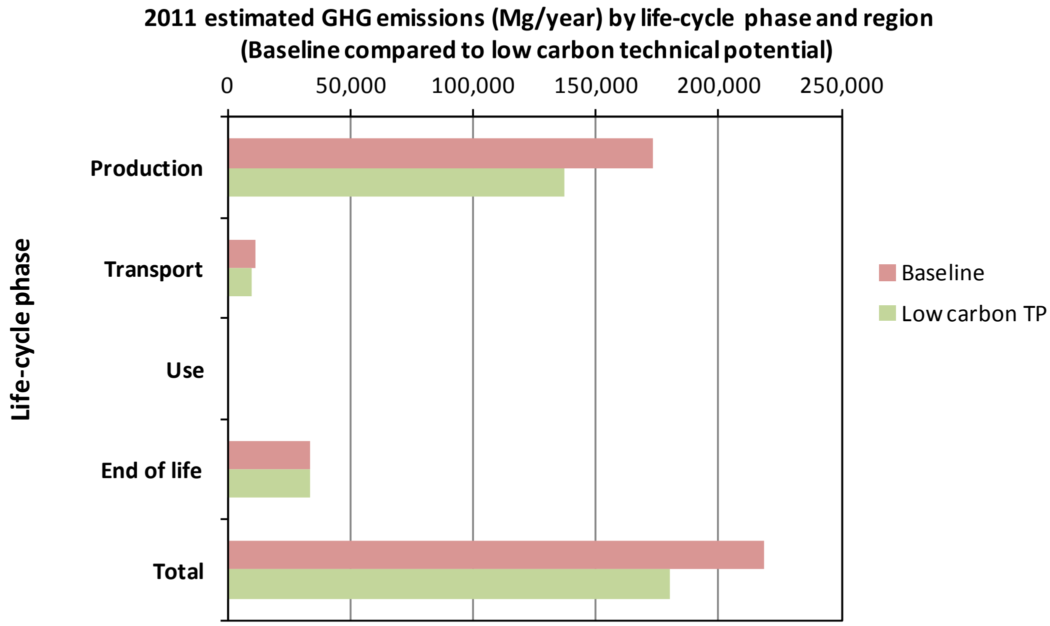
Estimated supply chain efficiency and GHG mitigation reductions by sector and emissions source



Estimated 2011 GHG emissions by life-cycle phase and region in the baseline scenario



2011 GHG emissions by life-cycle phase and region in the low carbon technical potential scenario



Side by side comparison of baseline and low carbon technical potential scenarios

Total savings = 17%

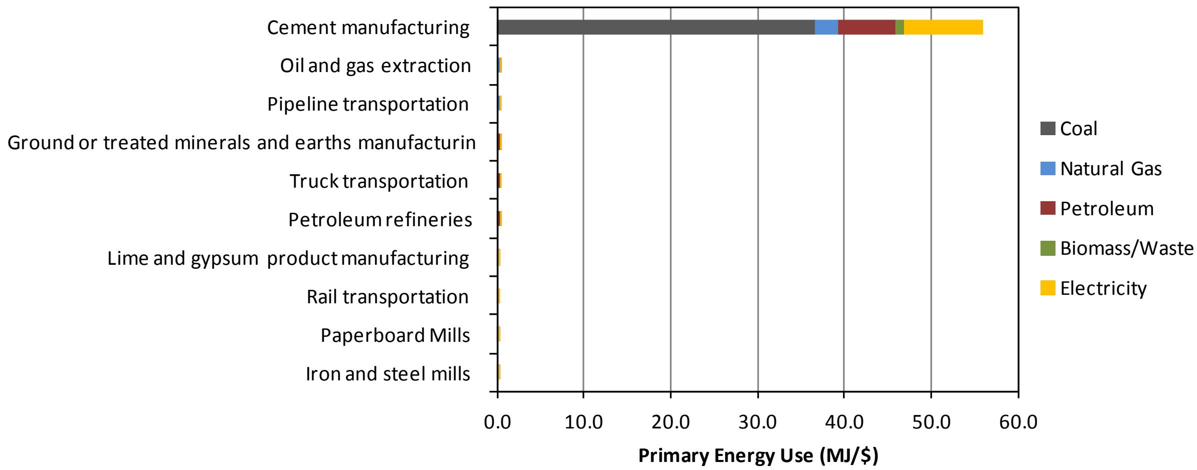
6: Cement (Masonry)

Product

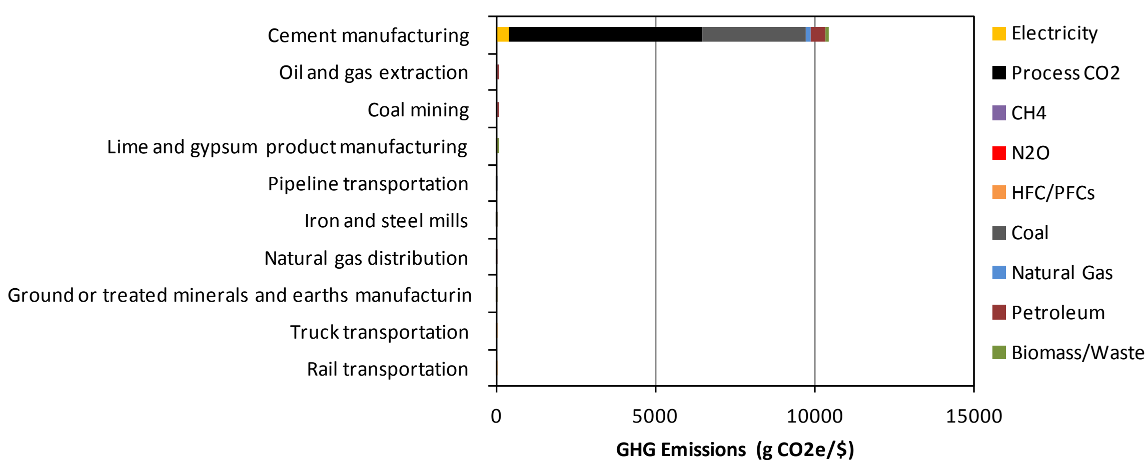
One metric ton of masonry cement

Life-cycle system description

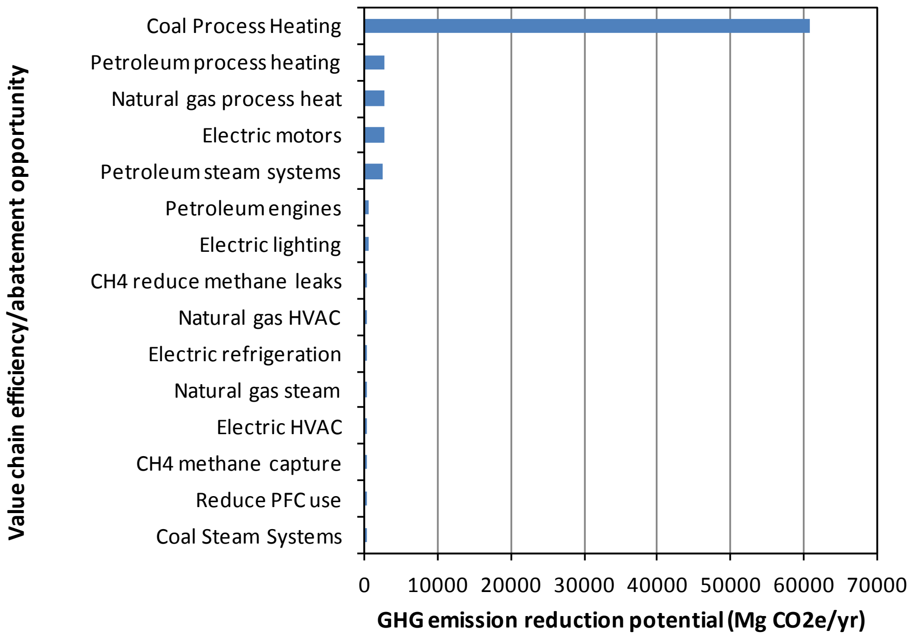
Cement is a fine powder, usually gray in color that consists of a mixture of hydraulic cement minerals to which one or more forms of calcium sulfate have been added. First, limestone and other minerals are mined. Next, crushing, grinding, and mixing of limestone and other quarried materials is done to produce a mixture of silicon, aluminum and iron oxides known as raw meal. Next, pyroprocessing of raw meal occurs to produce clinker, which is the primary component of cement. Pyroprocessing occurs in a large kiln that is heated through the direct combustion of fuels such as coal, natural gas, or waste tires. After cooling, the clinker is typically stored before being transported to the finish milling process, which grinds the clinker down into a fine powder together with additions (3-5% gypsum to control the setting properties of the cement) to form Portland cement. Masonry cement is typically packaged in heavy duty paper bags for shipment to the retailer. For use, water is added to the cement. Once put into place, the product is expected to stay installed for many years.



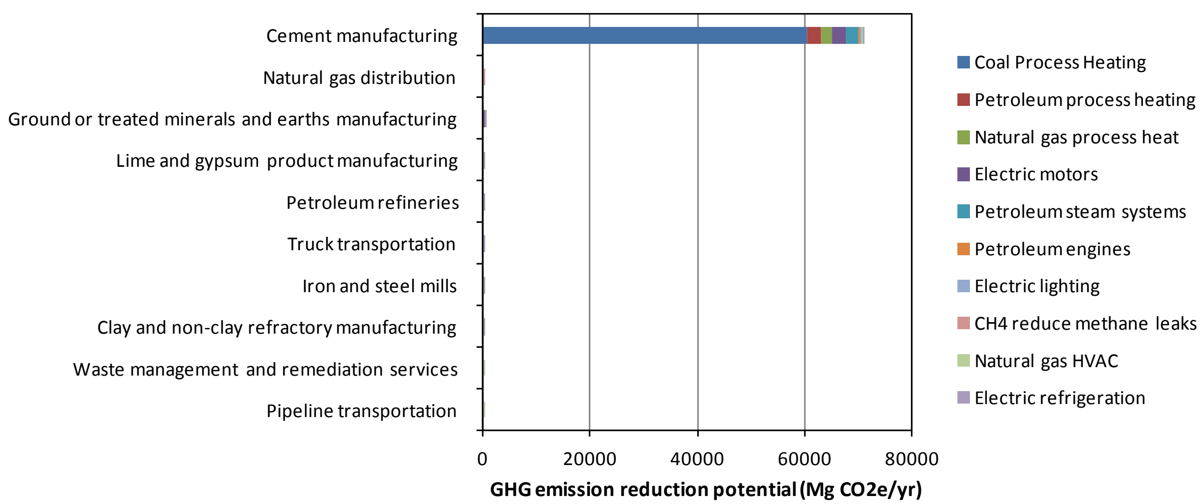
Top 10 contributing sectors to the energy footprint of product manufacture (MJ/$)



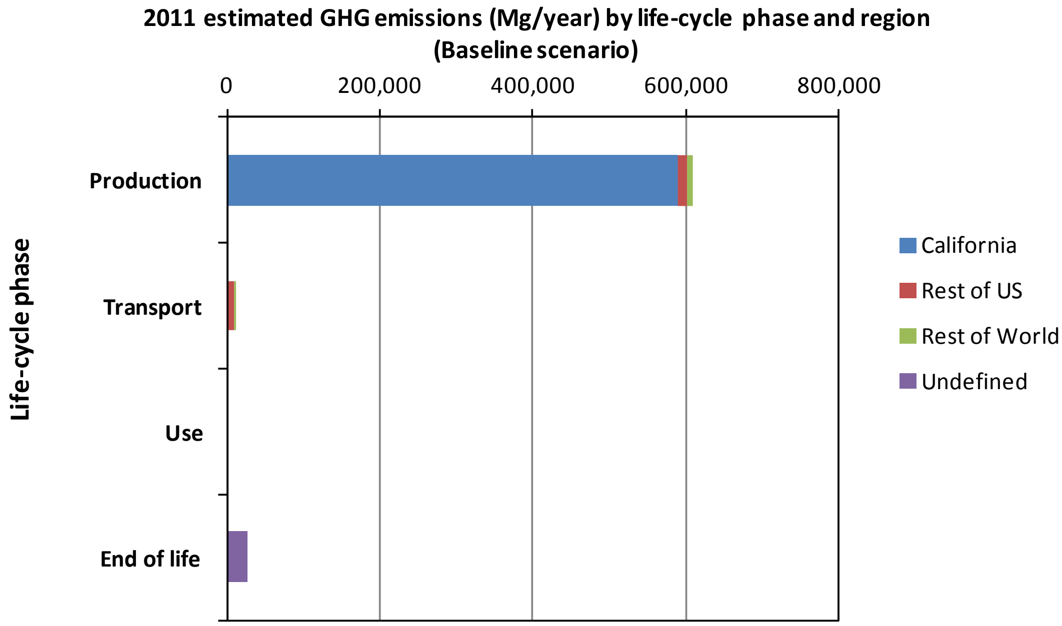
Top 10 contributing sectors to the GHG emissions footprint of product manufacture (MJ/$)



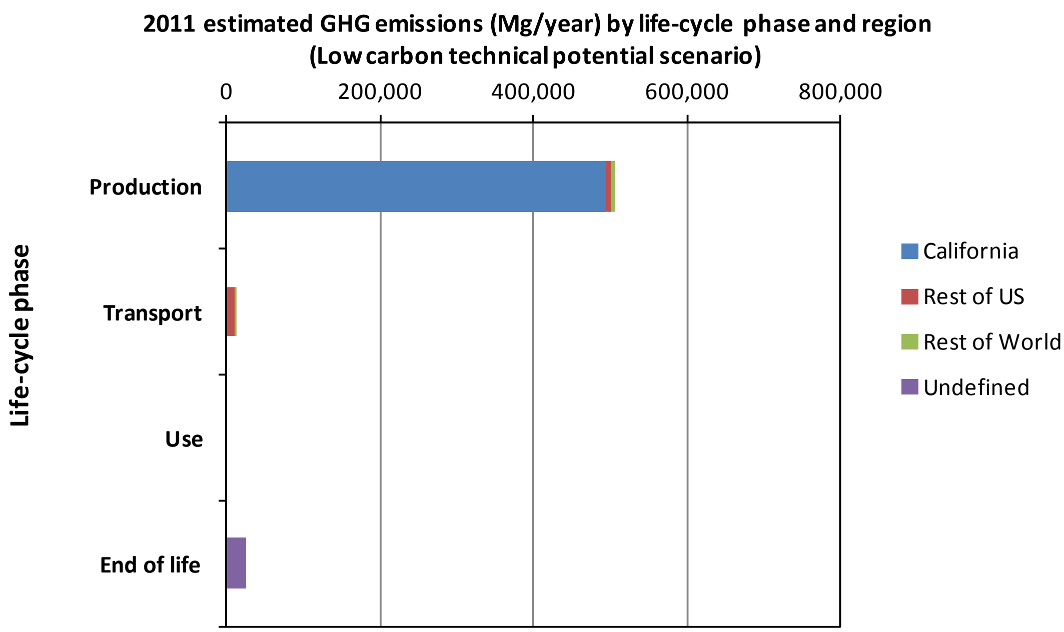
Estimated supply chain efficiency and GHG mitigation reductions by improvement opportunity and emissions source



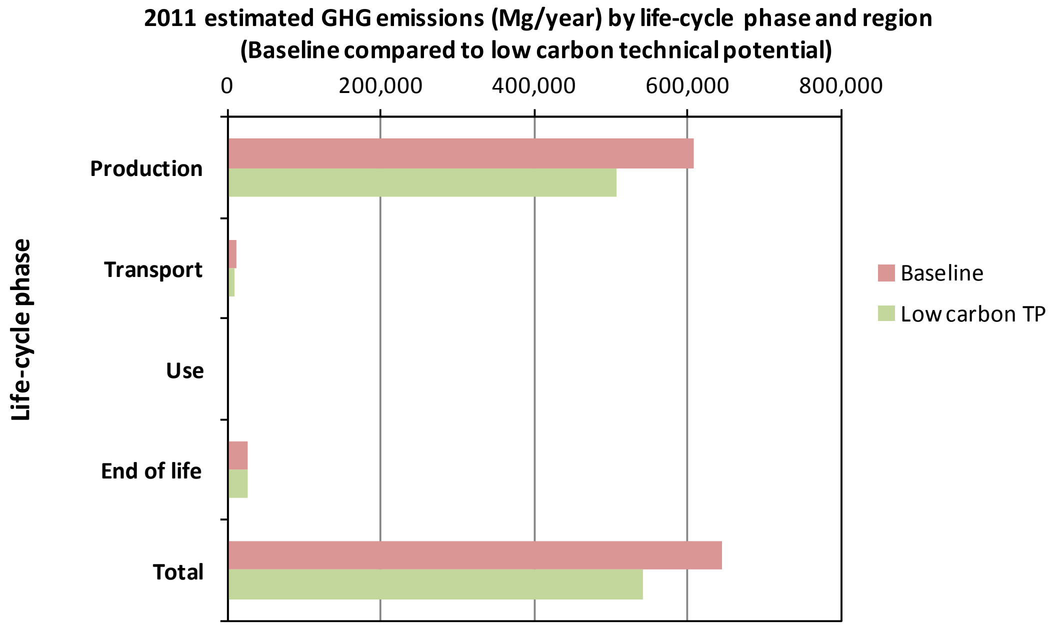
Estimated supply chain efficiency and GHG mitigation reductions by sector and emissions source



Estimated 2011 GHG emissions by life-cycle phase and region in the baseline scenario



2011 GHG emissions by life-cycle phase and region in the low carbon technical potential scenario



Side by side comparison of baseline and low carbon technical potential scenarios

Total savings = 16%

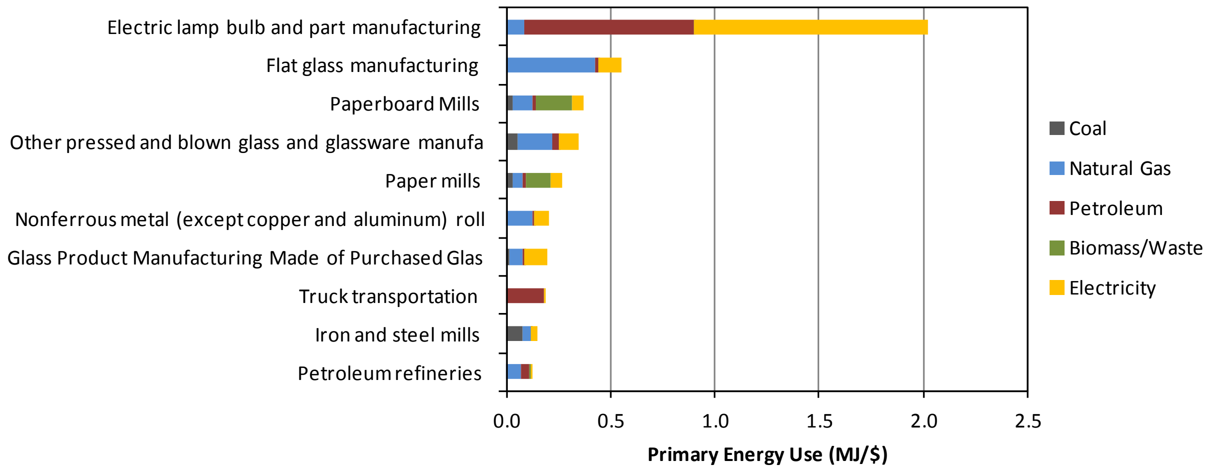
7: CFL Light bulb

Product

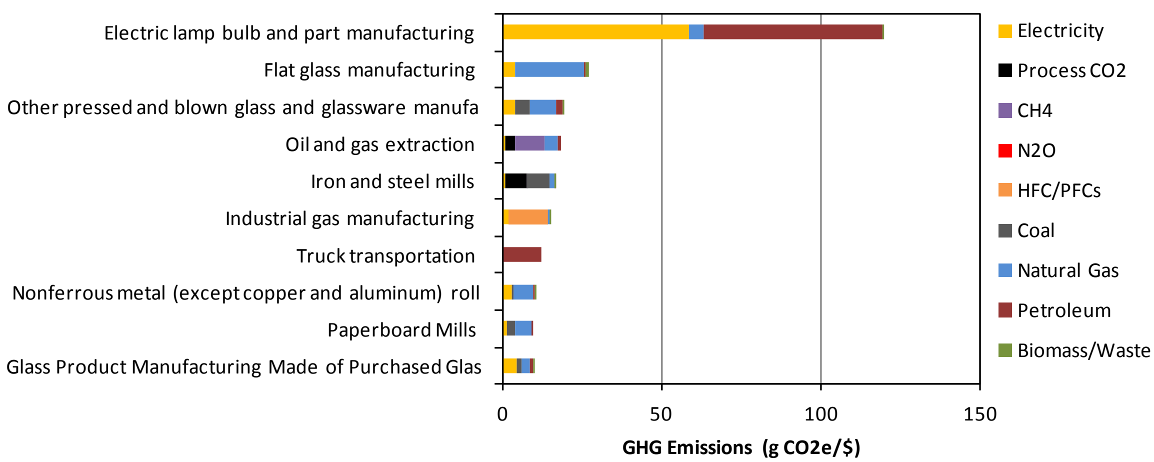
One 15 watt compact fluorescent light bulb

Life-cycle system description

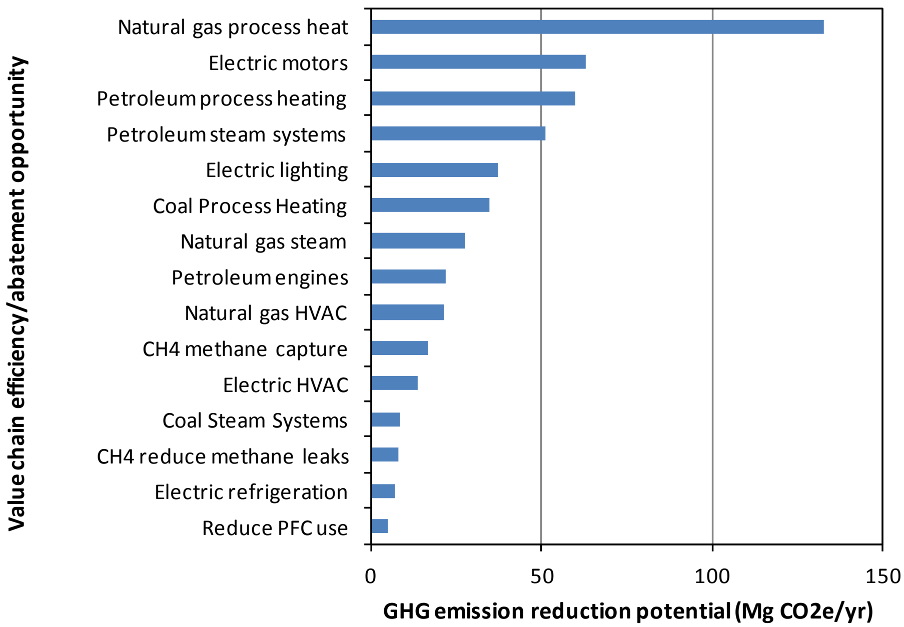
A CFL consists of glass for the tubing, aluminum and copper for caps, a printed circuit board, a plastic housing, and trace amounts of mercury in the vapor that fills the glass tubing. After manufacture, the CFL is typically packaged in either plastic or paper packaging for shipment to the retailer. The CFL consumes electricity throughout its useful life, and the amount of electricity consumed depends highly on the use patterns of the consumer. Under California law, CFLs must be discarded through an approved recycler.



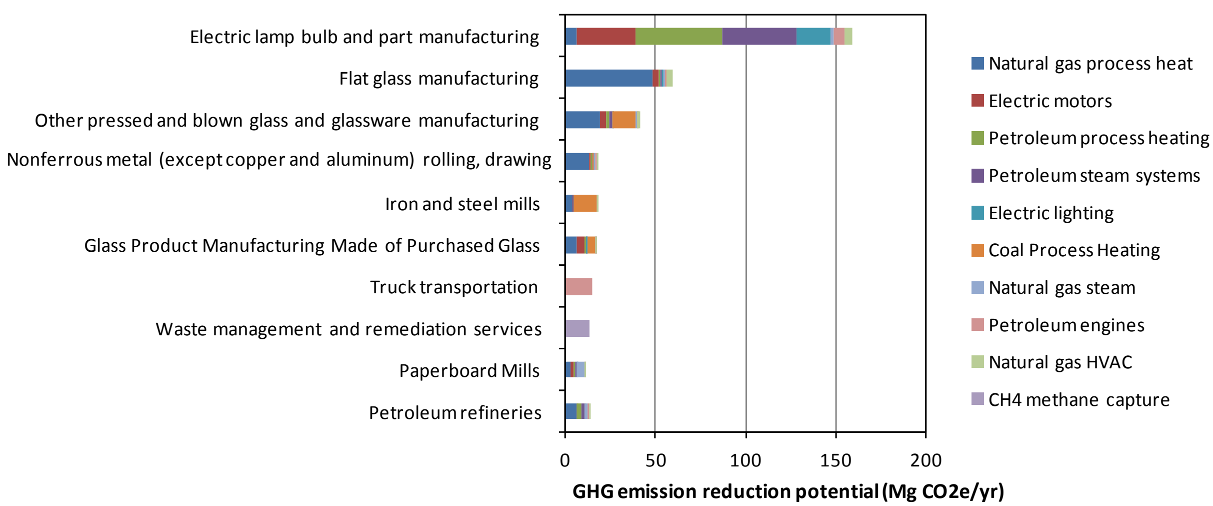
Top 10 contributing sectors to the energy footprint of product manufacture (MJ/$)



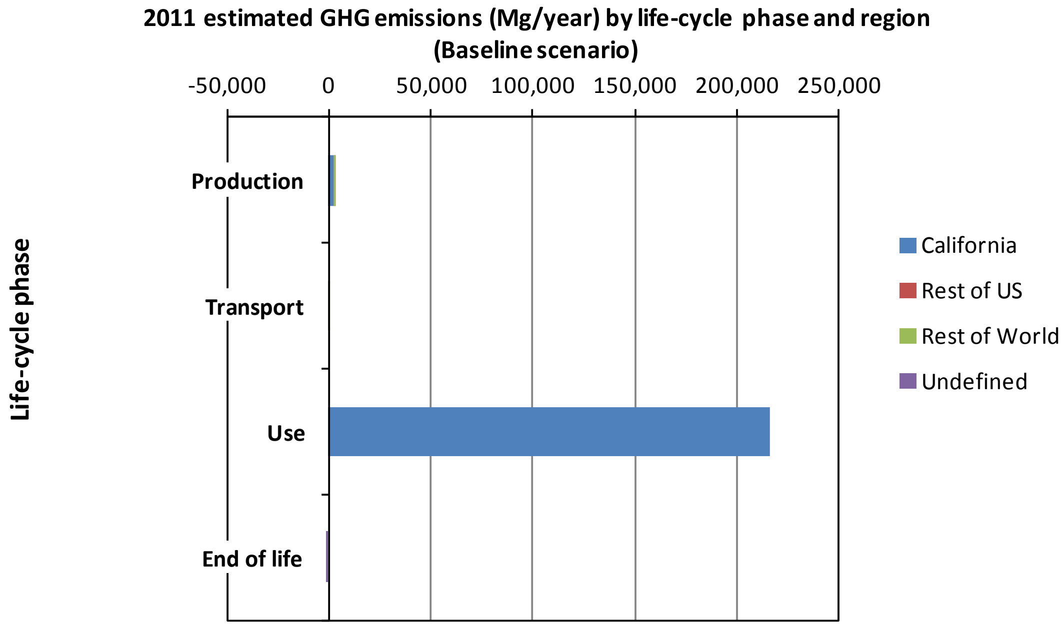
Top 10 contributing sectors to the GHG emissions footprint of product manufacture (MJ/$)



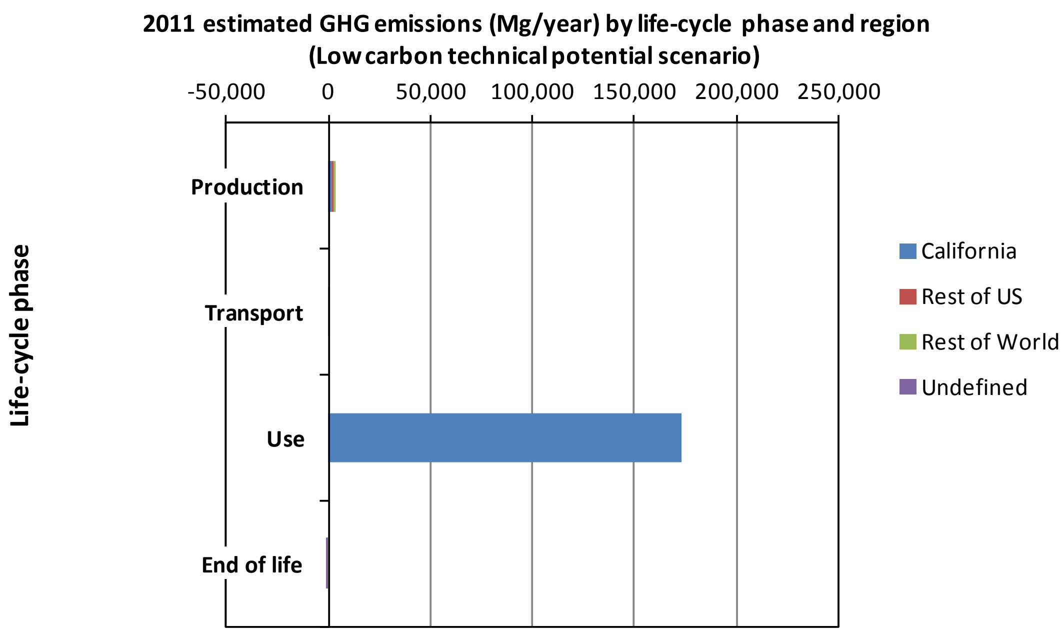
Estimated supply chain efficiency and GHG mitigation reductions by improvement opportunity and emissions source



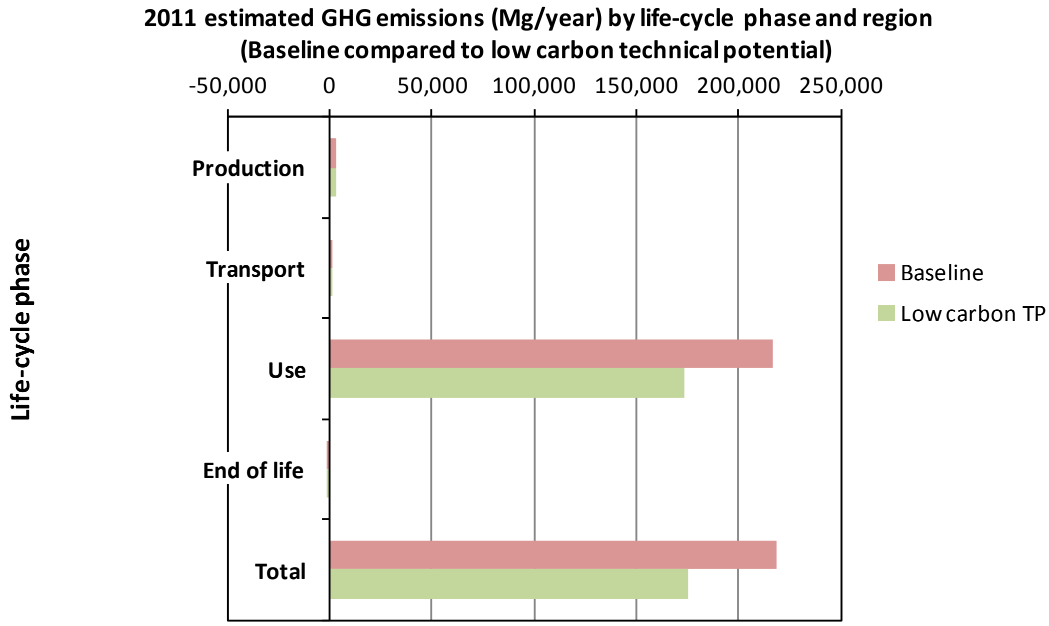
Estimated supply chain efficiency and GHG mitigation reductions by sector and emissions source



Estimated 2011 GHG emissions by life-cycle phase and region in the baseline scenario



2011 GHG emissions by life-cycle phase and region in the low carbon technical potential scenario



Side by side comparison of baseline and low carbon technical potential scenarios

Total savings = 20%

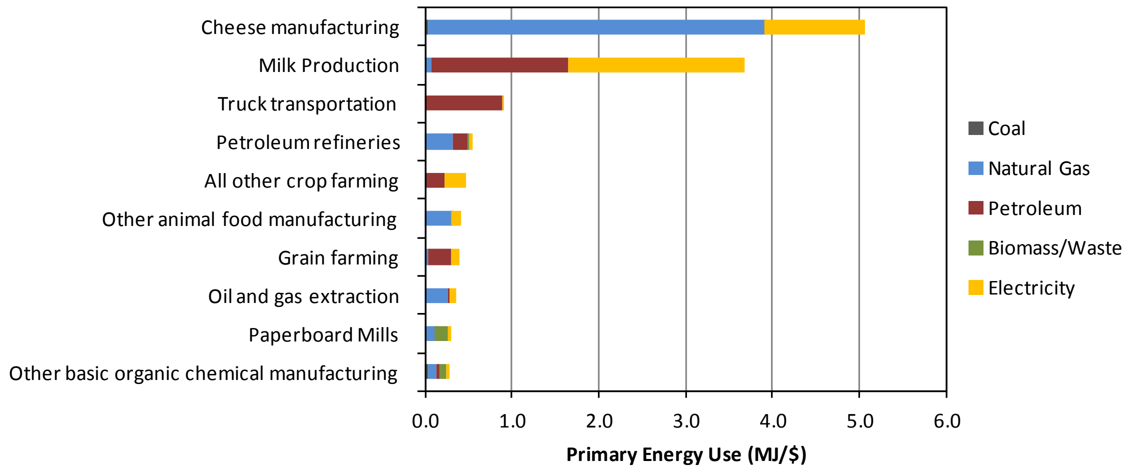
8: Cheese

Product

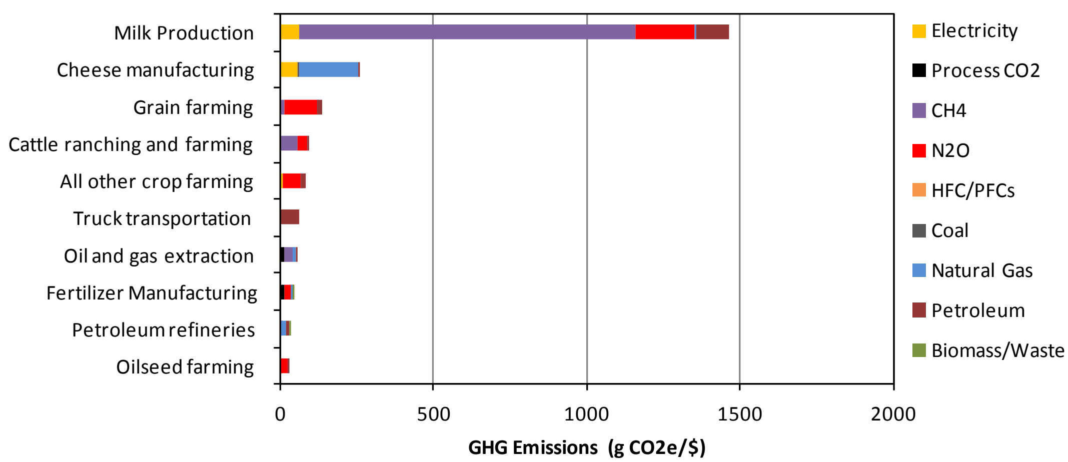
One kilogram of packaged natural cheese

Life-cycle system description

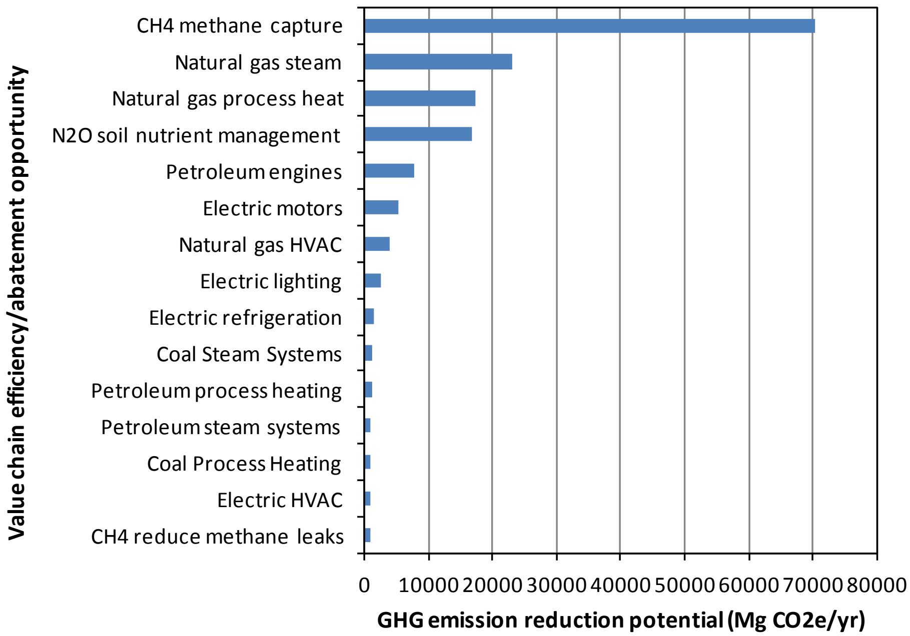
First, raw milk is produced at dairy farms and trucked to the dairy processer. Raw milk is clarified and standardized via centrifuge to achieve the specified level of milkfat. Once the desired composition is attained, the milk is pasteurized and filled into a cheese vat. In the cheese vat, rennet, enzymes, and/or bacterial cultures are added, depending on the type of cheese to be made. The mixture is then cooked to facilitate the biological processes that create cheese curds. Additional cooking is commonly used to "age" the cheese to the desired taste and attributes. After the cooking step, the curds are drained from the liquid whey byproduct of the cheesemaking process. Due to its high Biochemical Oxygen Content (BOC), whey is expensive to dispose of in liquid form. Because liquid whey is also perishable, the most common method of handling whey is to dry it into a powder, which is an energy intensive process. After draining, the curds are pressed together to give solid blocks. Some cheeses, such as mozzarella, are also stretched to give a "stringy" texture. The cheese is then packaged, aged, and stored prior to refrigerated shipment to the retailer (Brush et al. 2011). Cheese is then refrigerated at the retailer and by the consumer prior to consumption. The plastic packaging is sent to landfill at end of life. Depending on the consumer, the cheese may be fully eaten or partially disposed as waste, which can either be landfilled or composted.



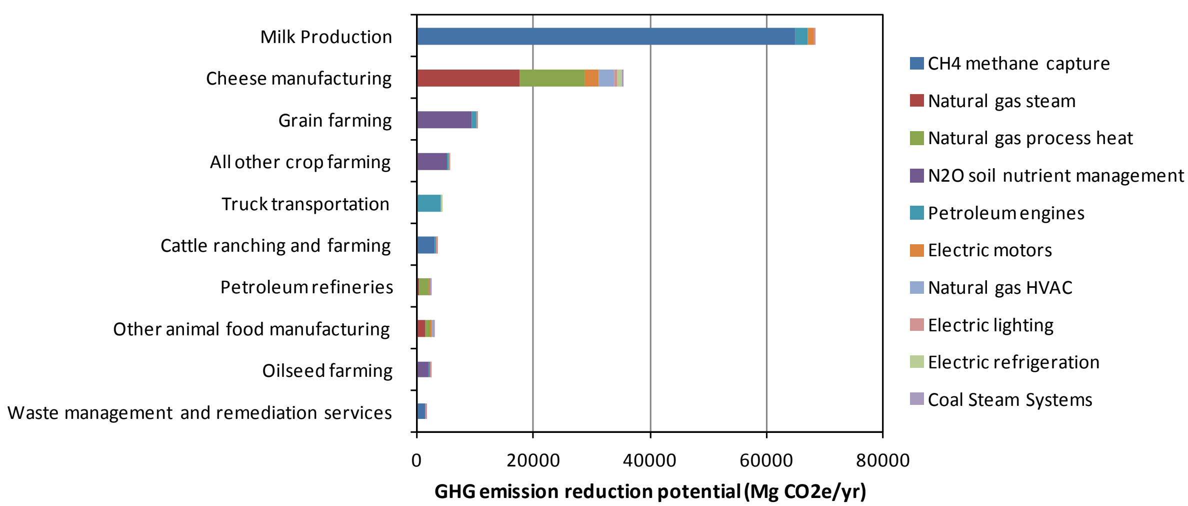
Top 10 contributing sectors to the energy footprint of product manufacture (MJ/$)



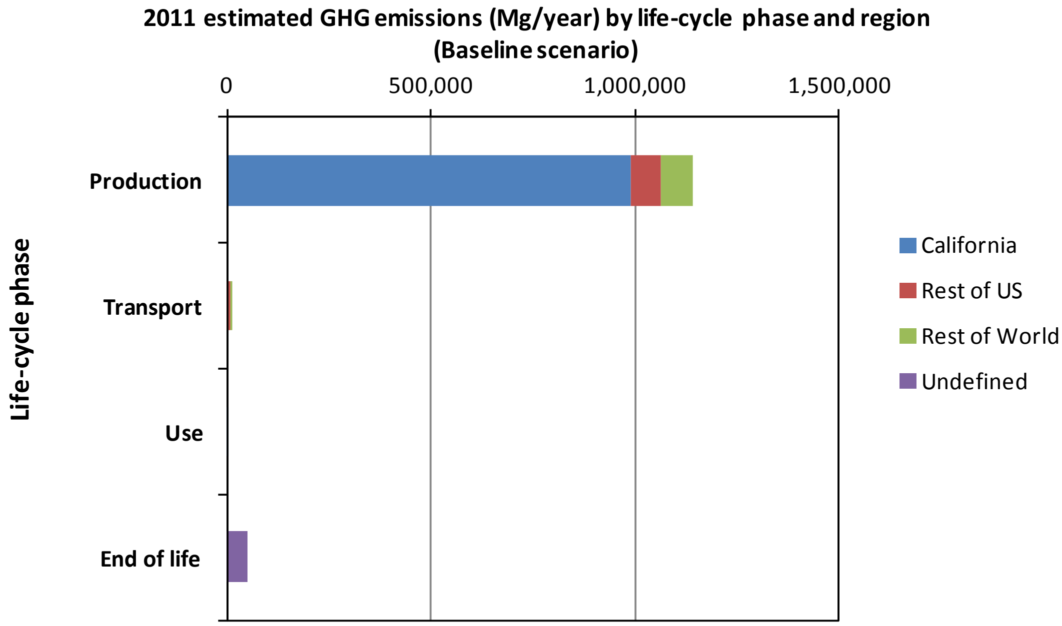
Top 10 contributing sectors to the GHG emissions footprint of product manufacture (MJ/$)



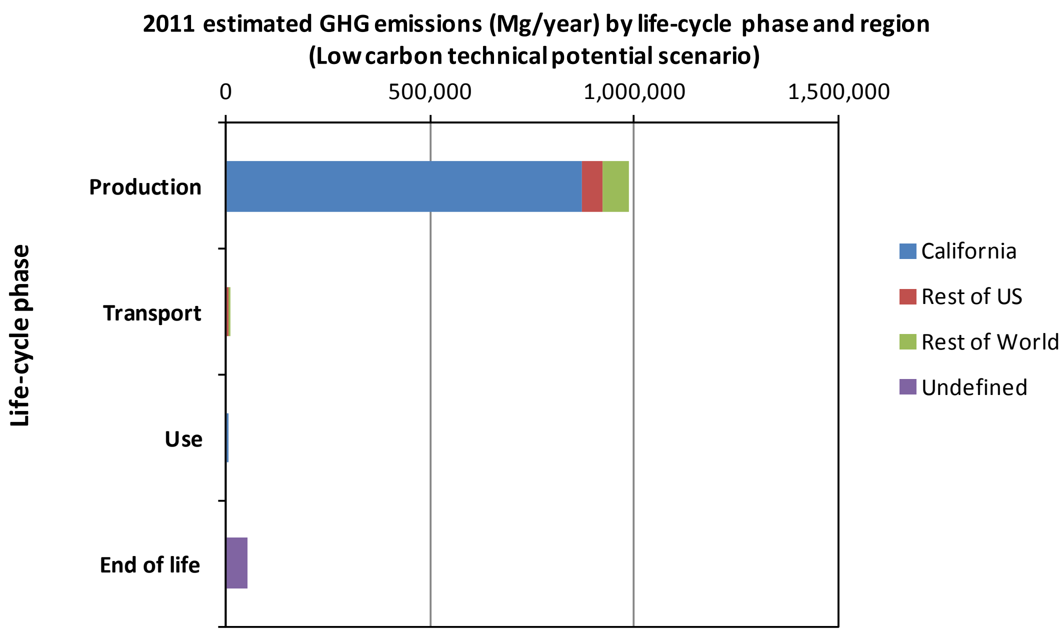
Estimated supply chain efficiency and GHG mitigation reductions by improvement opportunity and emissions source



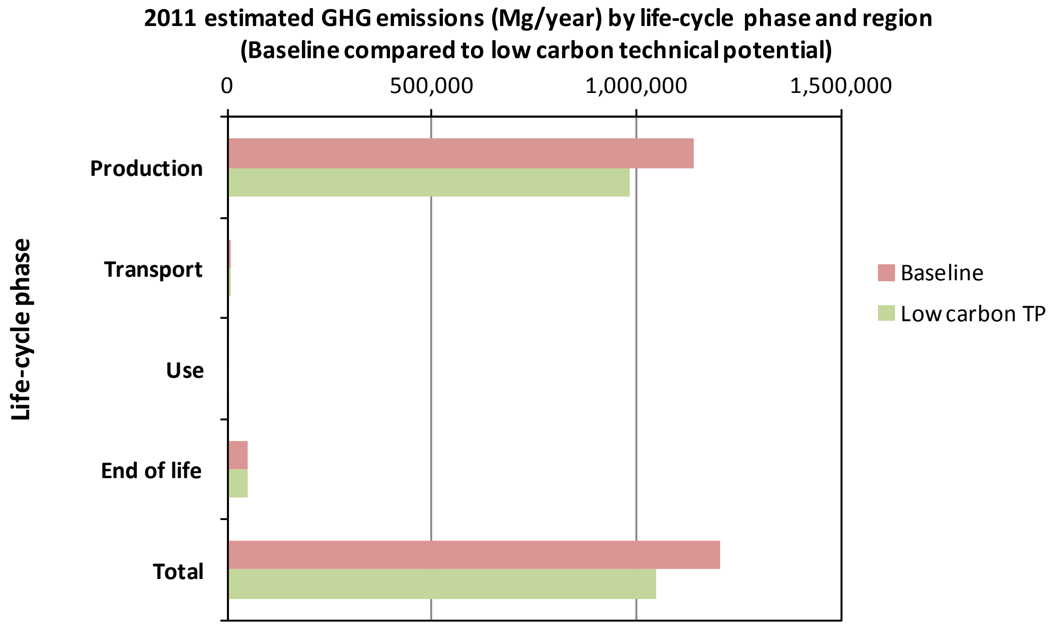
Estimated supply chain efficiency and GHG mitigation reductions by sector and emissions source



Estimated 2011 GHG emissions by life-cycle phase and region in the baseline scenario



2011 GHG emissions by life-cycle phase and region in the low carbon technical potential scenario



Side by side comparison of baseline and low carbon technical potential scenarios

Total savings = 14%

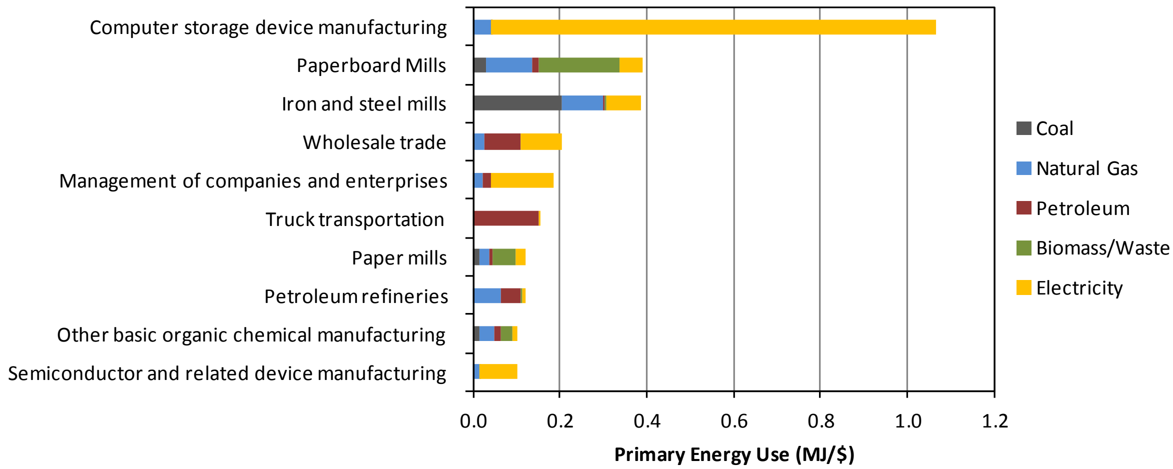
9: External hard drive

Product

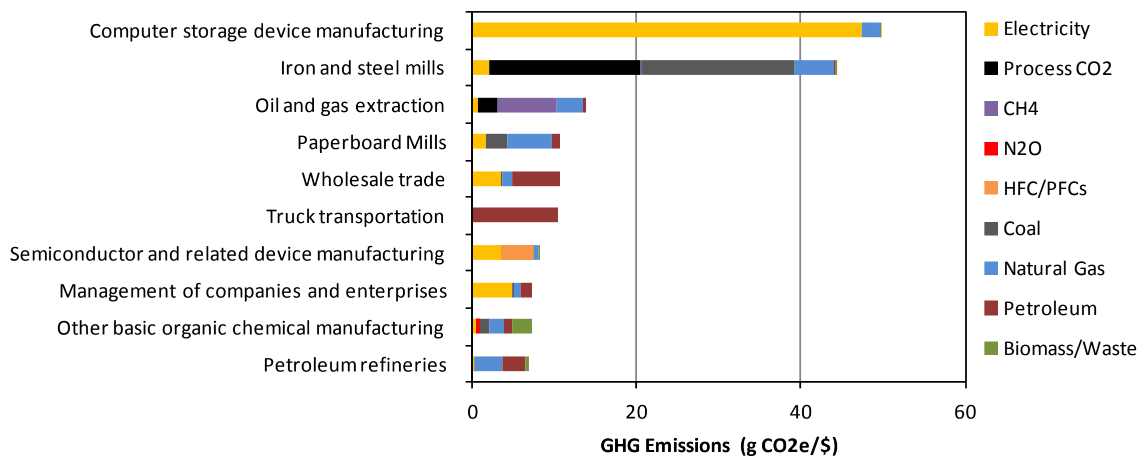
One external hard disk drive

Life-cycle system description

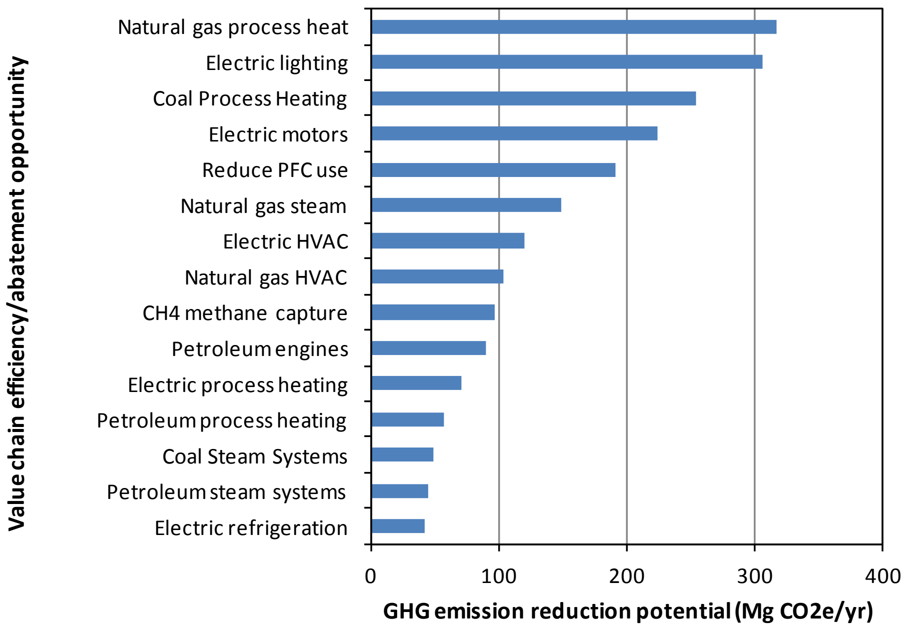
A typical hard drive assembly consists of several disk platters (which store the data), a read/write head actuator assembly (which writes and reads the data), a circuit board for control and data interfacing, motors to spin the platters and move the actuator assembly, a metal frame to contain the assembly, and a metal assembly cover. An external unit will contain a plastic housing to package the hard drive assembly, and a power supply. Similar to computers, a number of these components are made by dedicated supply chains; the components are then shipped to an assembly facility that assembles the final product, packages it, tests it, and ships it to the retail outlet. The drive consumes electricity throughout its useful life, and the amount of electricity consumed depends highly on the use patterns of the consumer. Under California's e-waste recycling laws, drives must be discarded through an approved electronics recycler. Depending on recycling practices, the unit can either be shredded to recover its metals content or manually disassembled for component and materials recovery, either domestically or overseas.



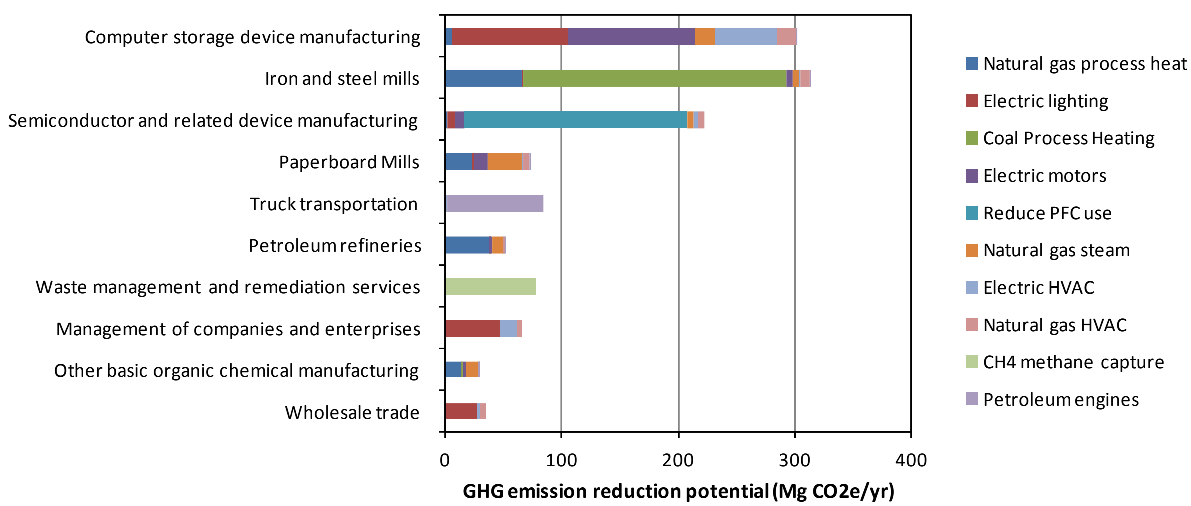
Top 10 contributing sectors to the energy footprint of product manufacture (MJ/$)



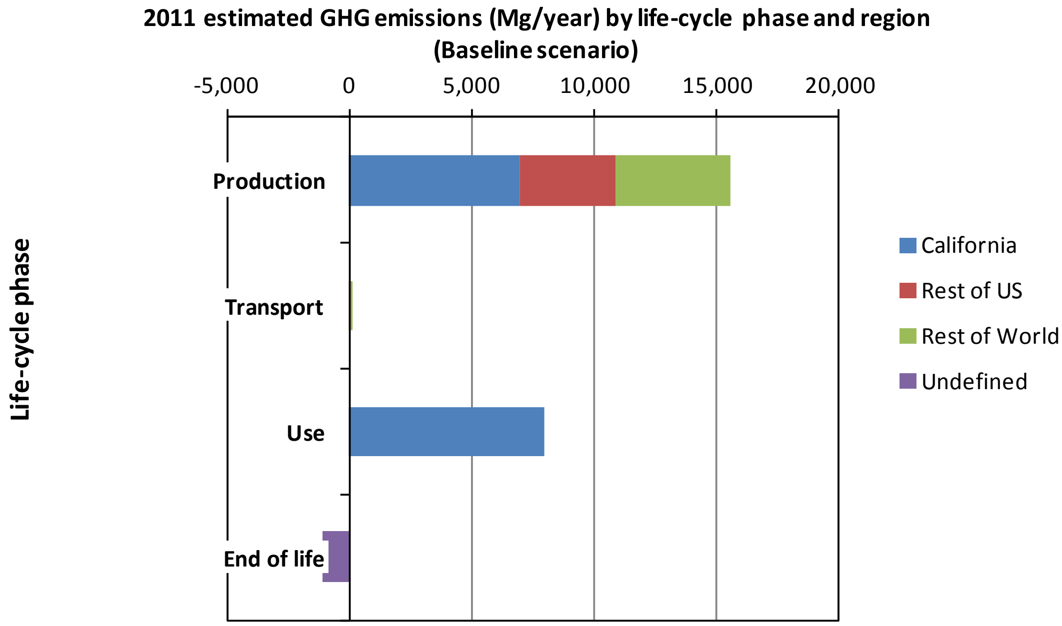
Top 10 contributing sectors to the GHG emissions footprint of product manufacture (MJ/$)



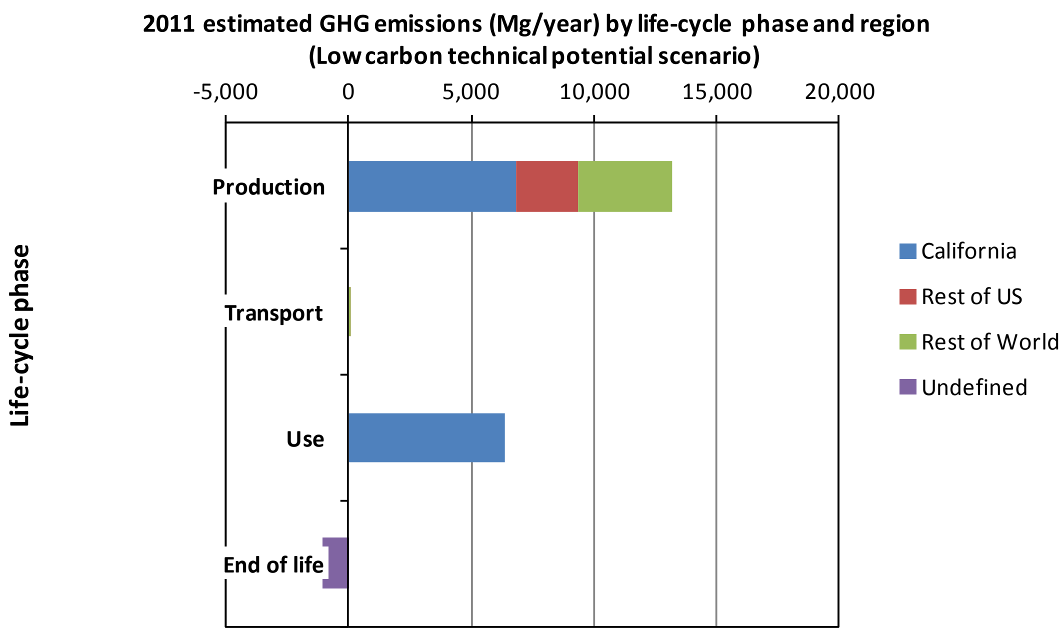
Estimated supply chain efficiency and GHG mitigation reductions by improvement opportunity and emissions source



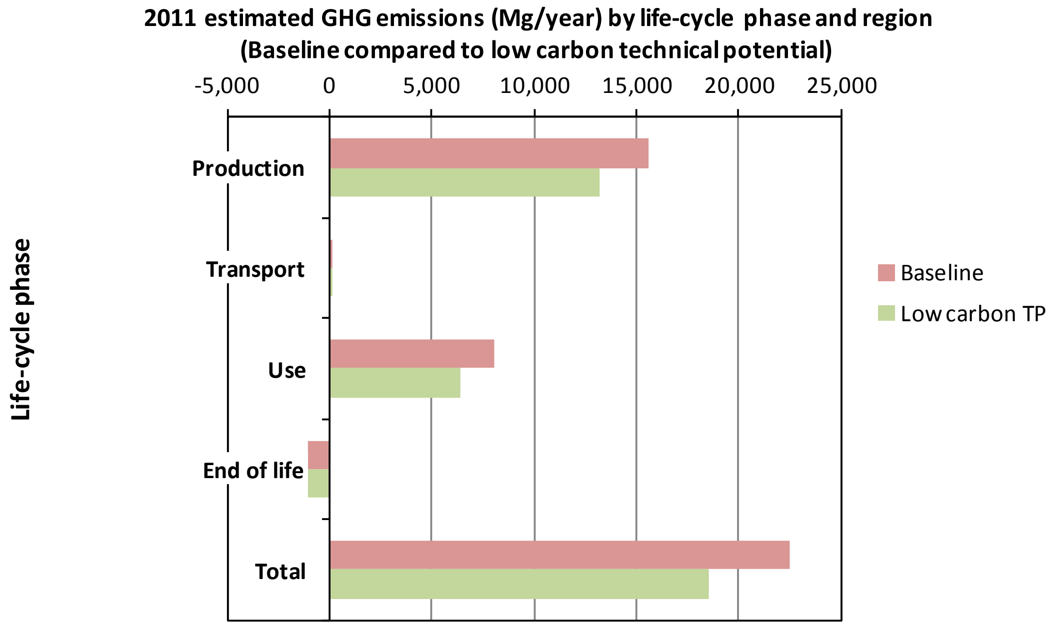
Estimated supply chain efficiency and GHG mitigation reductions by sector and emissions source



Estimated 2011 GHG emissions by life-cycle phase and region in the baseline scenario



2011 GHG emissions by life-cycle phase and region in the low carbon technical potential scenario



Side by side comparison of baseline and low carbon technical potential scenarios

Total savings = 18%

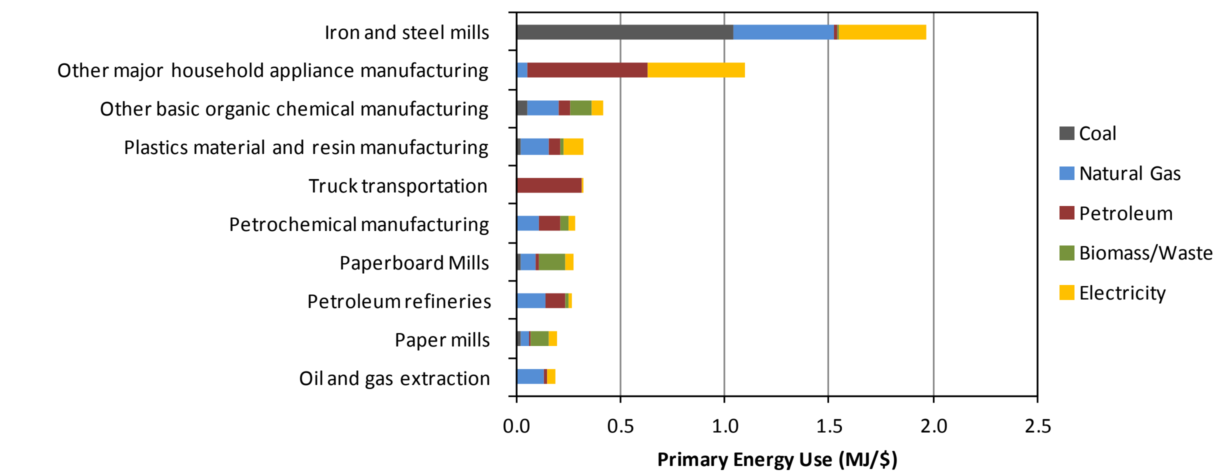
10: Hot water heater

Product

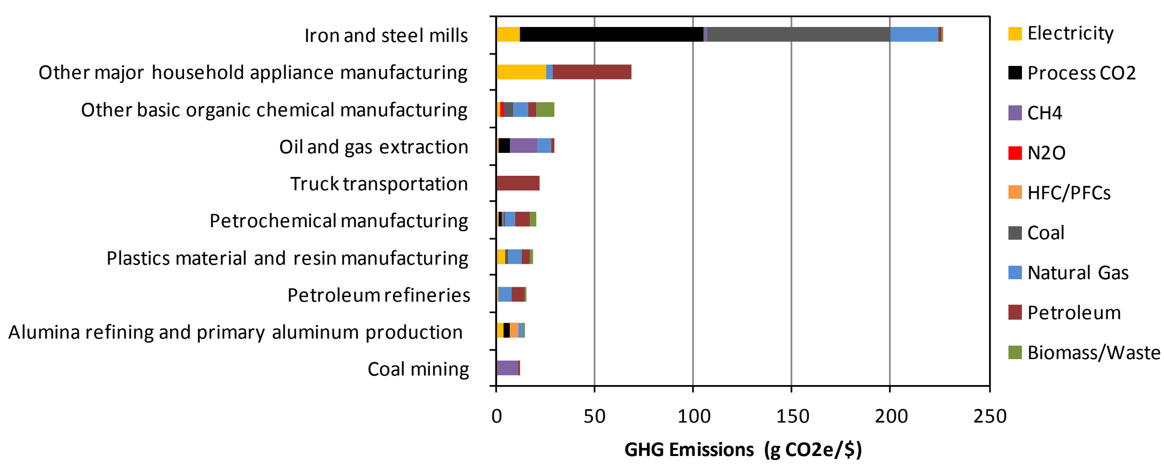
One gas fired tank storage water heater

Life-cycle system description

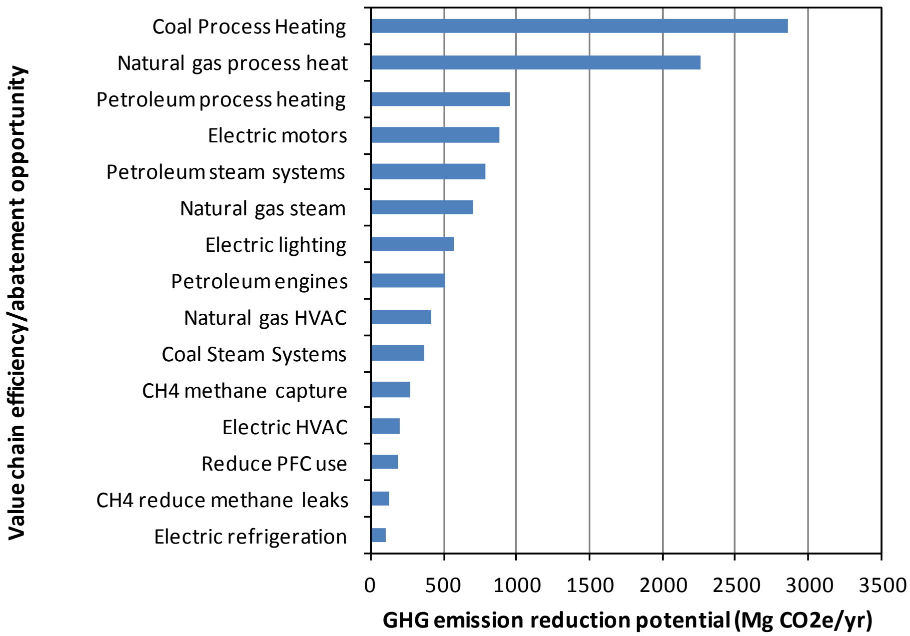
The primary materials in a tank storage water heater are steel, which is used for the tank body and structure, and rigid polyurethane, which is for insulation material. However, there are other materials contained in a water heater, including aluminum, brass, and other plastics (Lu et al. 2011). Water heater components are made at multiple suppliers, and shipped to a final assembly facility that assembles and packages the finished units for shipment to the retailer. The vast majority of a water heater's life-cycle energy use and GHG emissions occurs during the use phase. Most tank storage water heaters will be used for around 12 years (Lu et al. 2011), after which time they will be discarded. In California, discarded water heaters cannot be sent to landfill.



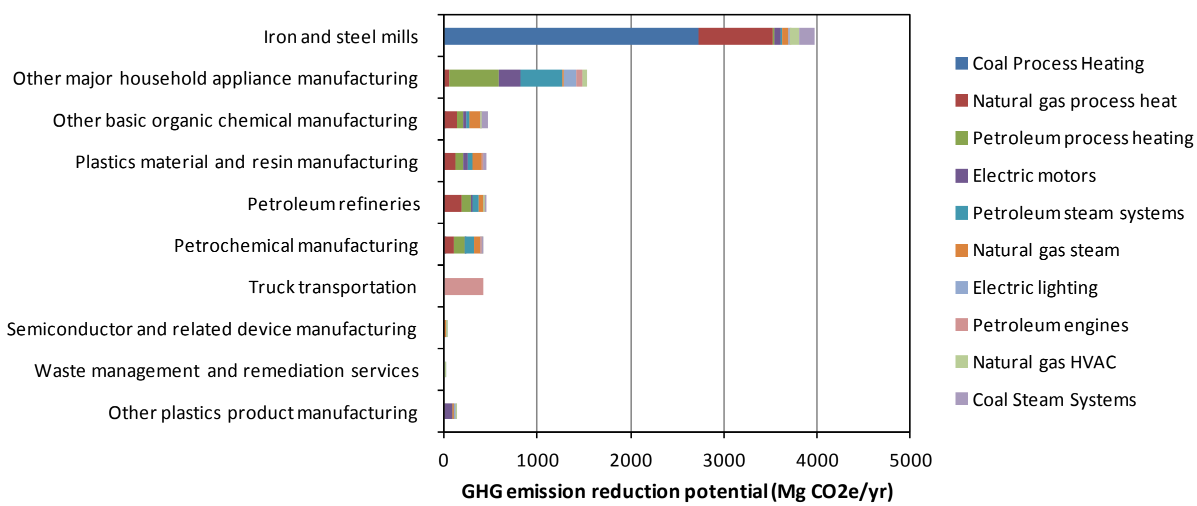
Top 10 contributing sectors to the energy footprint of product manufacture (MJ/$)



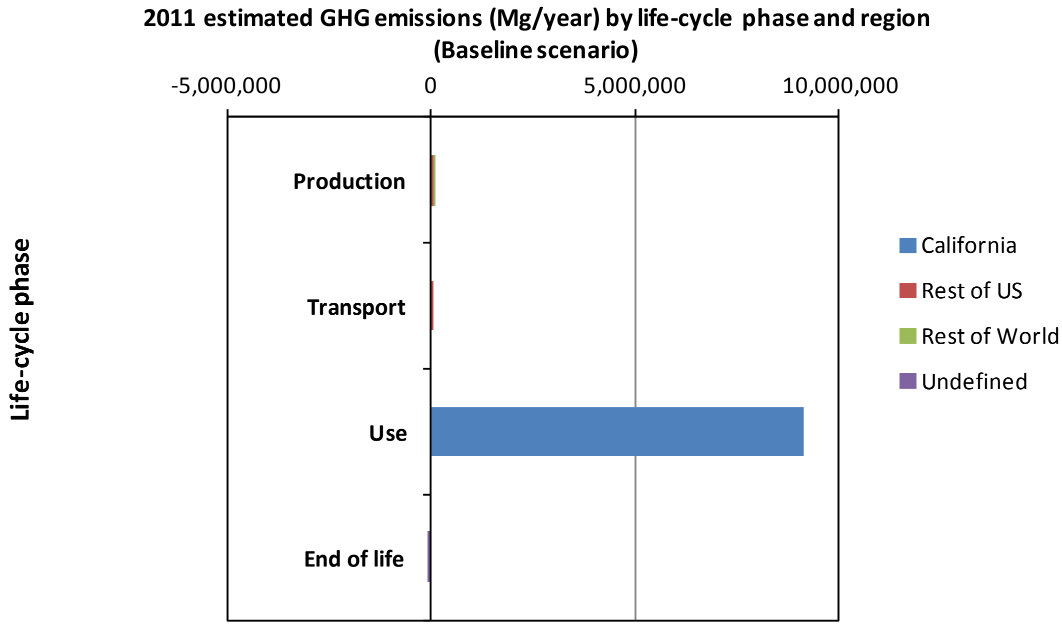
Top 10 contributing sectors to the GHG emissions footprint of product manufacture (MJ/$)



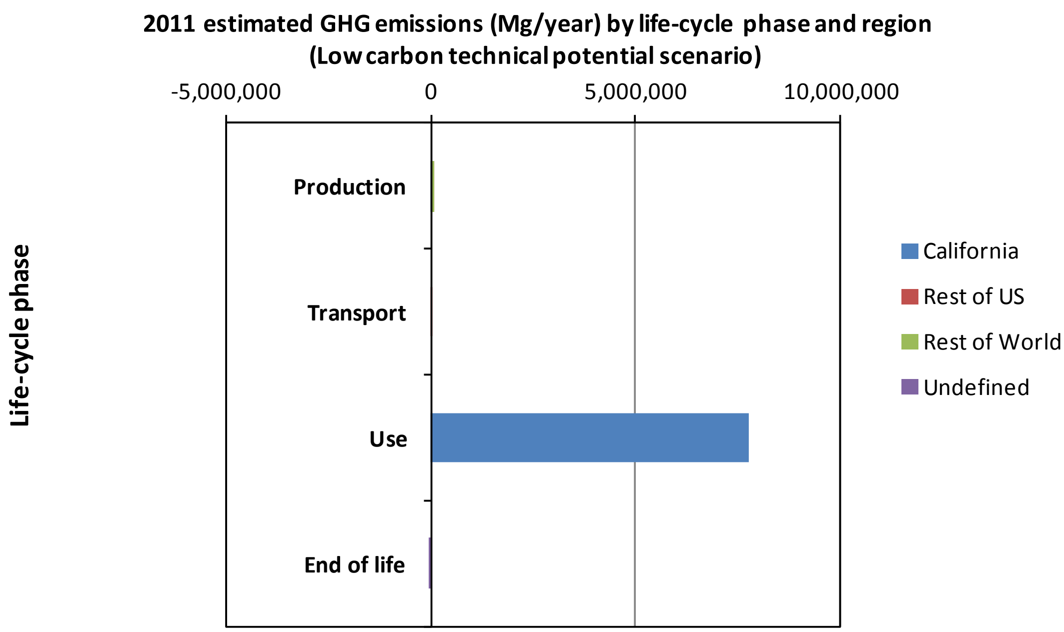
Estimated supply chain efficiency and GHG mitigation reductions by improvement opportunity and emissions source



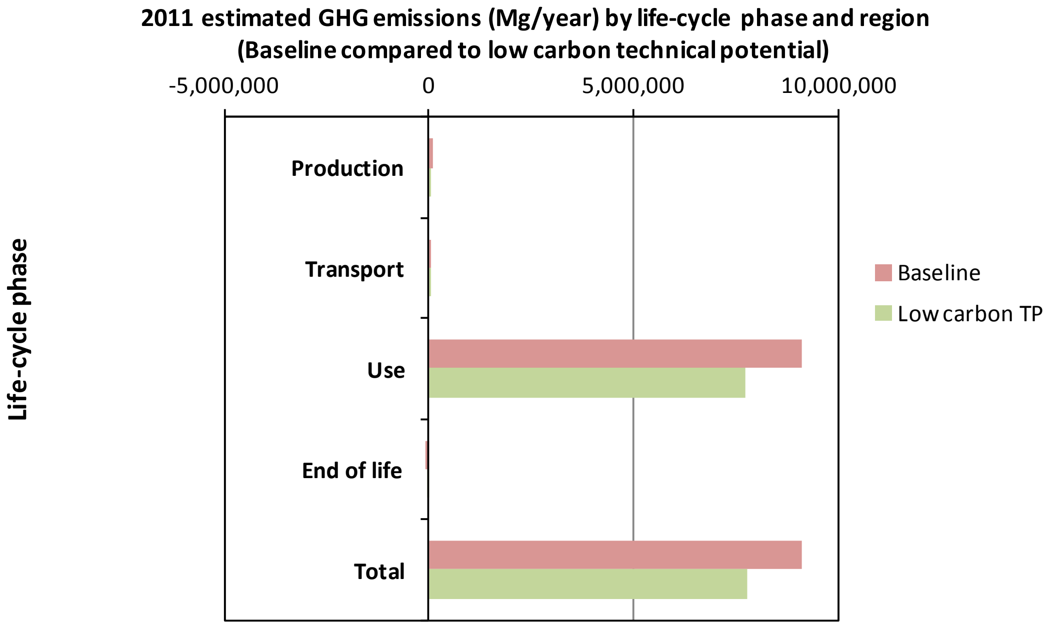
Estimated supply chain efficiency and GHG mitigation reductions by sector and emissions source



Estimated 2011 GHG emissions by life-cycle phase and region in the baseline scenario



2011 GHG emissions by life-cycle phase and region in the low carbon technical potential scenario



Side by side comparison of baseline and low carbon technical potential scenarios

Total savings = 15%

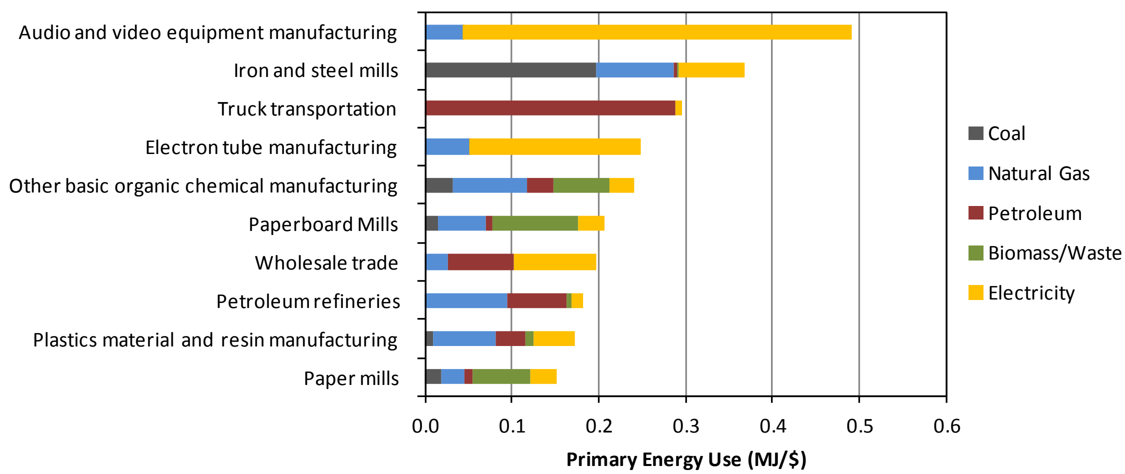
11: LCD Flat panel TV

Product

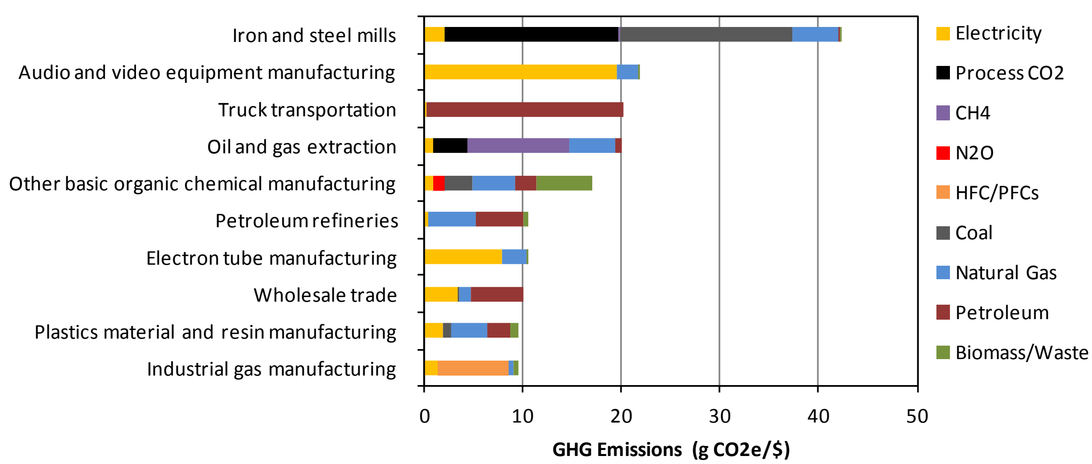
One LCD flat panel television

Life-cycle system description

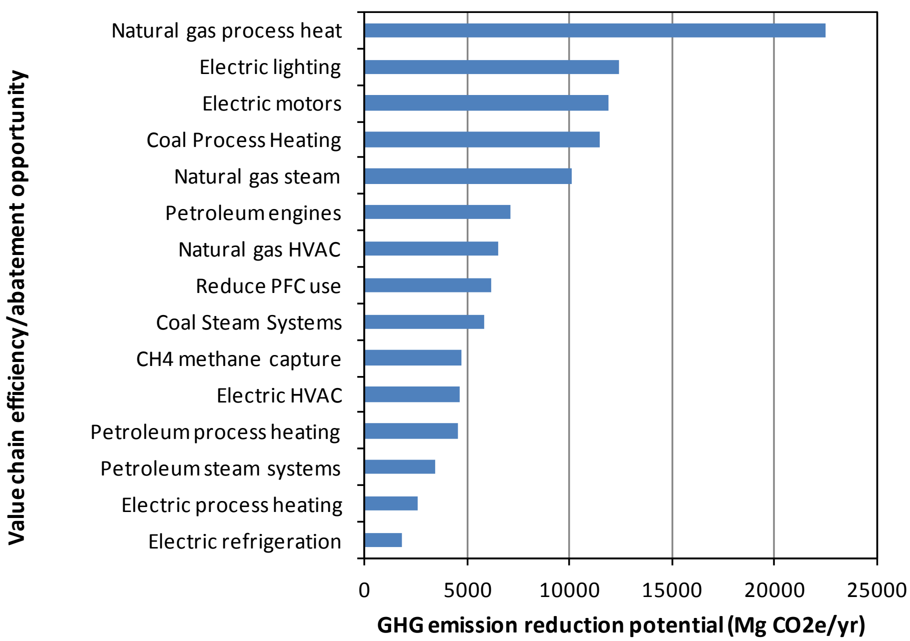
An LCD flat panel television contains a liquid crystal display panel, a backlight assembly, circuit boards, a power supply, and steel and plastic for its structure and housing (Fraunhofer 2007). Most of the main components are made by dedicated supply chains; the components are then shipped to the assembly facility that assembled the final product, packages it, and ships it to the retail outlet. There are dozens of components and materials in a flat panel TV, but the most energy and GHG emissions intensive components to manufacture are the LCD panel and the electronics (Fraunhofer 2007). The LCD flat panel TV consumes electricity throughout its useful life, and the amount of electricity consumed depends highly on the use patterns of the consumer. Under California's e-waste recycling laws, flat panel TVs must be discarded through an approved electronics recycler.



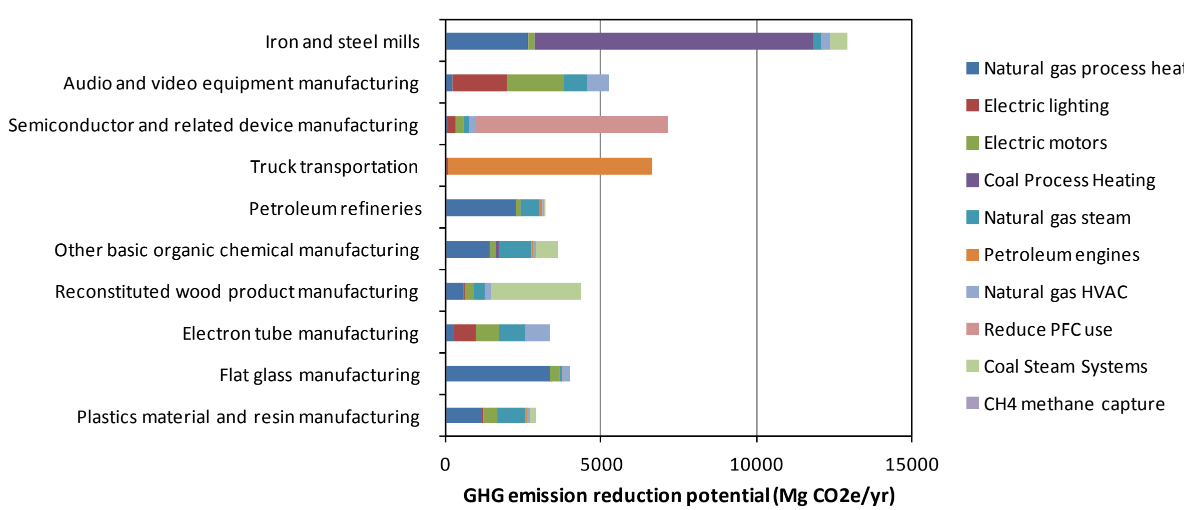
Top 10 contributing sectors to the energy footprint of product manufacture (MJ/$)



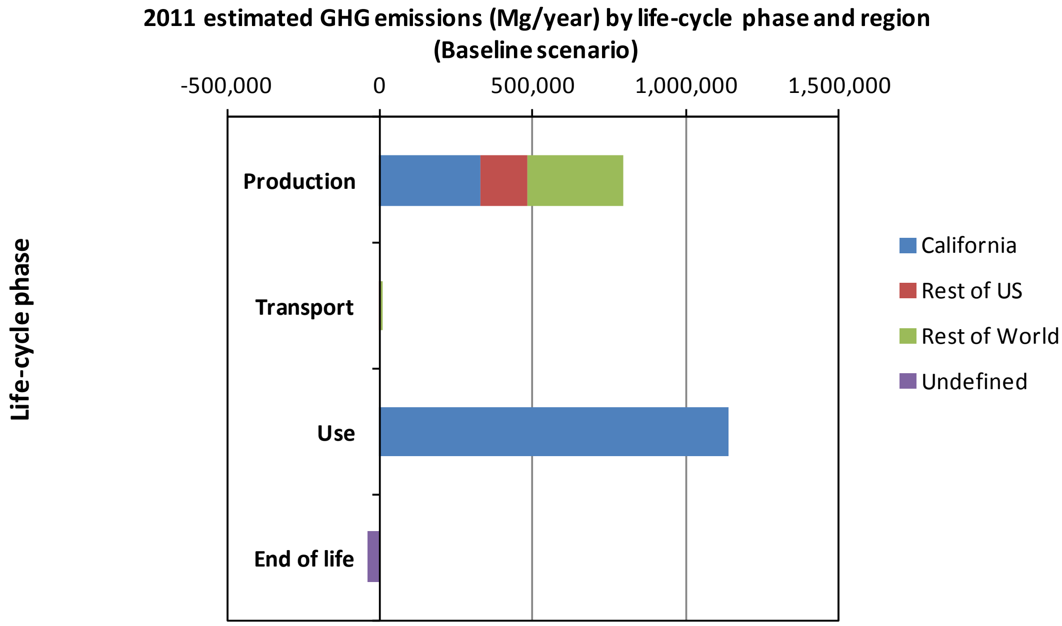
Top 10 contributing sectors to the GHG emissions footprint of product manufacture (MJ/$)



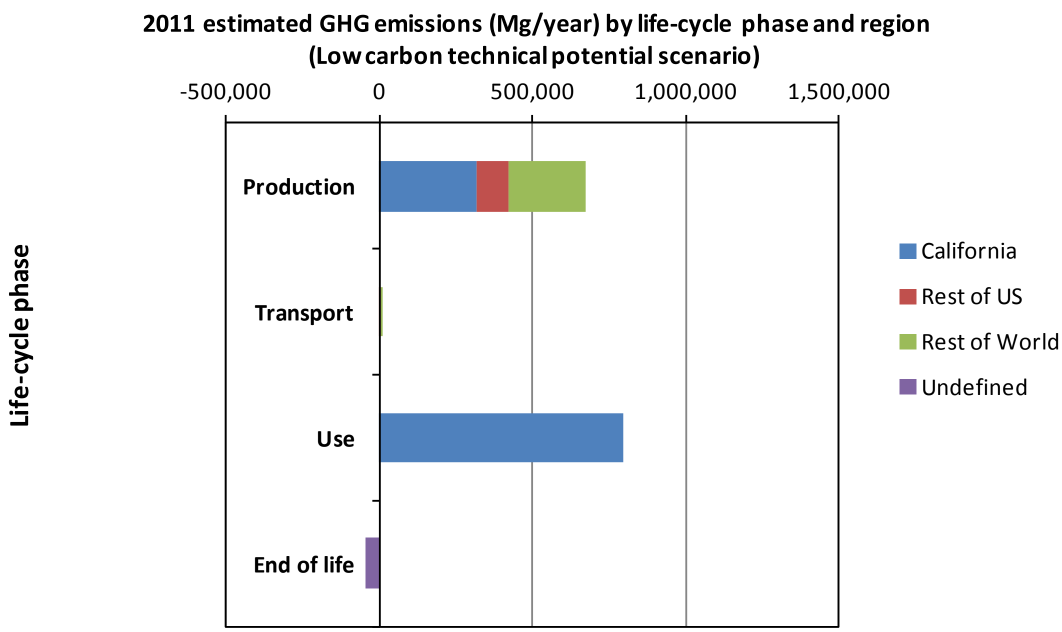
Estimated supply chain efficiency and GHG mitigation reductions by improvement opportunity and emissions source



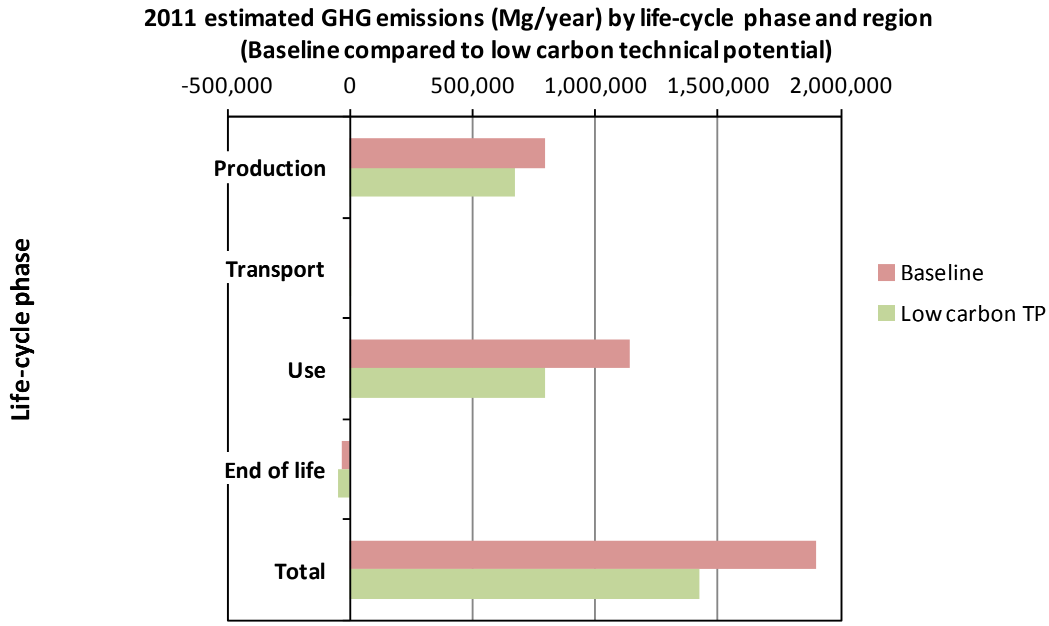
Estimated supply chain efficiency and GHG mitigation reductions by sector and emissions source



Estimated 2011 GHG emissions by life-cycle phase and region in the baseline scenario



2011 GHG emissions by life-cycle phase and region in the low carbon technical potential scenario



Side by side comparison of baseline and low carbon technical potential scenarios

Total savings = 25%

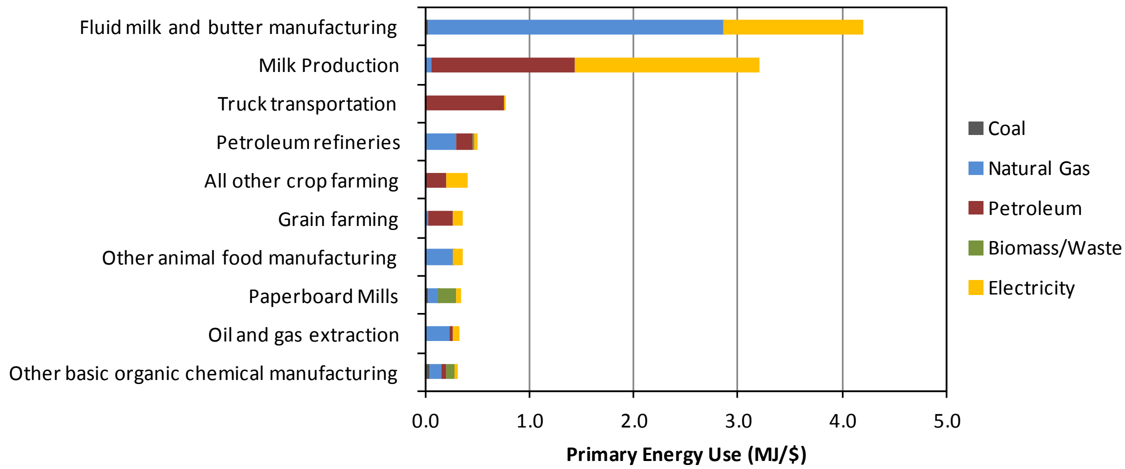
12: Milk

Product

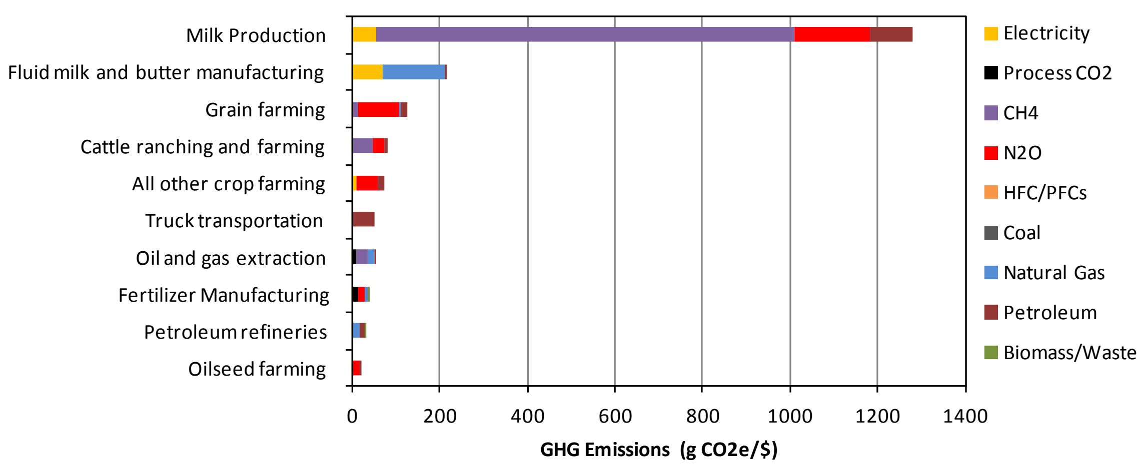
One gallon of milk packaged in an HDPE bottle

Life-cycle system description

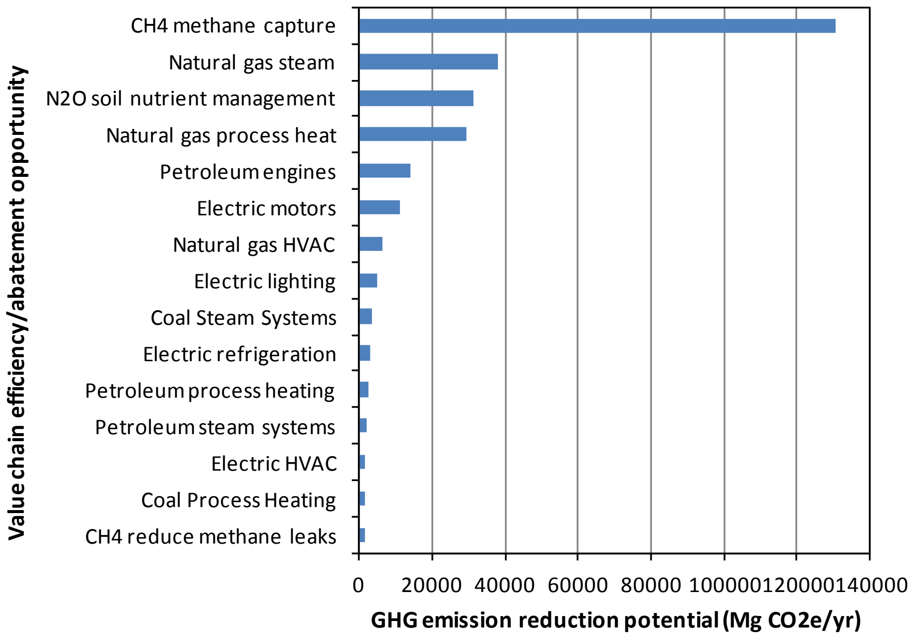
First, raw milk is produced at dairy farms and trucked to the dairy processer. Upon entering the facility, raw milk is clarified and cooled a few degrees prior to being transferred to cooled storage tanks. To produce pasteurized milk, the most common type in the U.S., the milk is standardized and pasteurized, with the homogenization step usually occurring prior to being cooled back down. The cooled milk is then packaged and kept in refrigerated storage until shipment (Brush et al. 2011). Shipment typically occurs by refrigerated truck to the retail outlet. HDPE bottles are manufactured from plastic resin, which is heated and molded into a finished bottle. The product is refrigerated at home prior to consumption. HDPE milk bottles are not subject to the California Redemption Value (CRV) fee, but can be recycled by the consumer. Thus, the recycling rate is expected to be lower than beverage containers covered by the CRV, which experience very high recycling rates in the state.



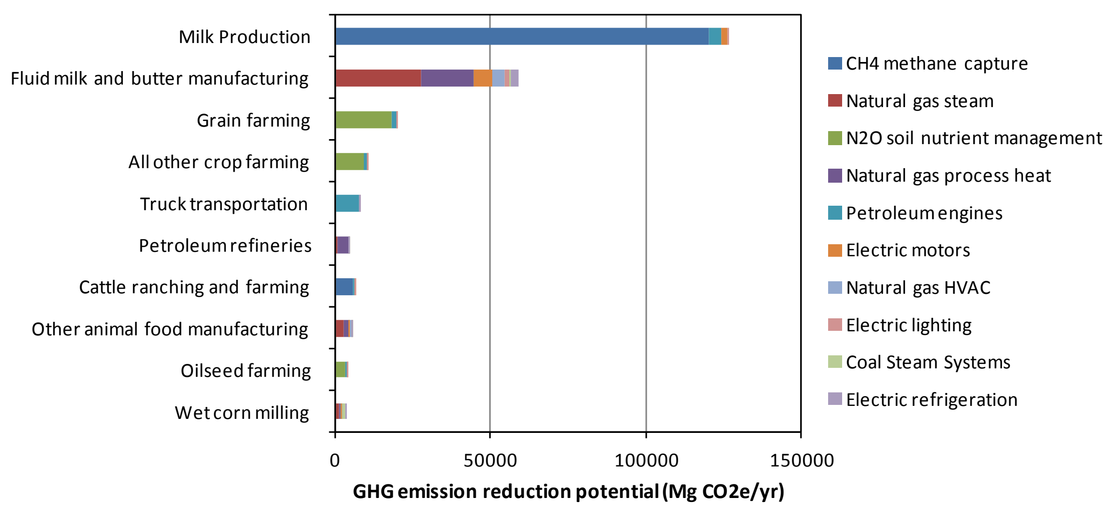
Top 10 contributing sectors to the energy footprint of product manufacture (MJ/$)



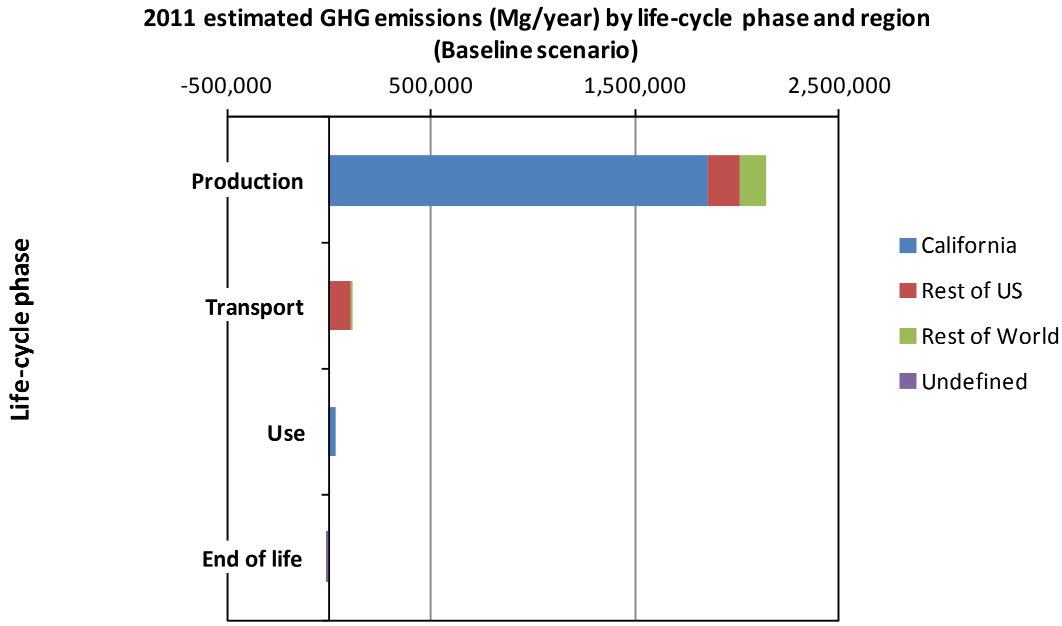
Top 10 contributing sectors to the GHG emissions footprint of product manufacture (MJ/$)



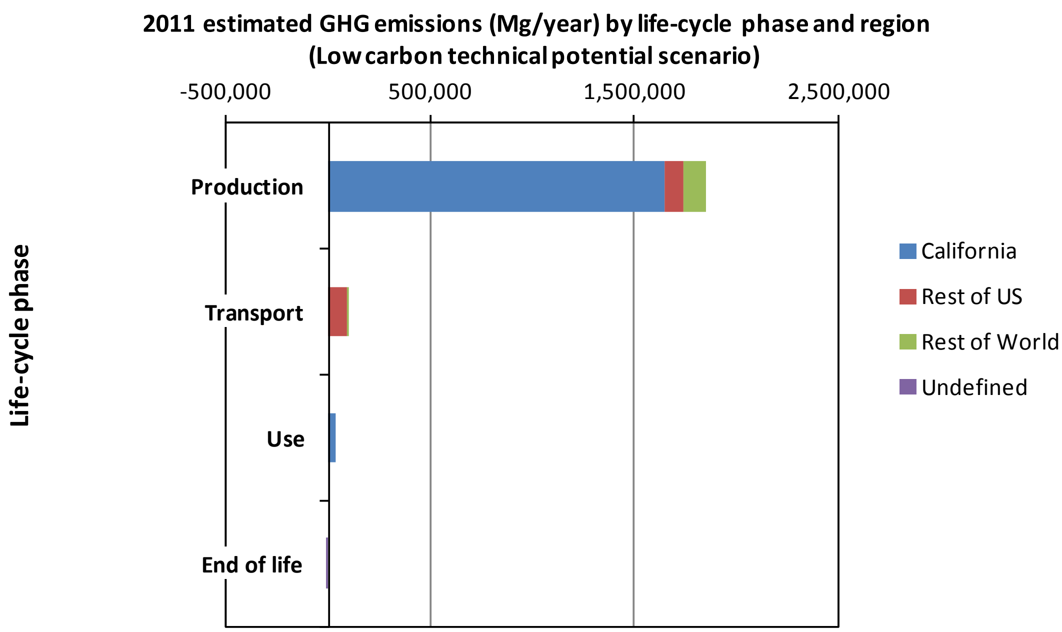
Estimated supply chain efficiency and GHG mitigation reductions by improvement opportunity and emissions source



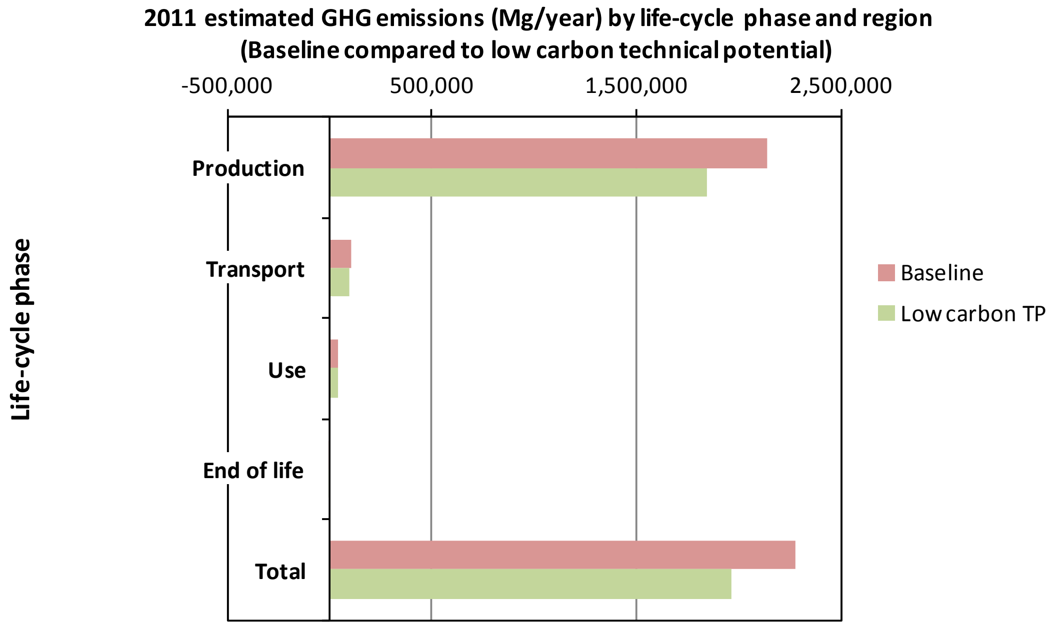
Estimated supply chain efficiency and GHG mitigation reductions by sector and emissions source



Estimated 2011 GHG emissions by life-cycle phase and region in the baseline scenario



2011 GHG emissions by life-cycle phase and region in the low carbon technical potential scenario



Side by side comparison of baseline and low carbon technical potential scenarios

Total savings = 15%

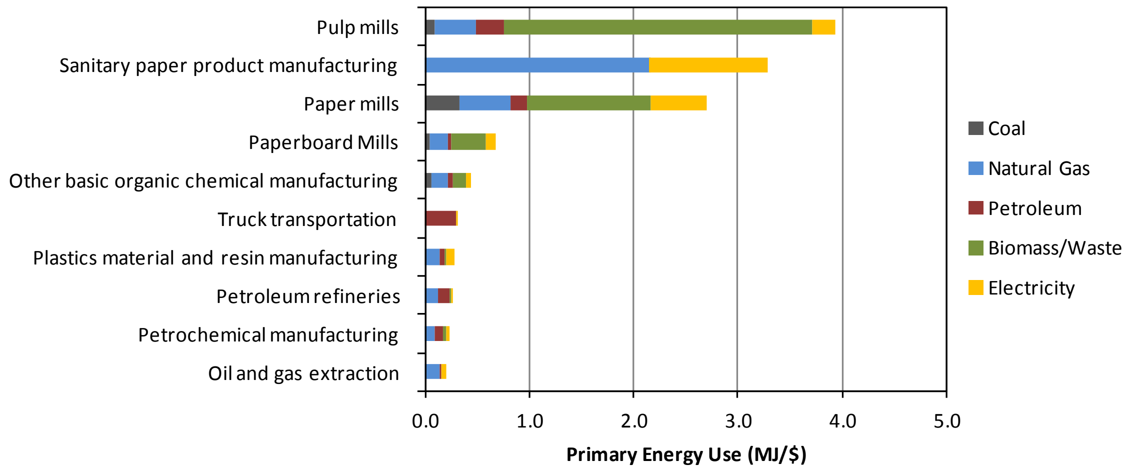
13: Paper towels

Product

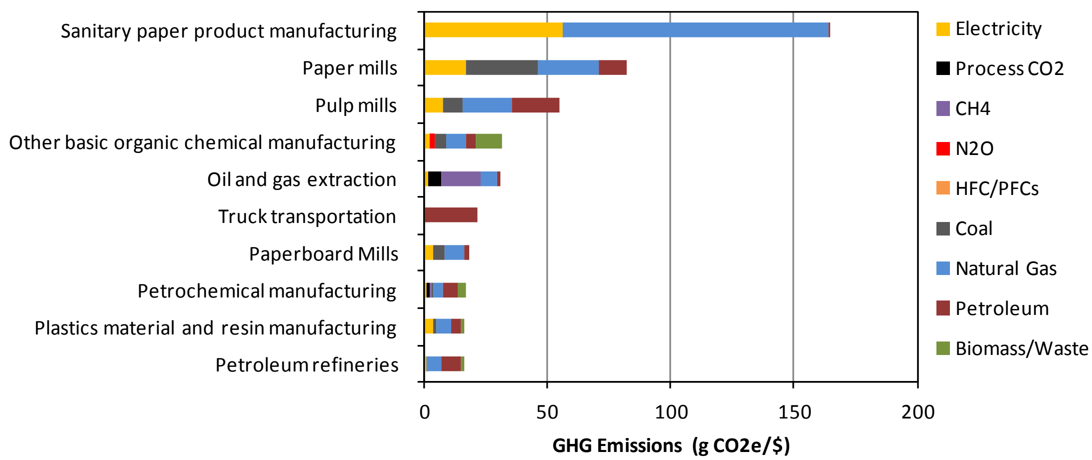
One kilogram of paper towels

Life-cycle system description

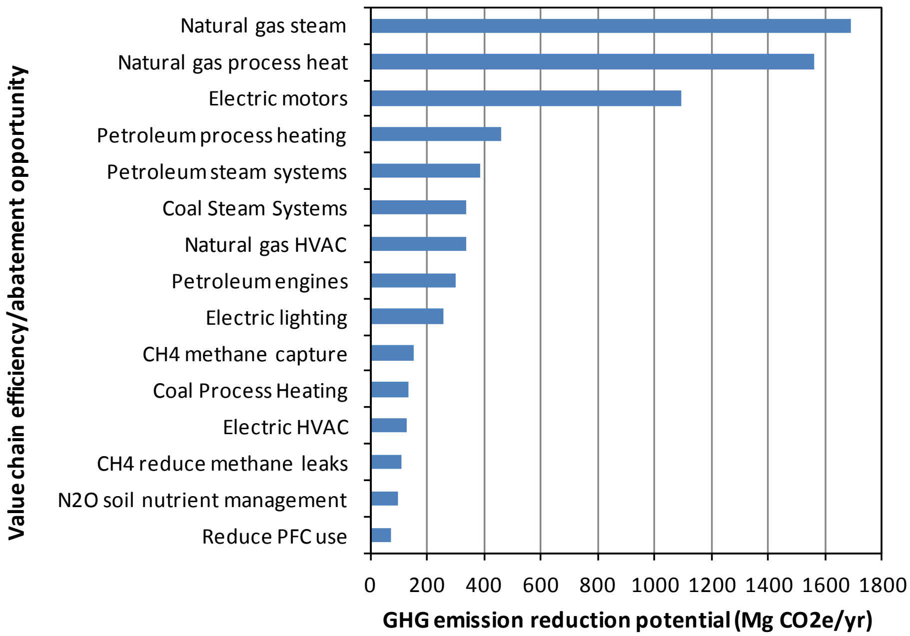
First, trees are harvested, sawn into logs, and shipped by truck to a pulp mill, or an integrated pulp and paper mill. The pulp and paper industry converts fibrous raw materials into pulp, paper, and paperboard products. Pulp mills manufacture only pulp, which is then sold and transported to paper and paperboard mills. A paper and paperboard mill may purchase pulp or manufacture its own pulp in house; in the latter case, such mills are referred to as integrated mills. The major processes employed in the pulp and paper industry include raw materials preparation (log debarking and wood chipping), pulping (chemical, semi-chemical, mechanical, and waste paper), bleaching, chemical recovery, pulp drying, and paper making. Bleached and chemically pulped processes require significant chemical inputs, and all pulp and paper mills are significant consumers of thermal fuels to make steam (typically for pulping, bleaching, cooking, evaporation, and drying) (Kramer et al. 2009). Paper towel rolls are packaged in plastic film, and shipped to the retailer. Depending on the consumer, after use a paper towel can be composted or sent to landfill.



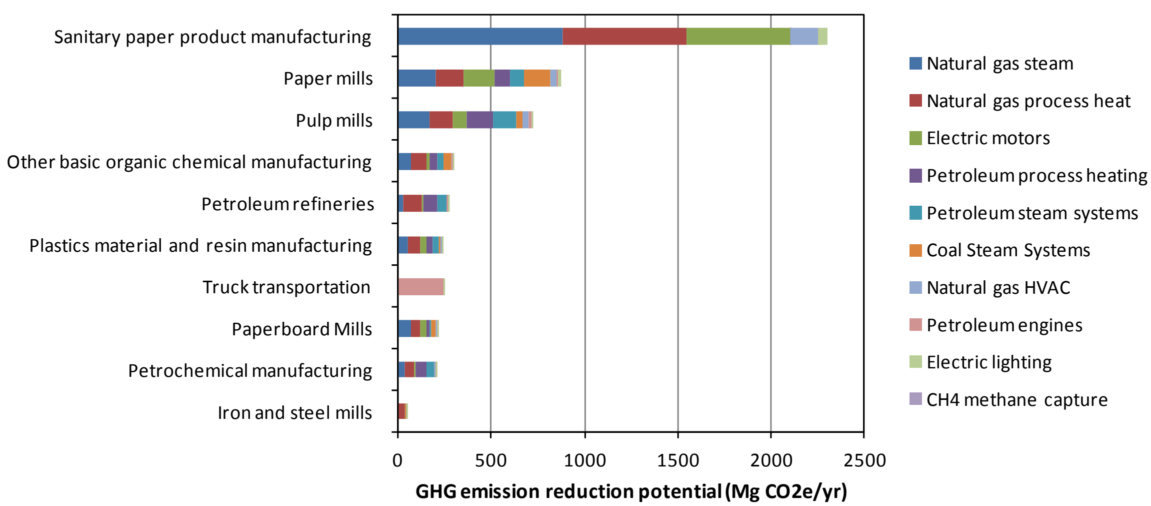
Top 10 contributing sectors to the energy footprint of product manufacture (MJ/$)



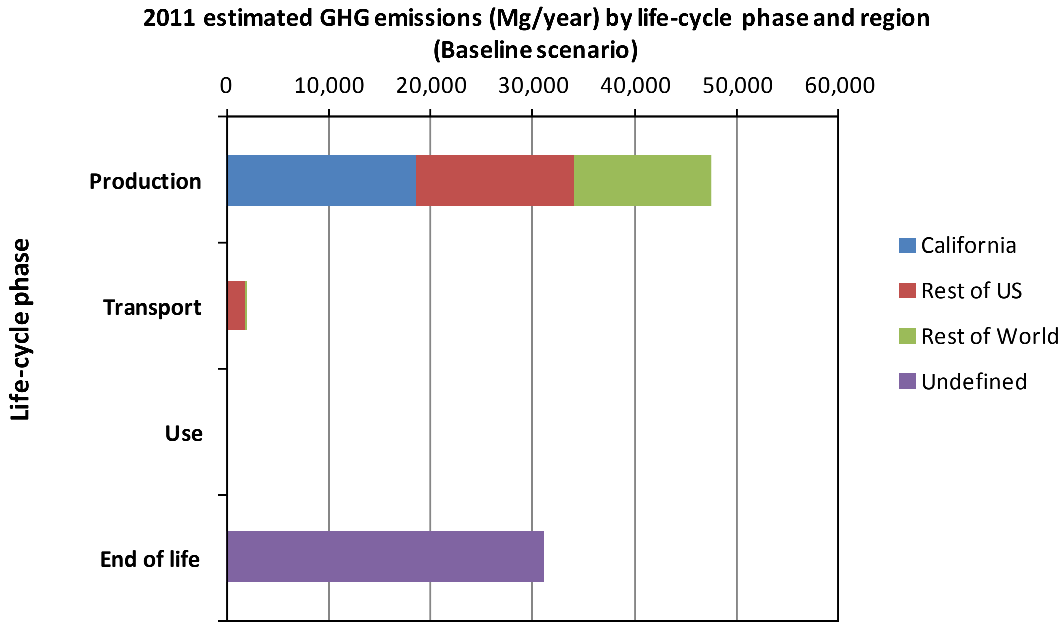
Top 10 contributing sectors to the GHG emissions footprint of product manufacture (MJ/$)



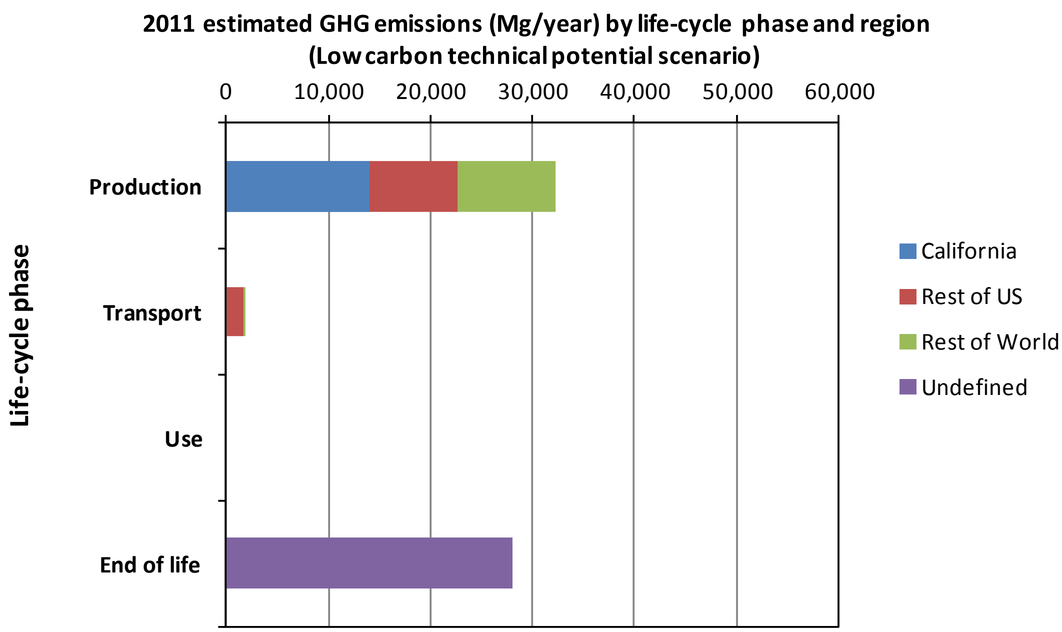
Estimated supply chain efficiency and GHG mitigation reductions by improvement opportunity and emissions source



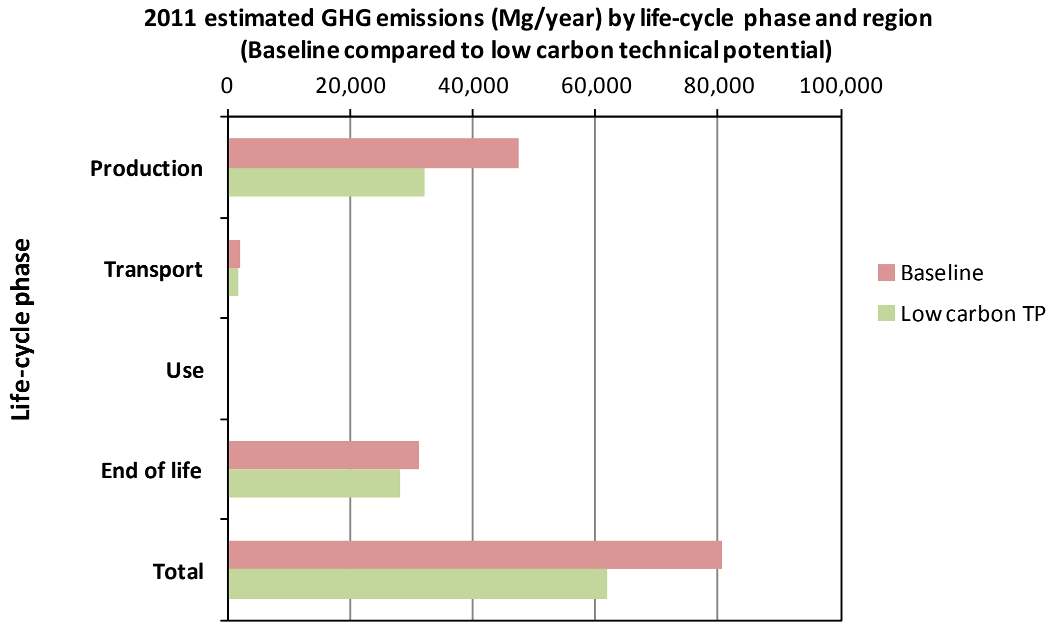
Estimated supply chain efficiency and GHG mitigation reductions by sector and emissions source



Estimated 2011 GHG emissions by life-cycle phase and region in the baseline scenario



2011 GHG emissions by life-cycle phase and region in the low carbon technical potential scenario



Side by side comparison of baseline and low carbon technical potential scenarios

Total savings = 23%

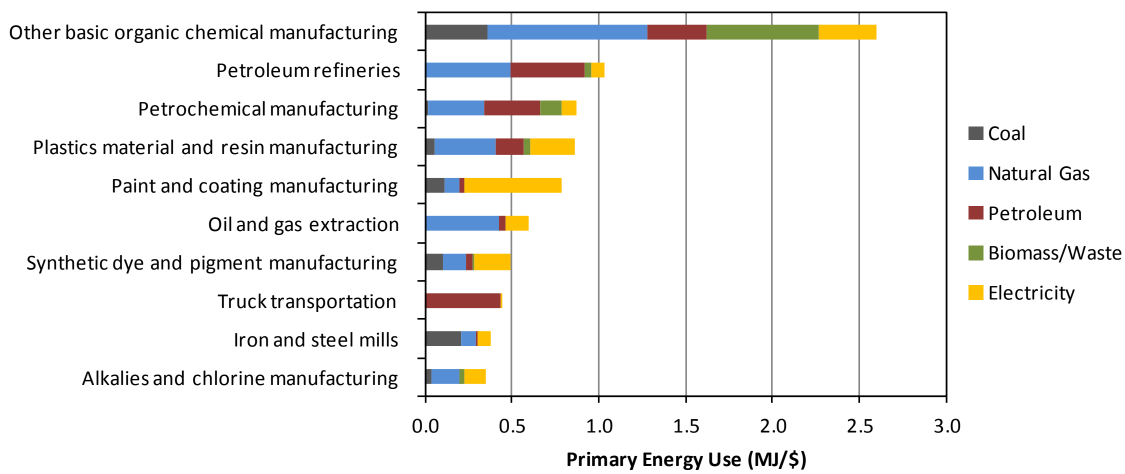
14: Paint

Product

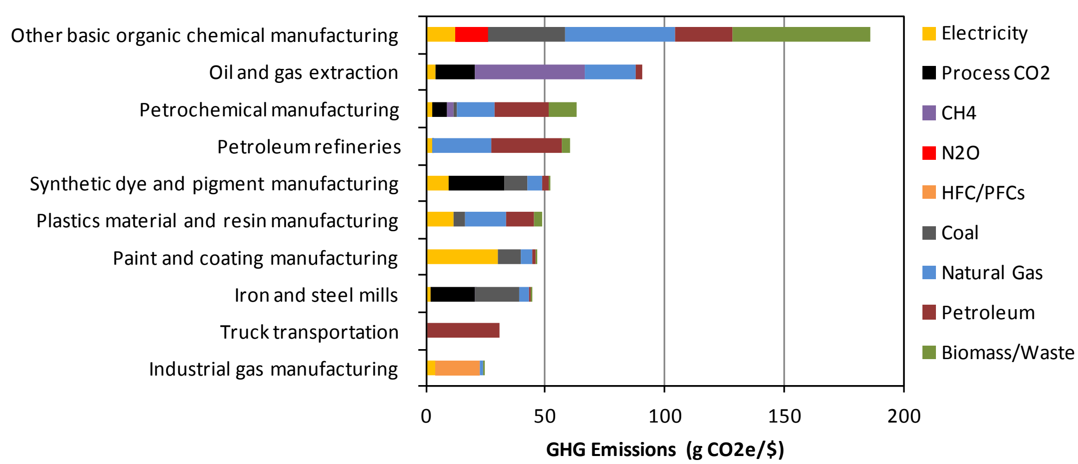
One gallon of paint in a steel container

Life-cycle system description

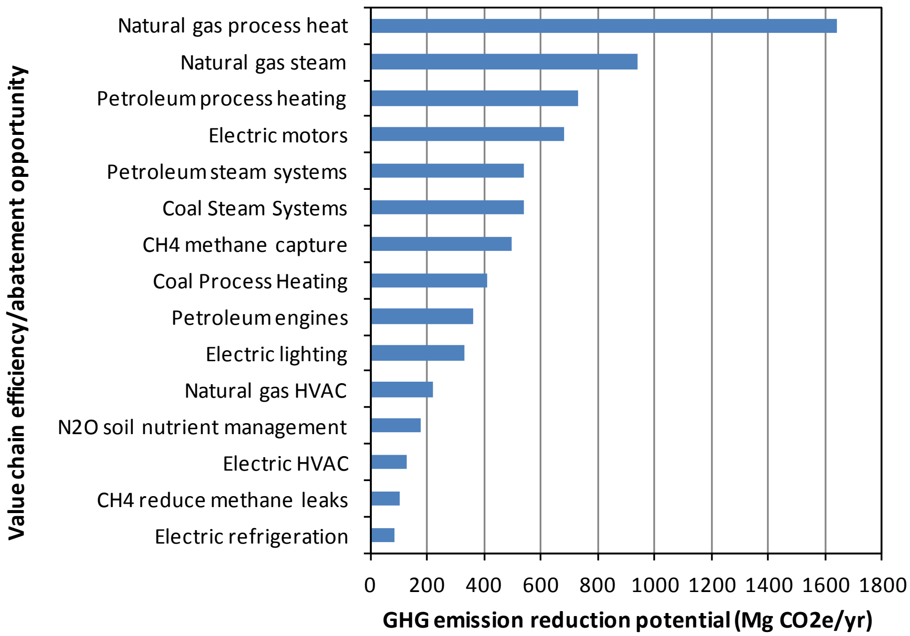
The primary ingredients in paints are pigments, binders, and solvents. Pigments provide the color to the paint, and can come from a range of different sources, including minerals, metals, and chemicals. Binding agents are made of natural or synthetic resins, and can be based on mineral or chemical feedstocks. The purpose of solvent is to provide viscosity to the paint for its application, and to aid in the curing process. Oil-based paints contain oil-based solvents, while water-based paints primarily use water for paint viscosity. Paints are manufactured from the petrochemicals manufacturing industry, which is an energy-intensive sector in the United States (Neelis et al. 2008). Most often paints are packaged in steel container, which can vary in size but the most common of which is the one gallon paint can. After use, the can is recyclable but its ultimate disposition path depends on the consumer.



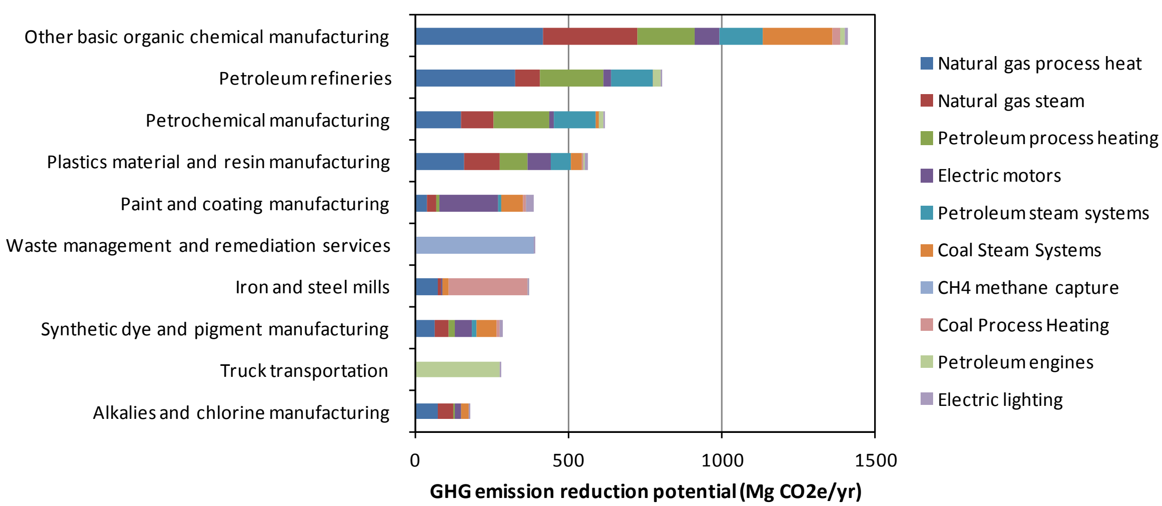
Top 10 contributing sectors to the energy footprint of product manufacture (MJ/$)



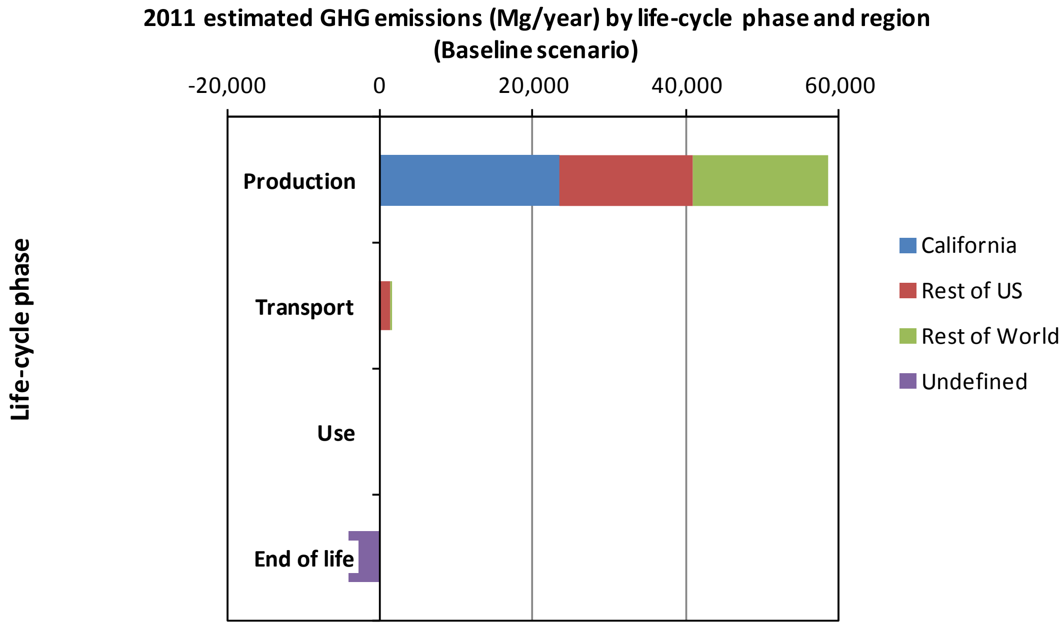
Top 10 contributing sectors to the GHG emissions footprint of product manufacture (MJ/$)



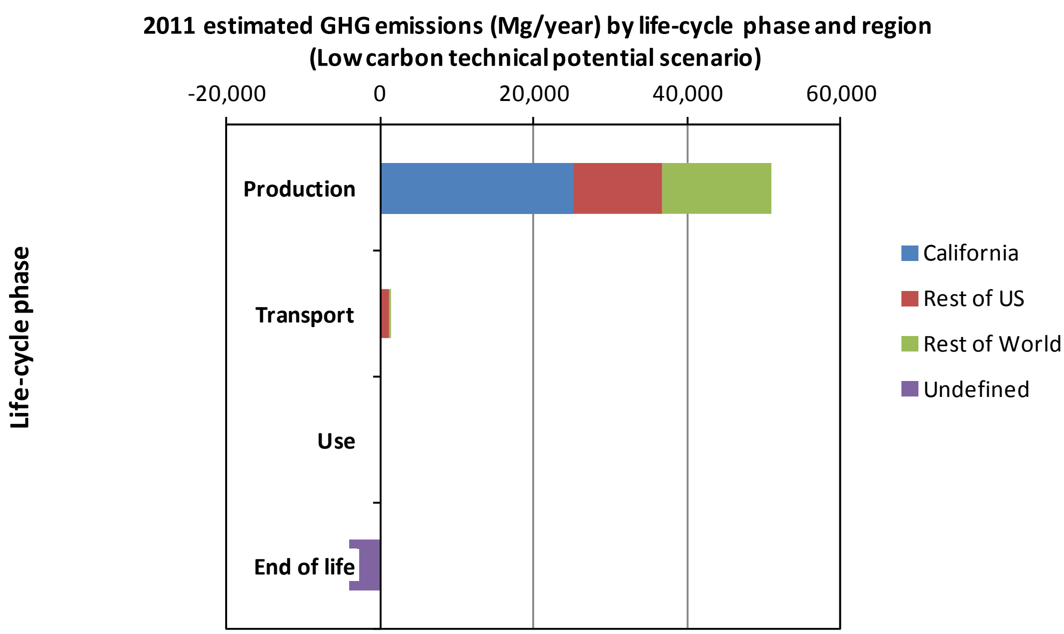
Estimated supply chain efficiency and GHG mitigation reductions by improvement opportunity and emissions source



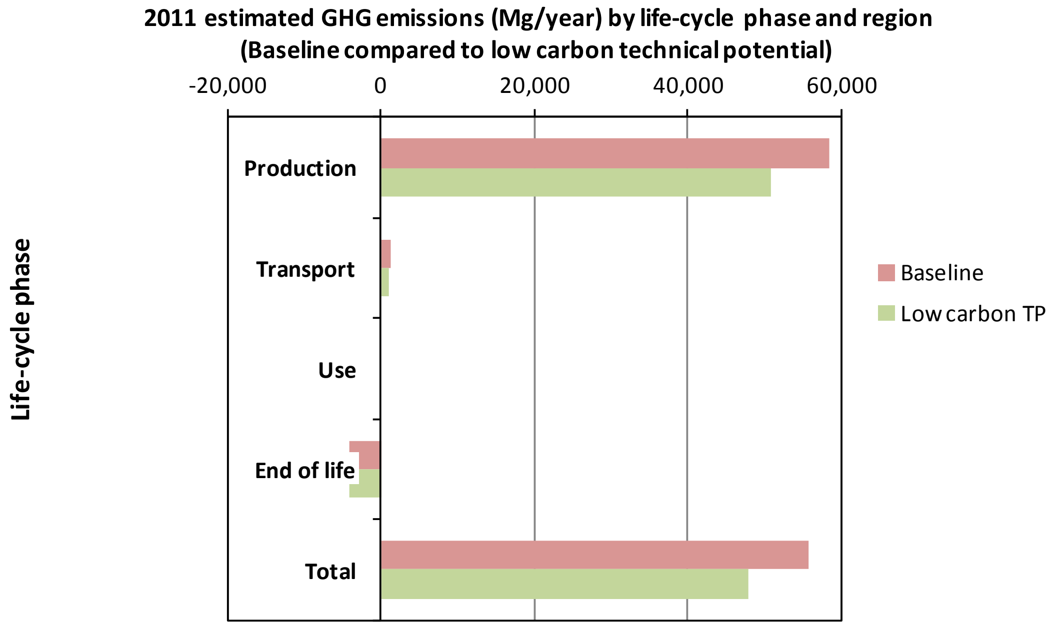
Estimated supply chain efficiency and GHG mitigation reductions by sector and emissions source



Estimated 2011 GHG emissions by life-cycle phase and region in the baseline scenario



2011 GHG emissions by life-cycle phase and region in the low carbon technical potential scenario



Side by side comparison of baseline and low carbon technical potential scenarios

Total savings = 14%

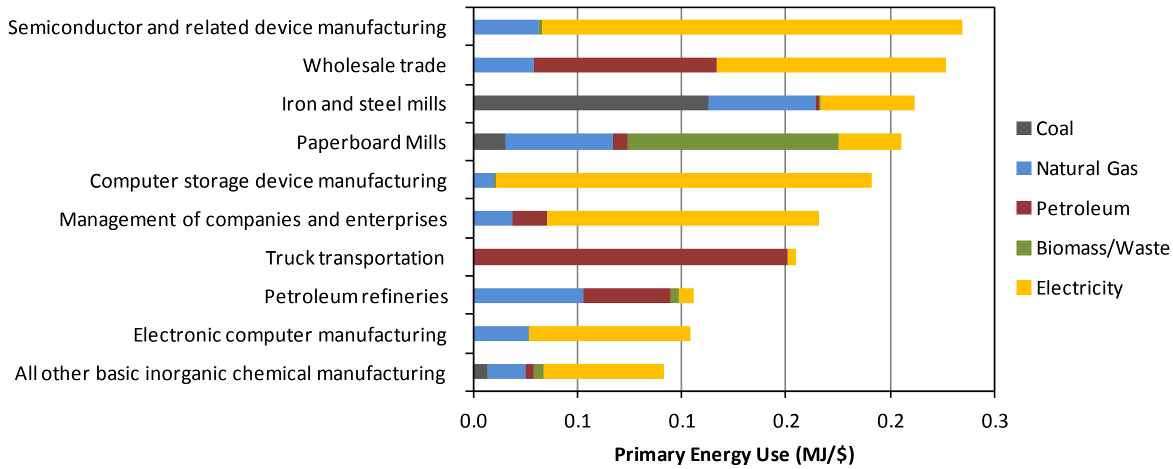
15: Personal computer

Product

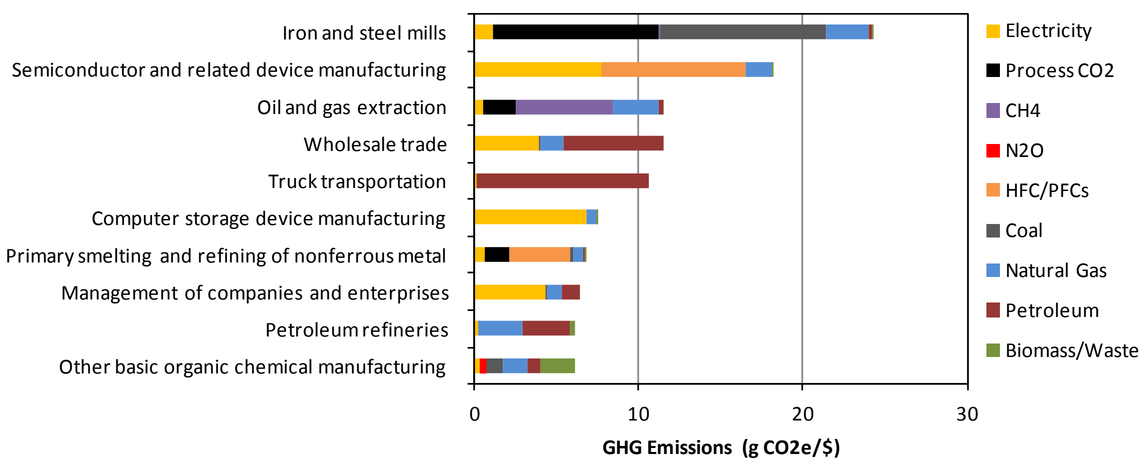
One desktop control unit

Life-cycle system description

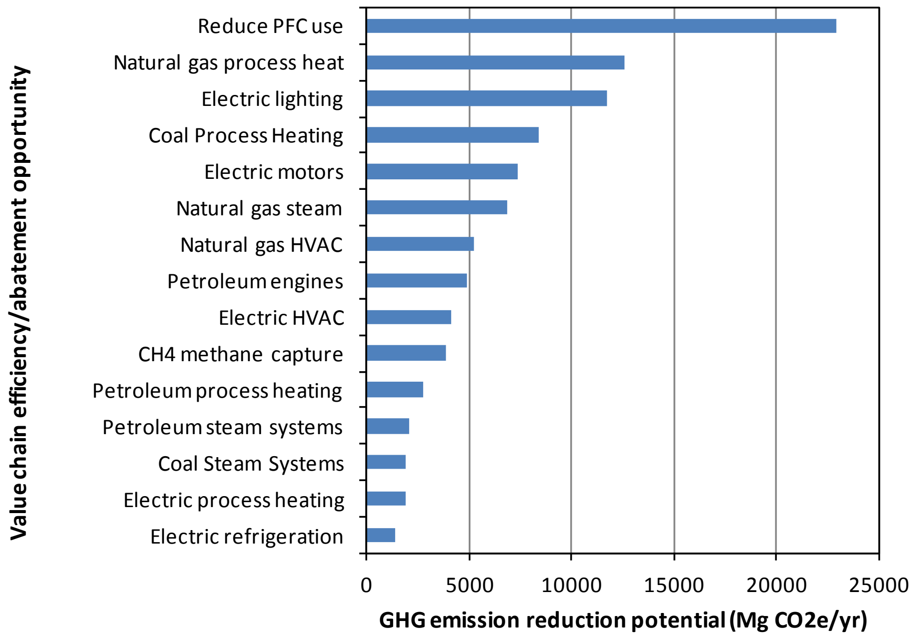
The control unit is the heart of a desktop PC, which containing the central processing unit, hard disk drive (HDD), memory modules, power supply, and auxiliary drives (e.g., floppy drives, CD-ROM drives, etc.). The control unit is typically housed in a chassis made of steel, aluminum, and/or plastics (Masanet et al. 2005). Most of the main components are made by dedicated supply chains; the components are then shipped to a computer assembly facility that assembled the final product, packages it, and ships it to the retail outlet. There are dozens of components and materials in a control unit, but the most energy and GHG emissions intensive components to manufacture are the semiconductor chips and printed circuit boards (Masanet et al. 2005). The desktop control unit consumes electricity throughout its useful life, and the amount of electricity consumed depends highly on the use patterns of the consumer. Under California's e-waste recycling laws, desktop control units must be discarded through an approved electronics recycler. Depending on recycling practices, the control unit can either be shredded to recover its metals content or manually disassembled for component and materials recovery, either domestically or overseas.



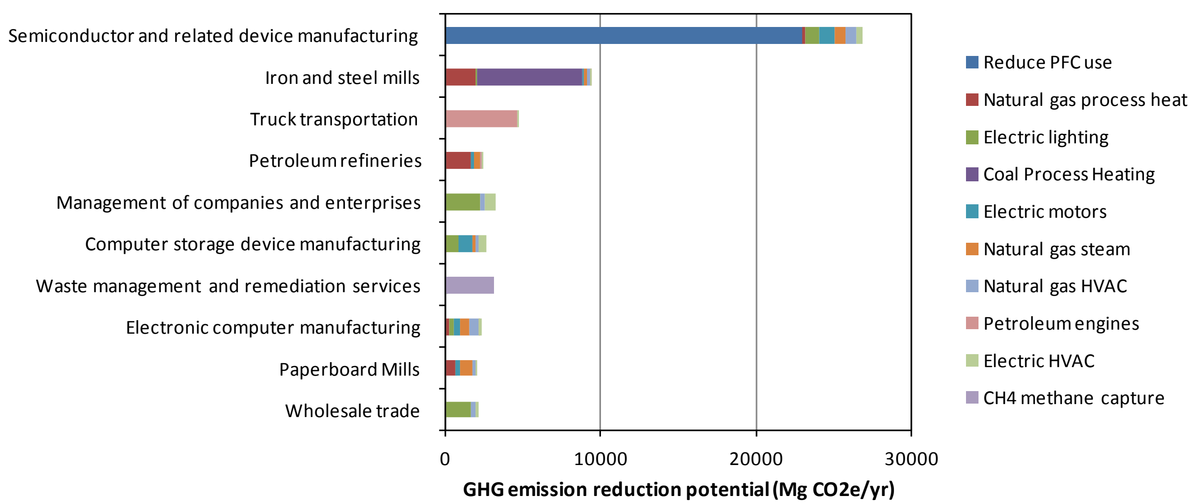
Top 10 contributing sectors to the energy footprint of product manufacture (MJ/$)



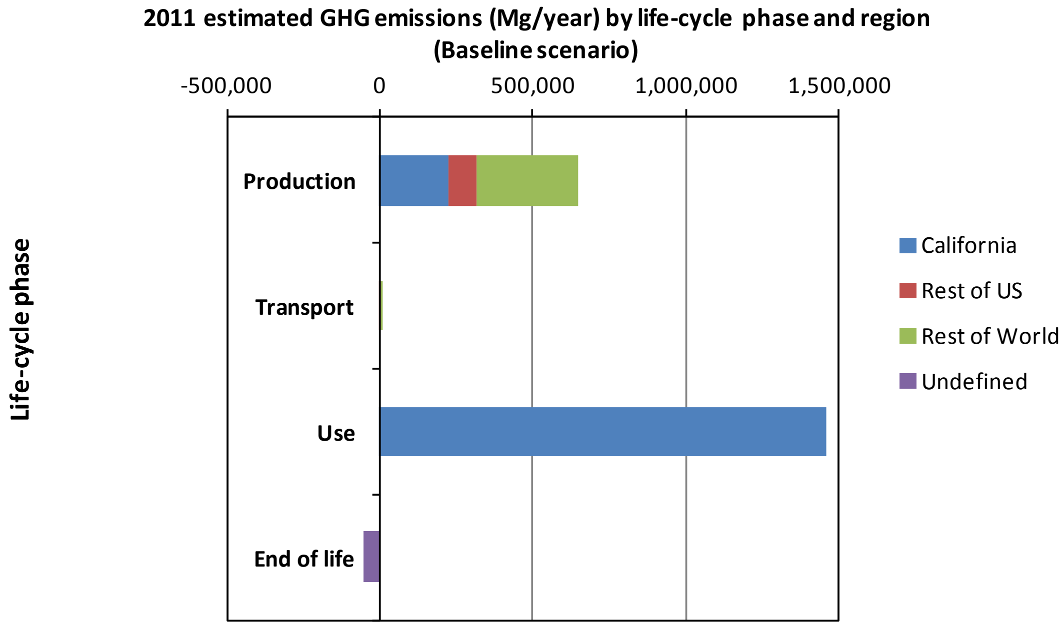
Top 10 contributing sectors to the GHG emissions footprint of product manufacture (MJ/$)



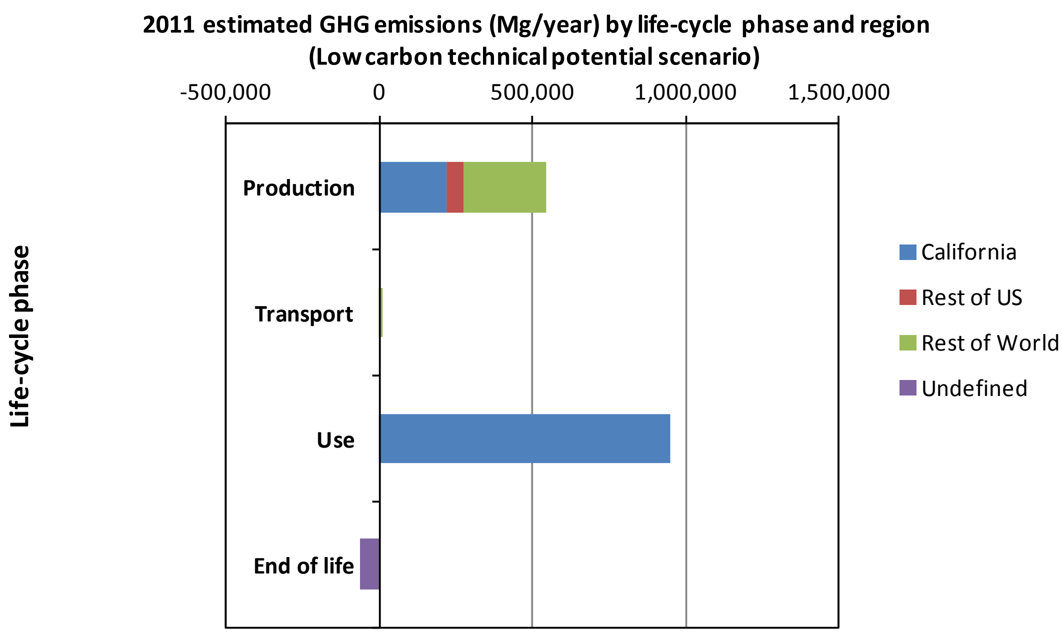
Estimated supply chain efficiency and GHG mitigation reductions by improvement opportunity and emissions source



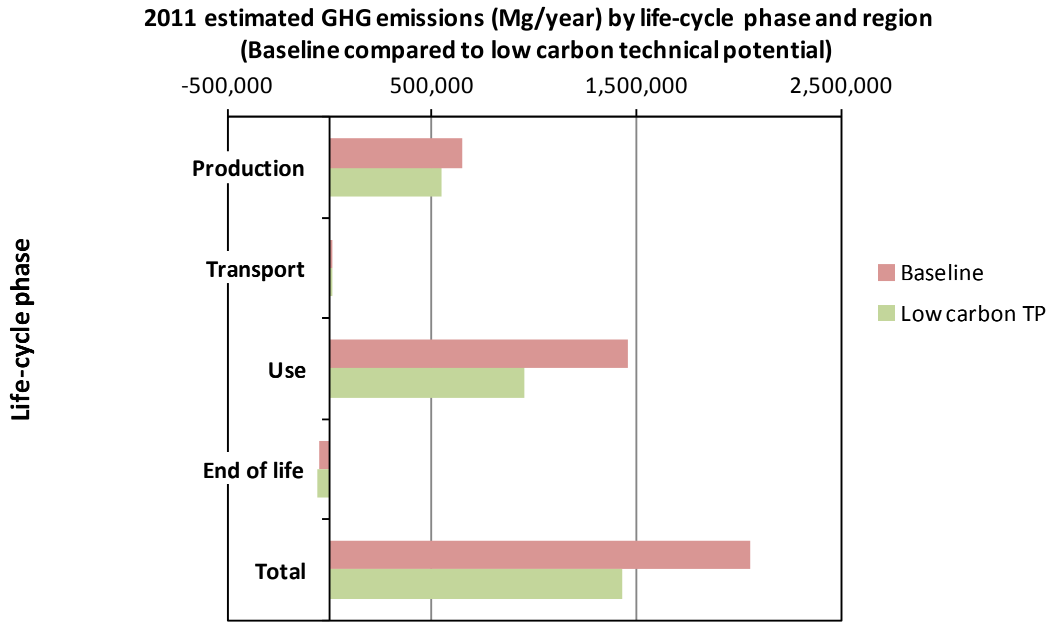
Estimated supply chain efficiency and GHG mitigation reductions by sector and emissions source



Estimated 2011 GHG emissions by life-cycle phase and region in the baseline scenario



2011 GHG emissions by life-cycle phase and region in the low carbon technical potential scenario



Side by side comparison of baseline and low carbon technical potential scenarios

Total savings = 30%

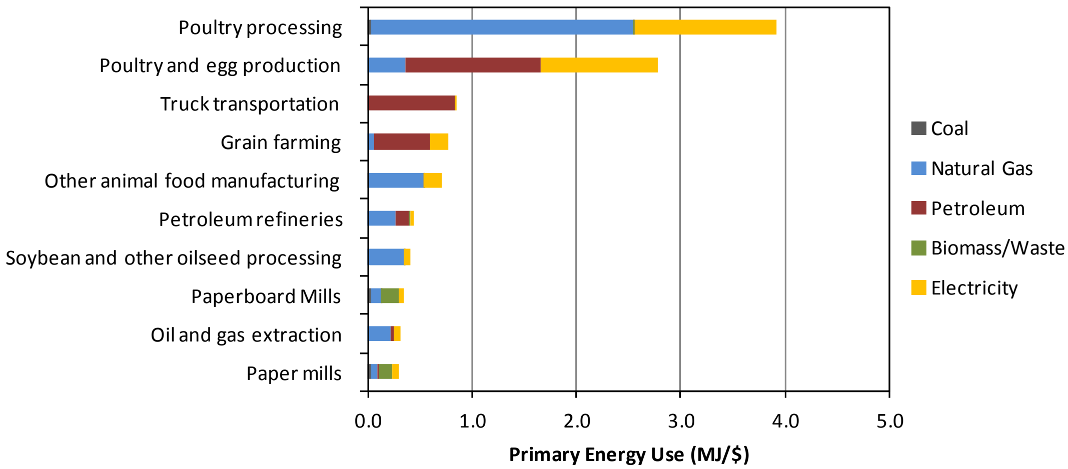
16: Poultry

Product

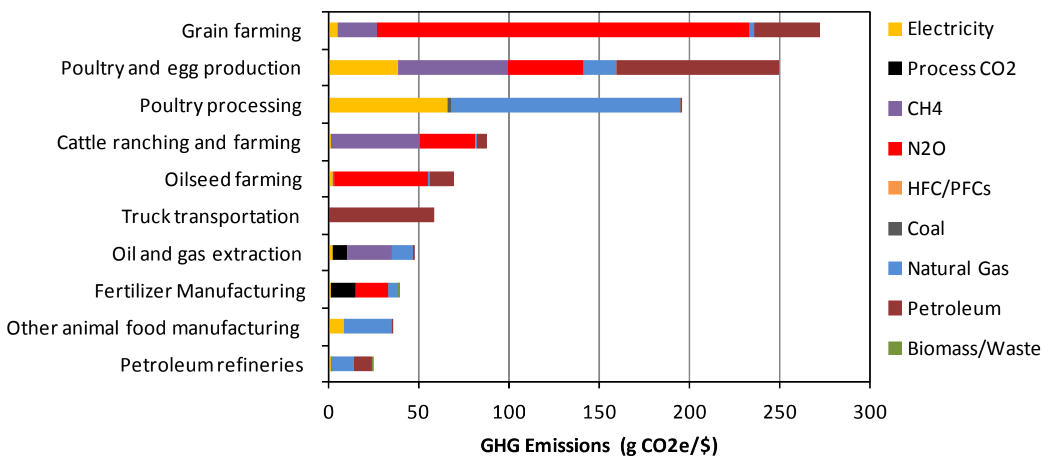
One kilogram of packaged chicken

Life-cycle system description

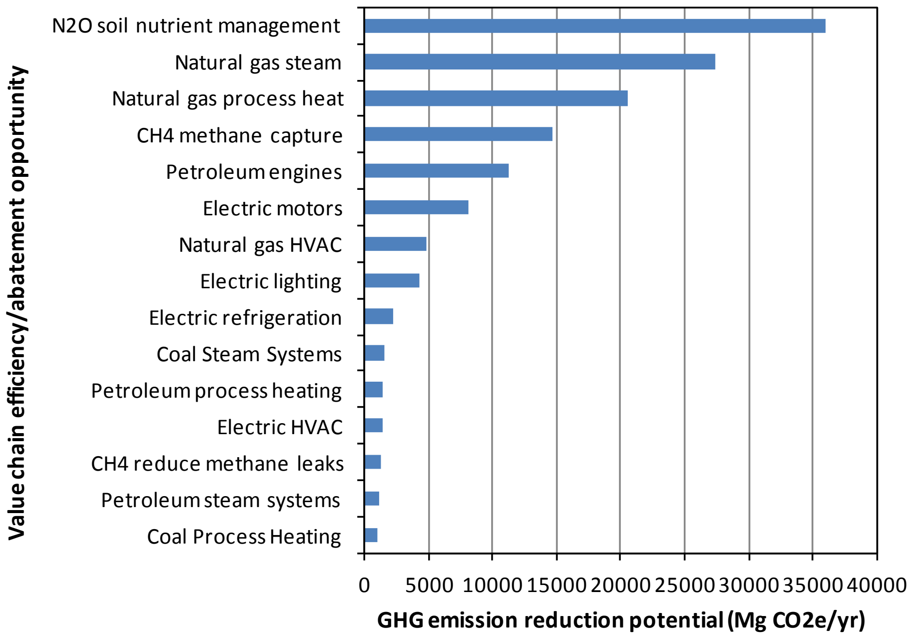
Chickens are first raised on a poultry farm, where they born in hatcheries and are typically raised on grain-based chicken feed. Depending on the farm, they can be raised in large houses (which are lighted and ventilated) or in outdoor fashion. Poultry farms are significant generators of manure, which can be used as fertilizer. In the processing plant, chickens are killed, scalded, plucked, and processed into different end use products such as whole chickens and chicken parts. They are then packaged in plastic and refrigerated prior to shipment to the retailer. After consumption, the plastic packaging is sent to landfill at end of life. Depending on the consumer, the product may be fully eaten or partially disposed as waste, which can either be landfilled or composted.



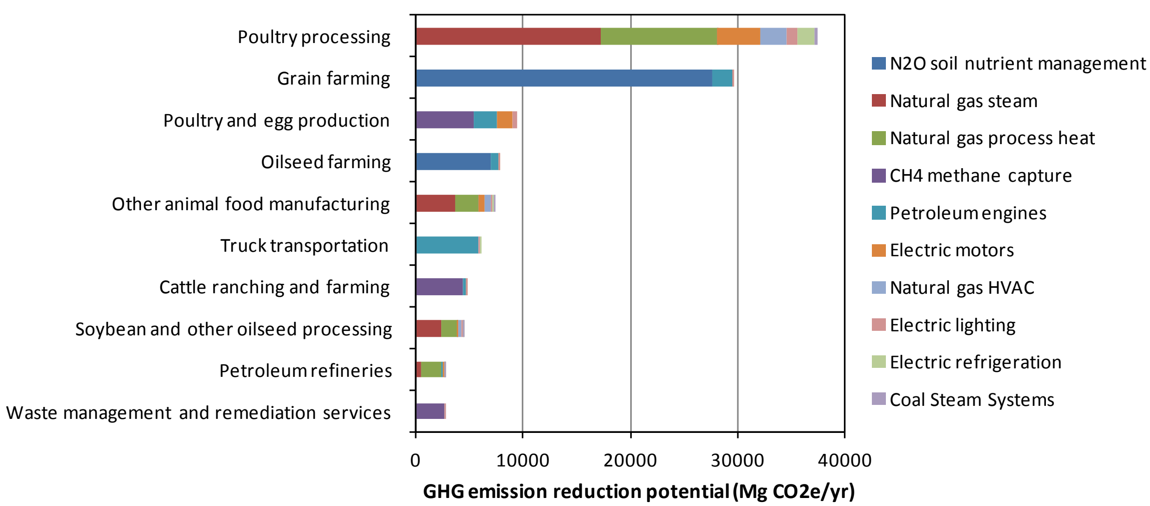
Top 10 contributing sectors to the energy footprint of product manufacture (MJ/$)



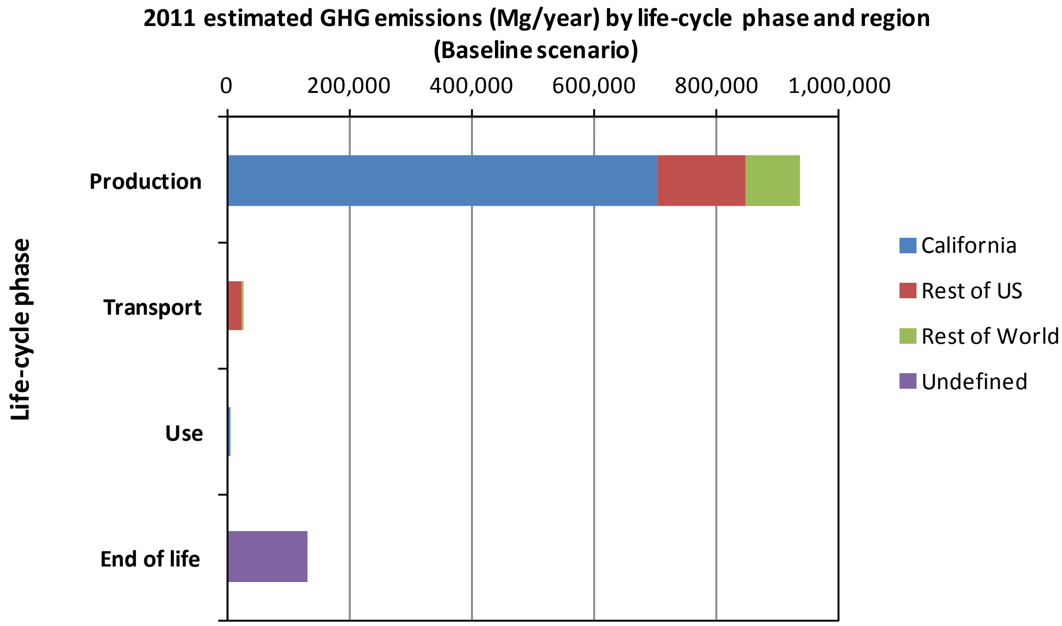
Top 10 contributing sectors to the GHG emissions footprint of product manufacture (MJ/$)



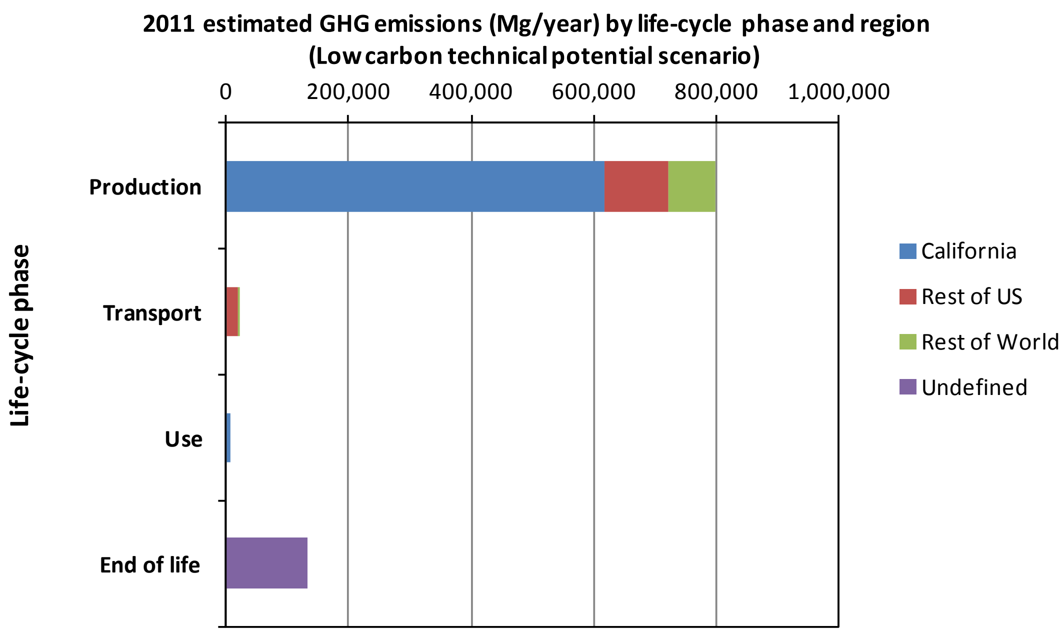
Estimated supply chain efficiency and GHG mitigation reductions by improvement opportunity and emissions source



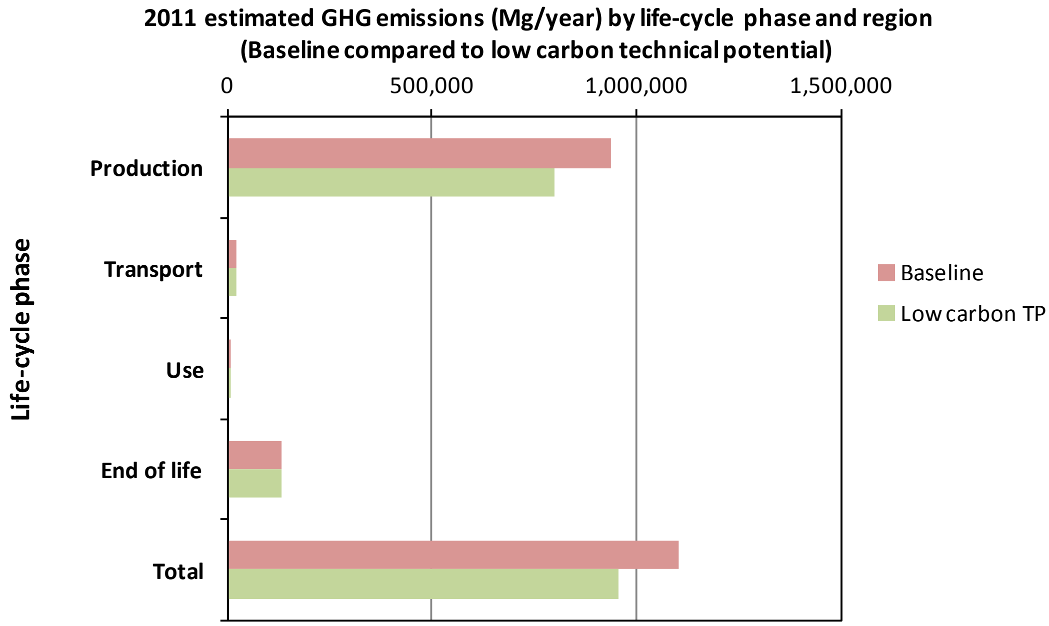
Estimated supply chain efficiency and GHG mitigation reductions by sector and emissions source



Estimated 2011 GHG emissions by life-cycle phase and region in the baseline scenario



2011 GHG emissions by life-cycle phase and region in the low carbon technical potential scenario



Side by side comparison of baseline and low carbon technical potential scenarios

Total savings = 13%

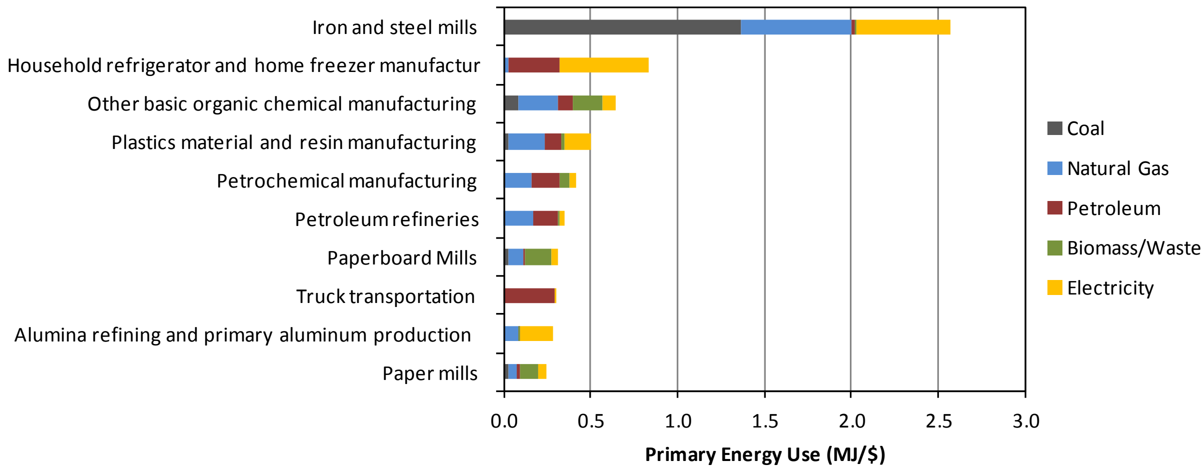
17: Refrigerator

Product

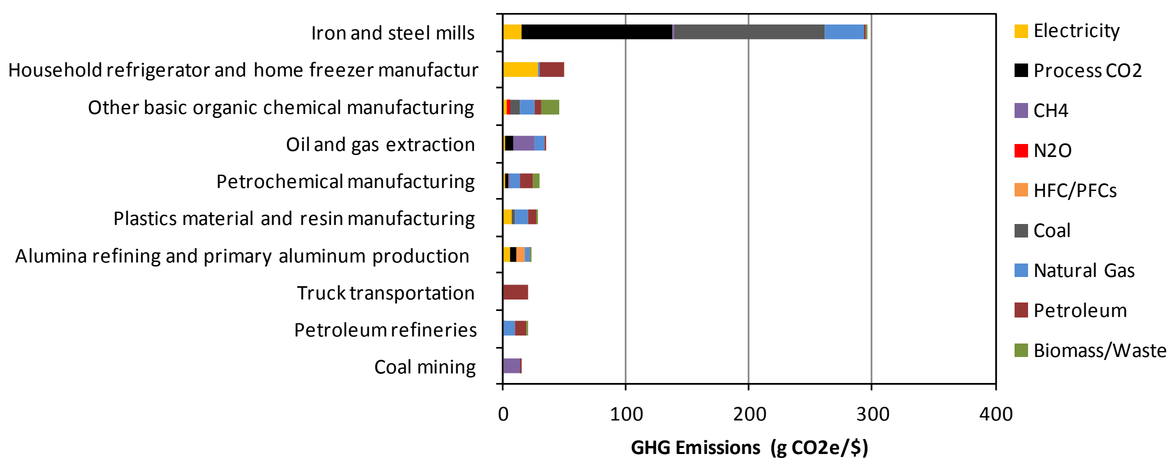
One 22 cubic foot refrigerator

Life-cycle system description

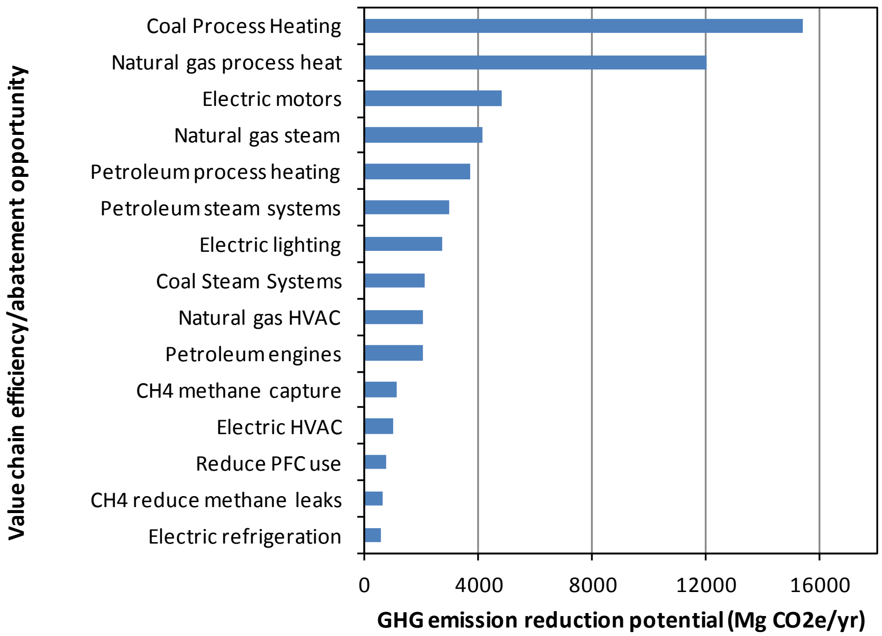
The primary materials in a refrigerator are steel, which is used for the refrigerators body and refrigeration compressor unit, and plastics, which are used for the internal chambers as well as for insulation materials. However, there are a number of other materials contained in a refrigerator, including glass, copper, and aluminum (ISIS 2007). Refrigerator components are made at multiple suppliers, and shipped to a final assembly facility that assembles, tests, and packages the finished units for shipment to the retailer. The vast majority of a refrigerator's life-cycle energy use and GHG emissions occurs during the use phase. Most primary refrigerators will be used for 10-15 years (U.S. EPA 2011), after which time they will either be discarded or used as secondary refrigerators in a garage or basement (KEMA 2010). In California, discarded refrigerators cannot be sent to landfill.



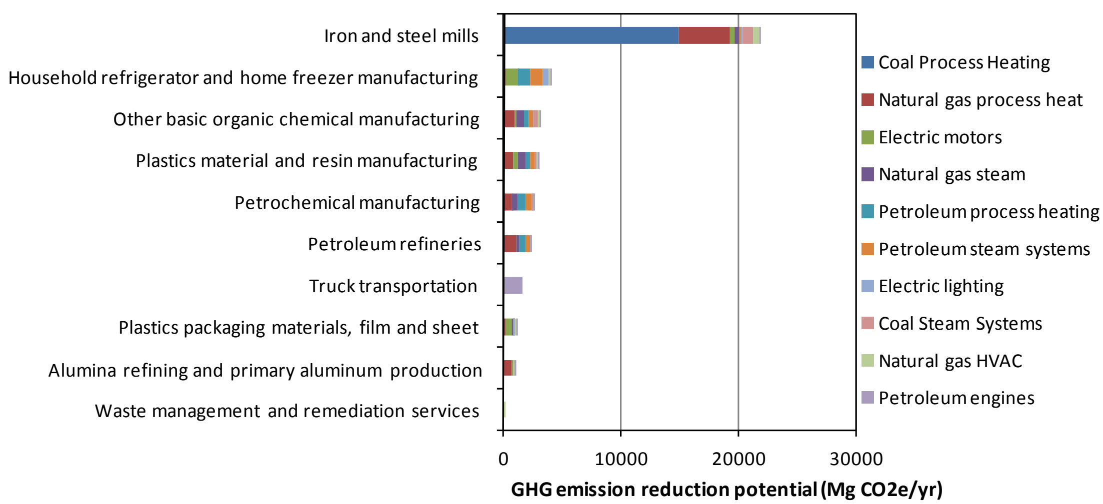
Top 10 contributing sectors to the energy footprint of product manufacture (MJ/$)



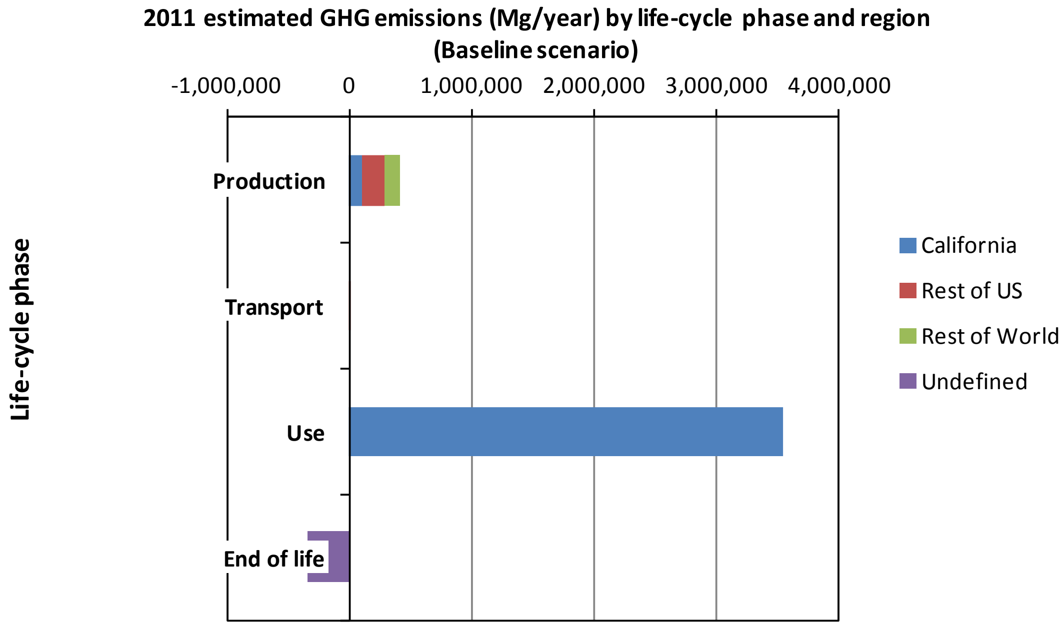
Top 10 contributing sectors to the GHG emissions footprint of product manufacture (MJ/$)



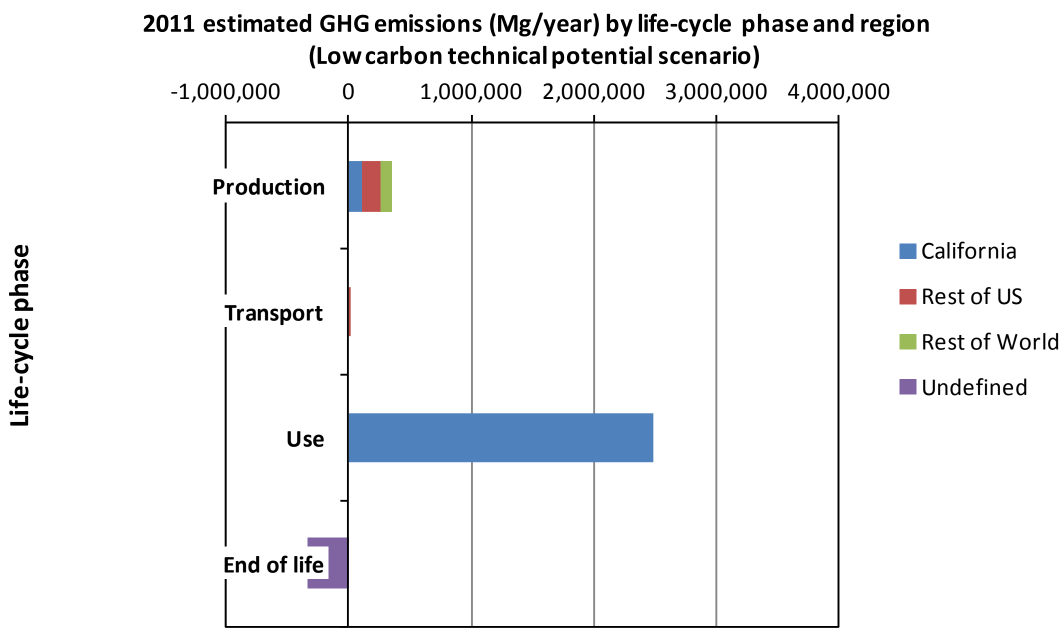
Estimated supply chain efficiency and GHG mitigation reductions by improvement opportunity and emissions source



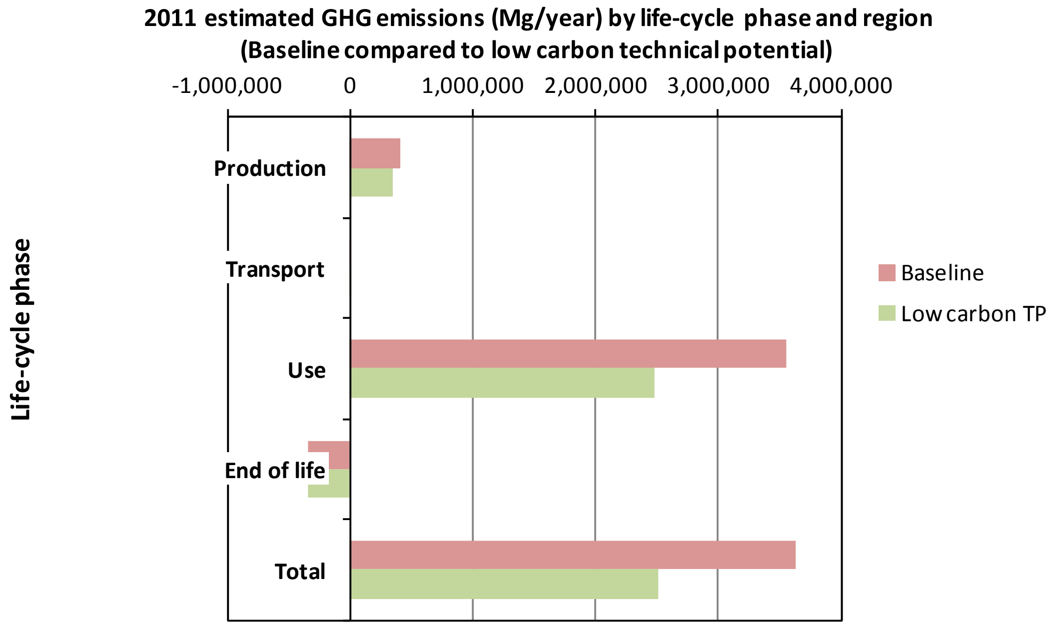
Estimated supply chain efficiency and GHG mitigation reductions by sector and emissions source



Estimated 2011 GHG emissions by life-cycle phase and region in the baseline scenario



2011 GHG emissions by life-cycle phase and region in the low carbon technical potential scenario



Side by side comparison of baseline and low carbon technical potential scenarios

Total savings = 31%

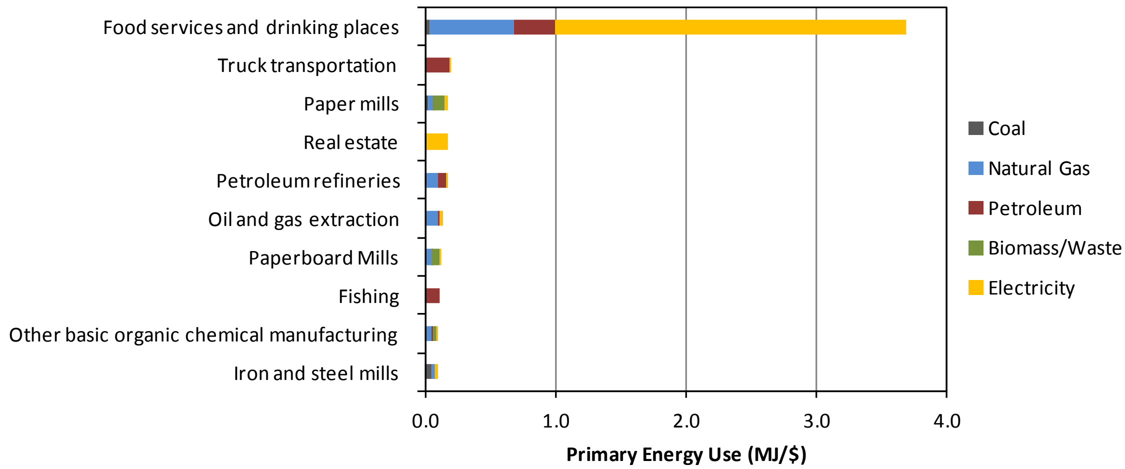
18: Restaurant

Product

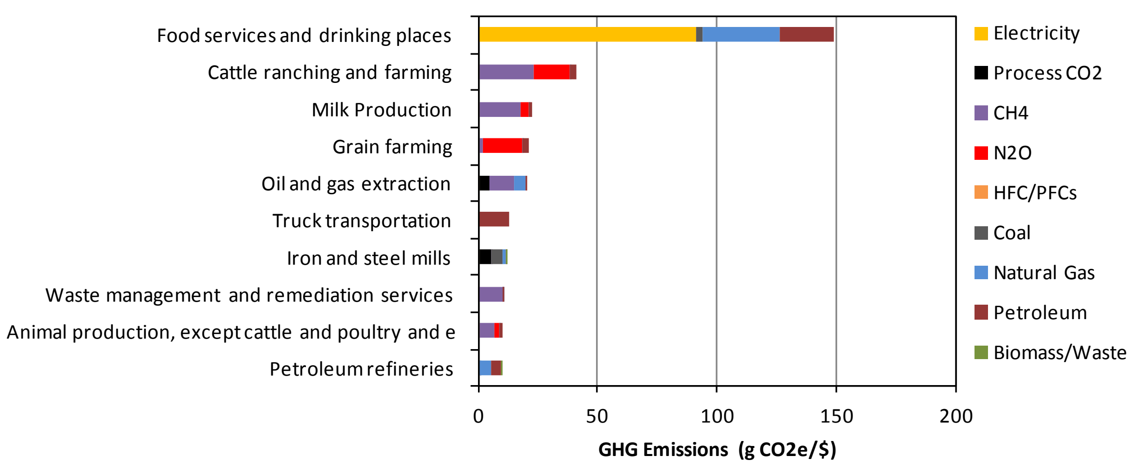
One dollar (in $2002 producer prices) of spending at a restaurant

Life-cycle system description

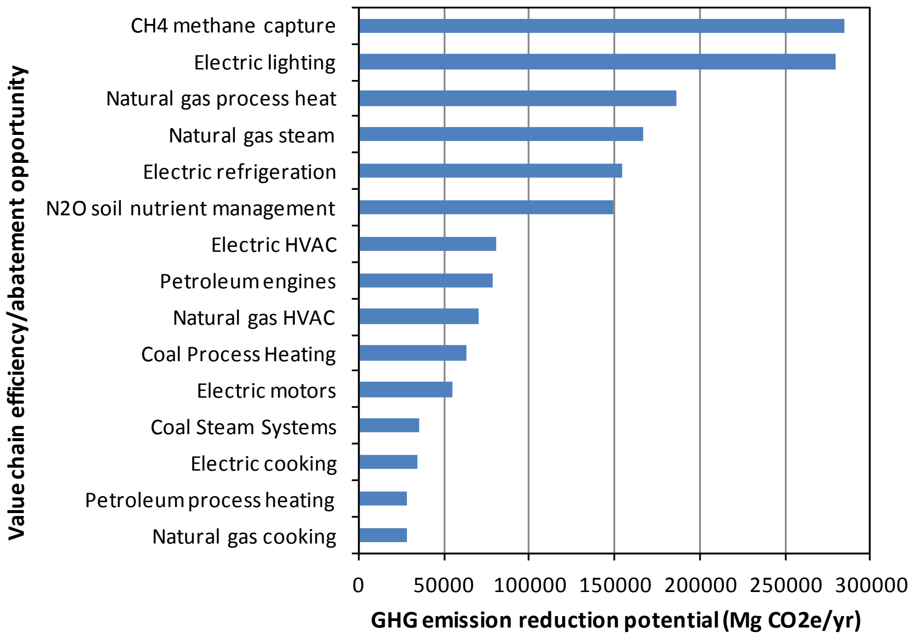
Restaurants are commercial entities, which can range in function (from snack bars to fast foods to sit-down dining) and vary widely in terms of their menus, purchases, and consumer prices. In California, the typical restaurant will consume most of its energy in water heating (primarily for washing), cooking, refrigeration, lighting, and space conditioning (Itron 2006). However, a significant portion of a restaurant's life-cycle GHG emissions occur in the production chain for the foods and beverages they purchase, prepare, and sell. Restaurants can also generate significant quantities of food waste and wastewater.



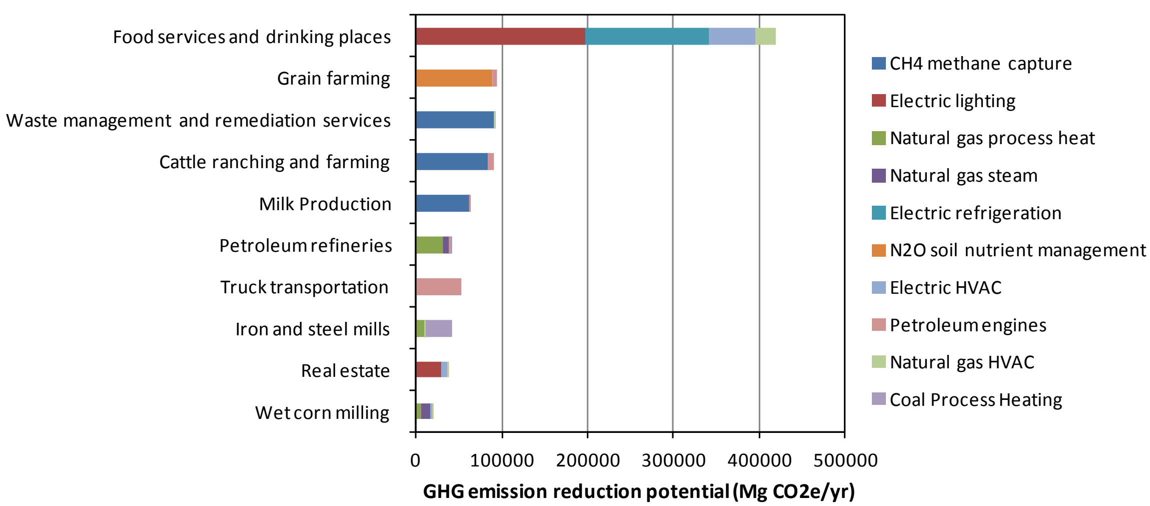
Top 10 contributing sectors to the energy footprint of product manufacture (MJ/$)



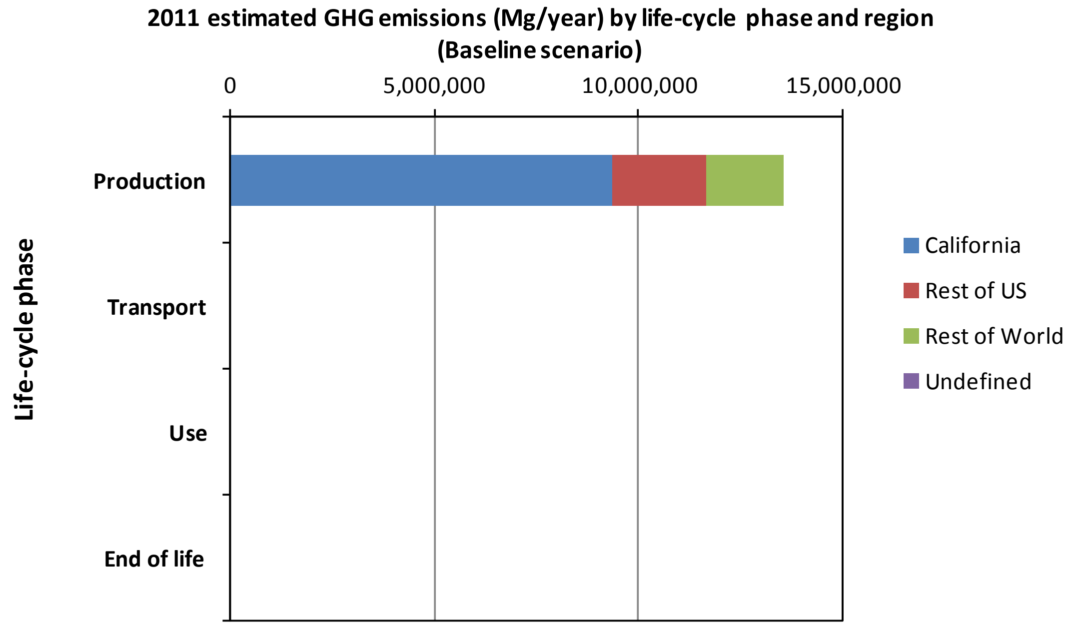
Top 10 contributing sectors to the GHG emissions footprint of product manufacture (MJ/$)



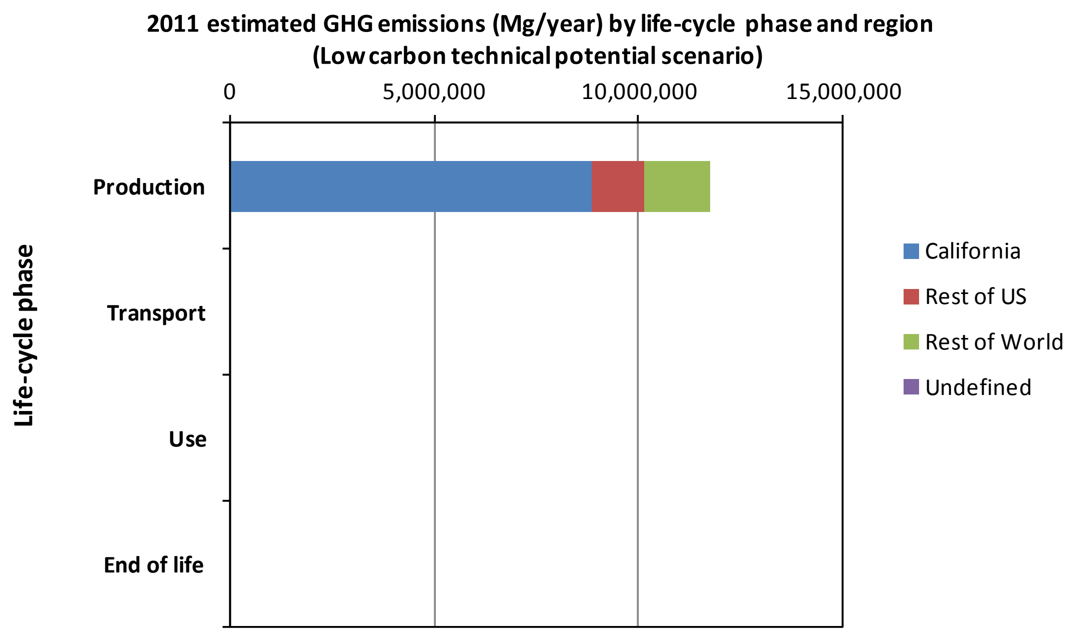
Estimated supply chain efficiency and GHG mitigation reductions by improvement opportunity and emissions source



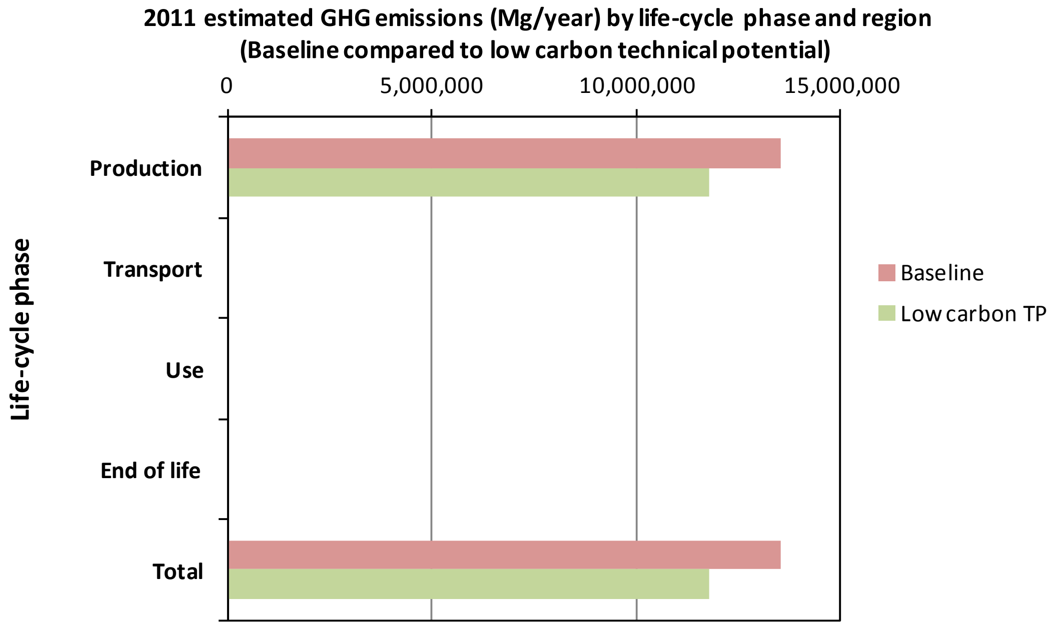
Estimated supply chain efficiency and GHG mitigation reductions by sector and emissions source



Estimated 2011 GHG emissions by life-cycle phase and region in the baseline scenario



2011 GHG emissions by life-cycle phase and region in the low carbon technical potential scenario



Side by side comparison of baseline and low carbon technical potential scenarios

Total savings = 13%

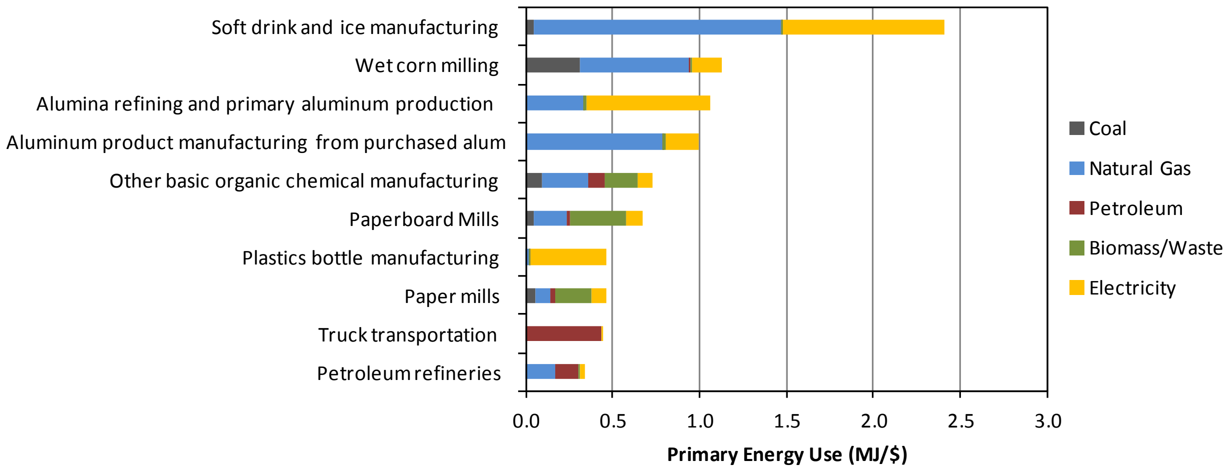
19: Soft drink

Product

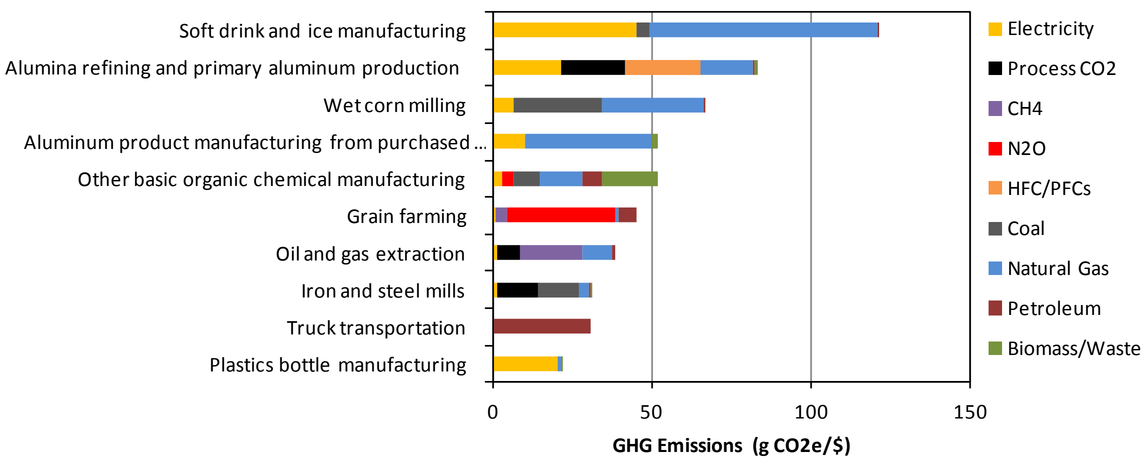
One 16 ounce plastic bottle of carbonated soft drink

Life-cycle system description

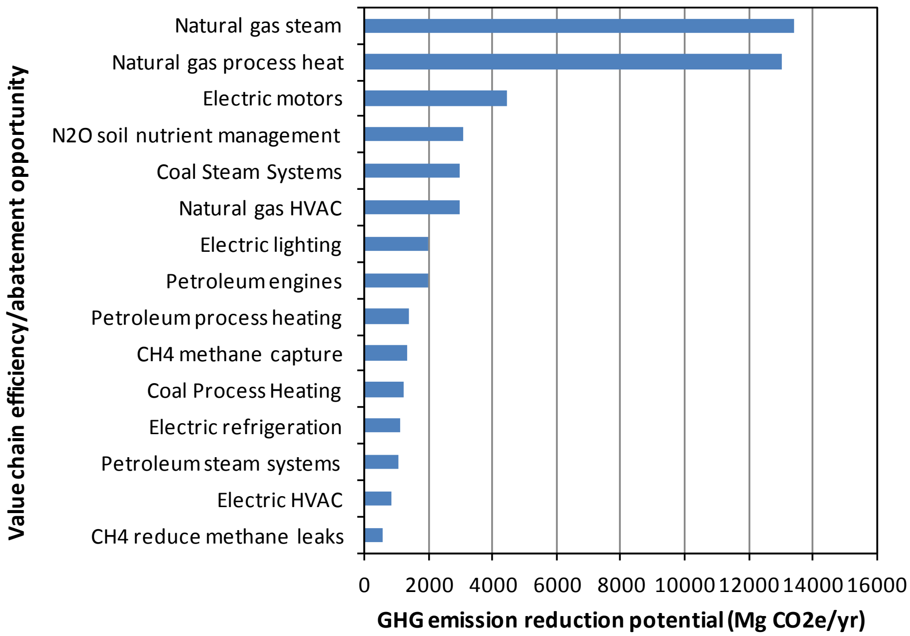
The primary ingredients in a carbonated soft drink are carbonated water, a sweetener (typically high fructose corn syrup or a non-caloric sweetener for diet drinks), and flavorings. High fructose corn syrup manufacture is an energy intensive process, which involves harvesting the corn, trucking it to a corn miller, processing at a corn miller (which includes energy and water intensive steeping and separations) to produce syrup, and shipping to the soft drink manufacturer. The soft drink manufacturer typically makes carbonated water onsite. Ingredients are mixed and added to the carbonated water, which is then bottled and packaged for final shipment to the retailer. Soft drink bottles are made of PET, which is a plastic resin that must be heated and molded into shape to create the bottle. Most soft drinks are refrigerated prior to consumption. In California, around 85% of PET bottles are currently recycled thanks to the California Redemption Value (CRV) fee (CalRecycle 2011).



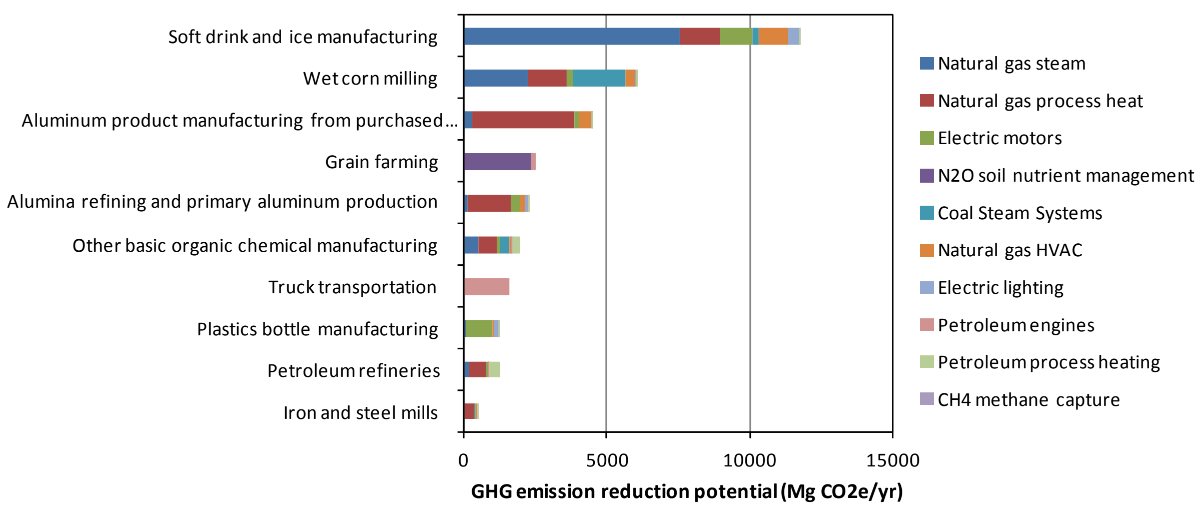
Top 10 contributing sectors to the energy footprint of product manufacture (MJ/$)



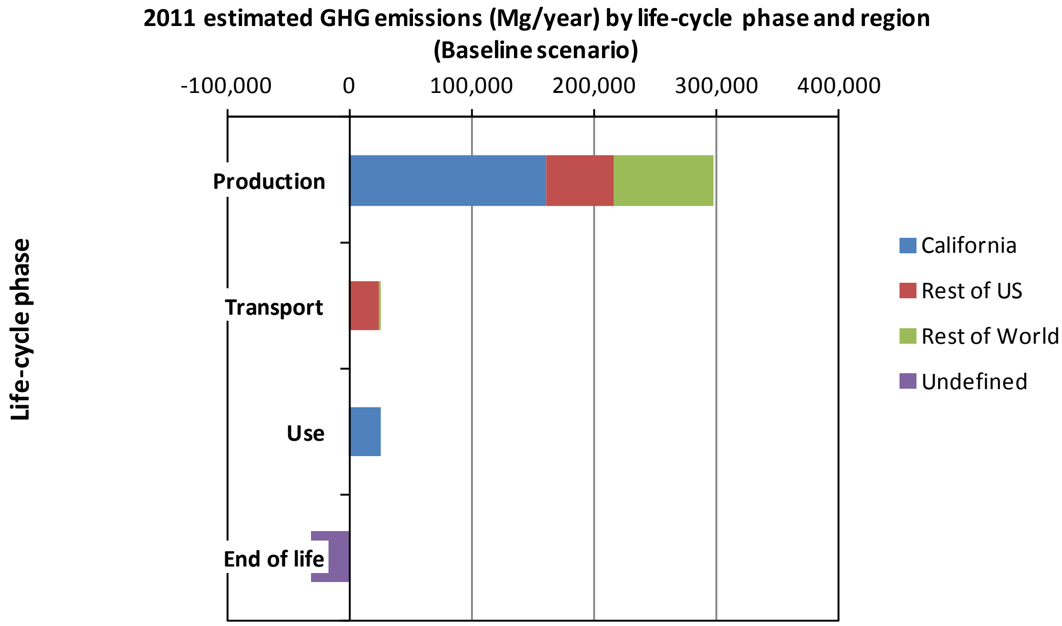
Top 10 contributing sectors to the GHG emissions footprint of product manufacture (MJ/$)



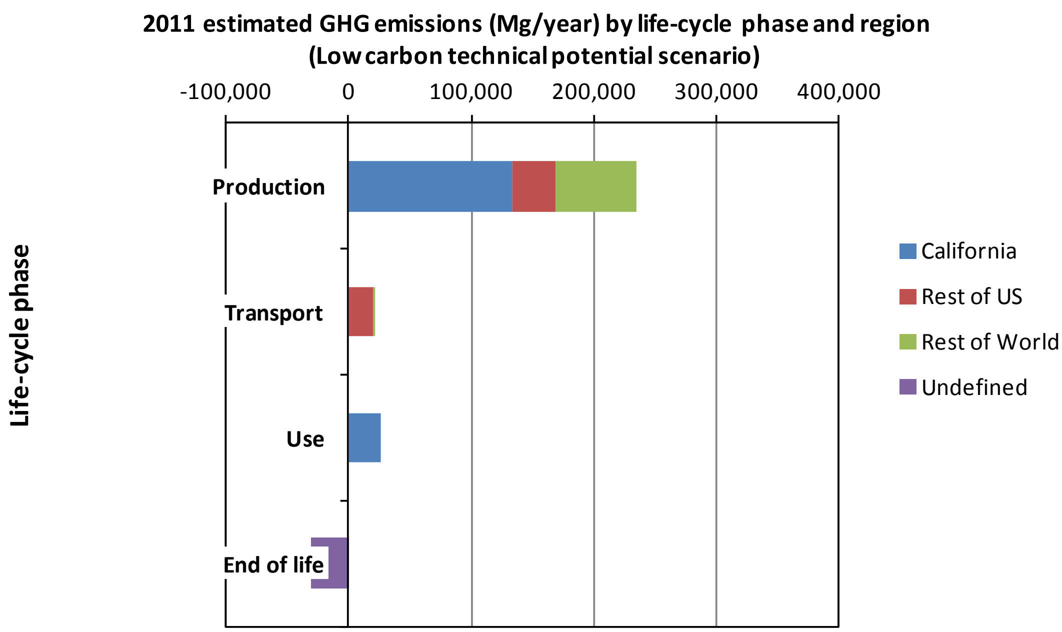
Estimated supply chain efficiency and GHG mitigation reductions by improvement opportunity and emissions source



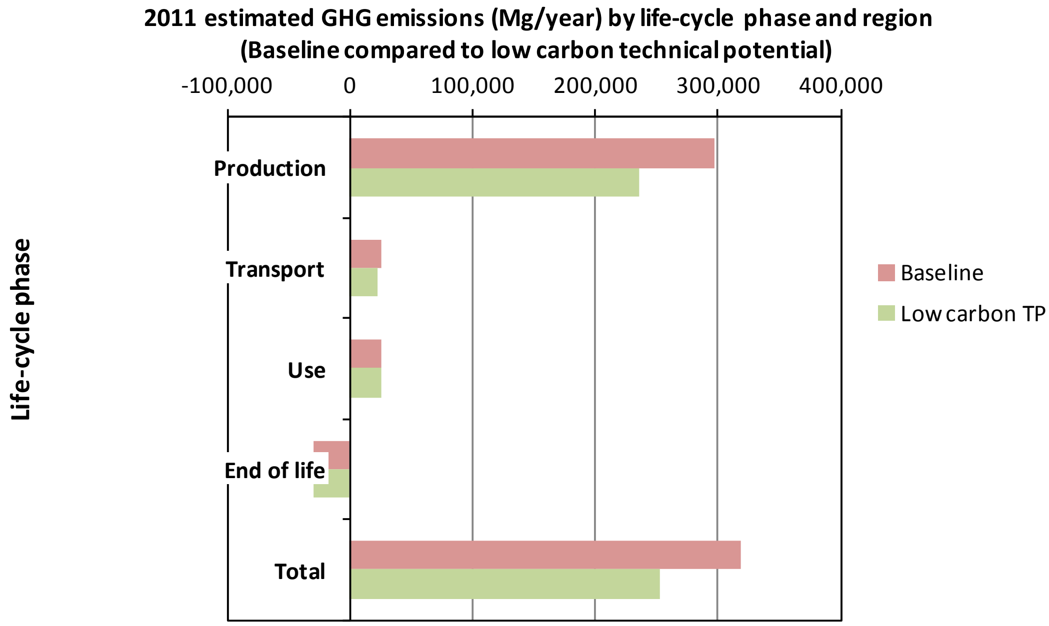
Estimated supply chain efficiency and GHG mitigation reductions by sector and emissions source



Estimated 2011 GHG emissions by life-cycle phase and region in the baseline scenario



2011 GHG emissions by life-cycle phase and region in the low carbon technical potential scenario



Side by side comparison of baseline and low carbon technical potential scenarios

Total savings = 21%

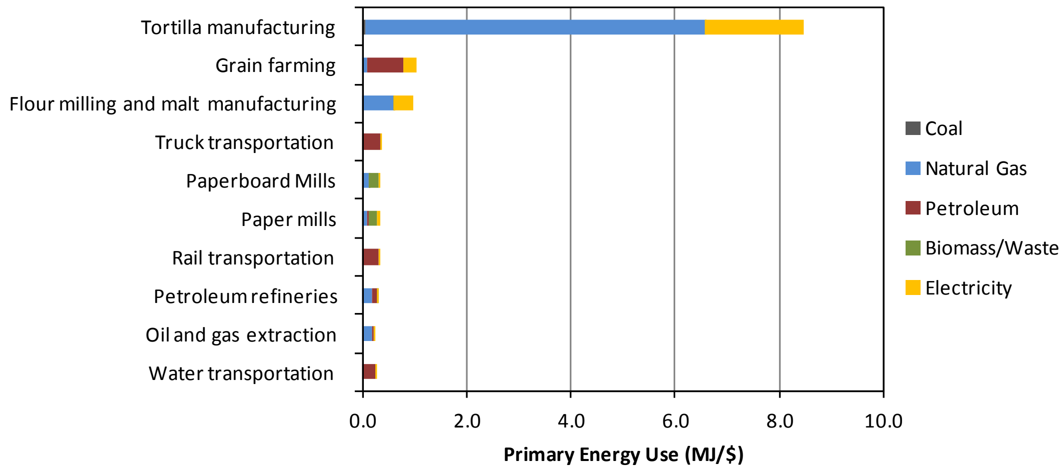
20: Tortillas

Product

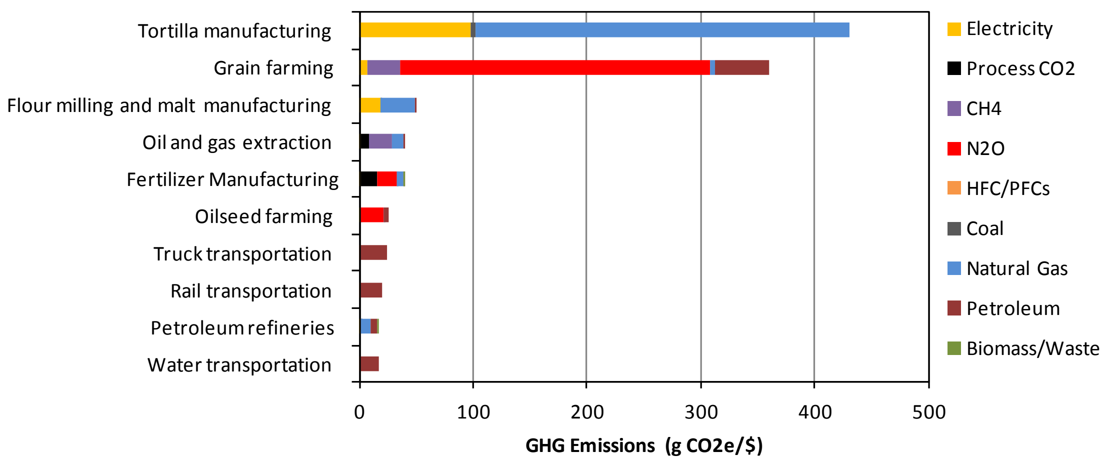
One kilogram of packaged tortillas

Life-cycle system description

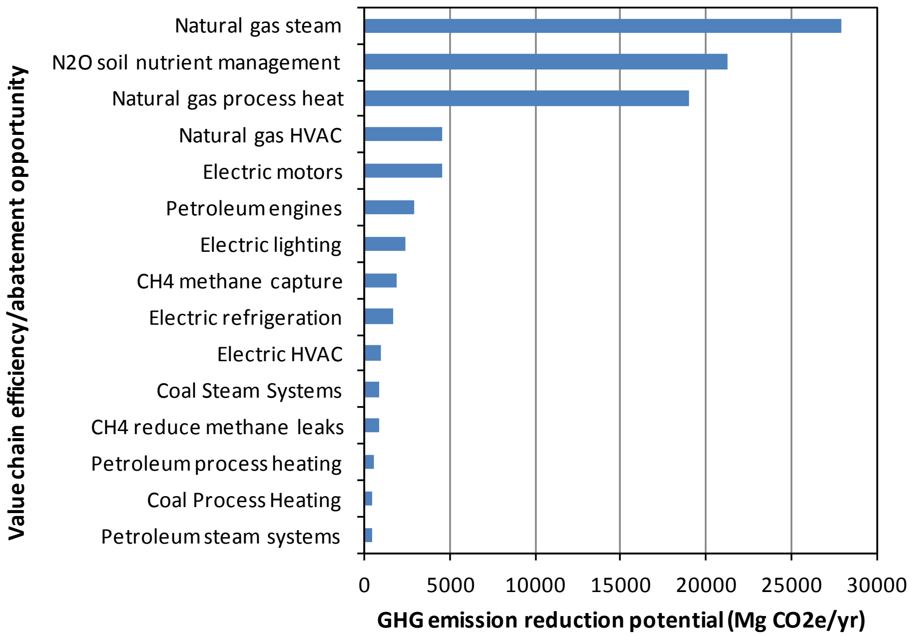
Tortillas can be made from either corn meal or wheat flour, both of which are produced first by crop harvesting and processing prior to shipment to the tortilla factory. There water, oil, and other ingredients are added to make a dough, which is fed into an automated press that creates the flat tortilla shape. The dough is then baked into a finished product; tortillas are compiled into small stacks which are typically packaged in plastic film bags and shipped to the retailer. The plastic packaging is sent to landfill at end of life. Depending on the consumer, the tortillas may be fully eaten or partially disposed as waste, which can either be landfilled or composted.



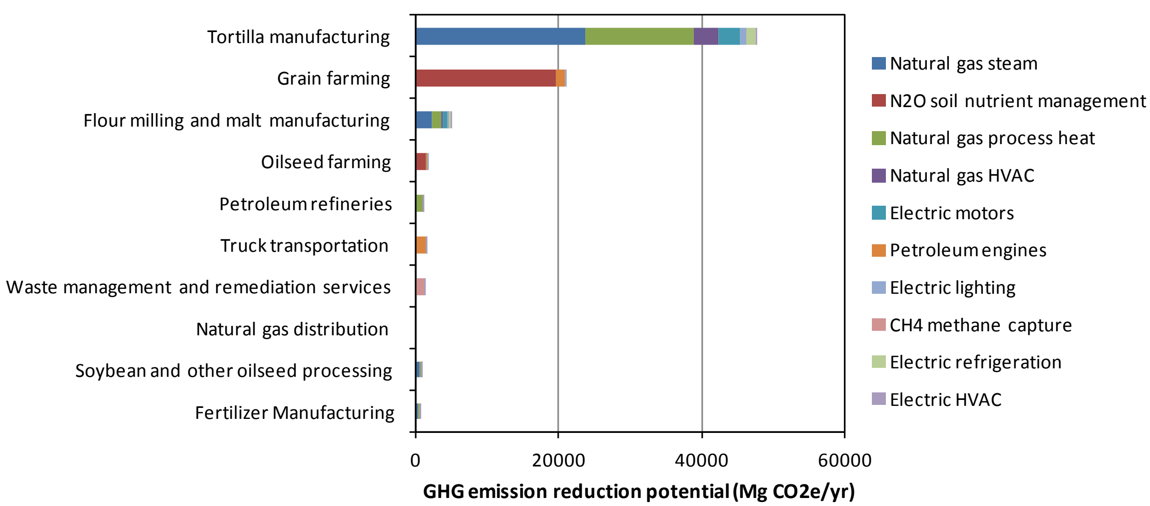
Top 10 contributing sectors to the energy footprint of product manufacture (MJ/$)



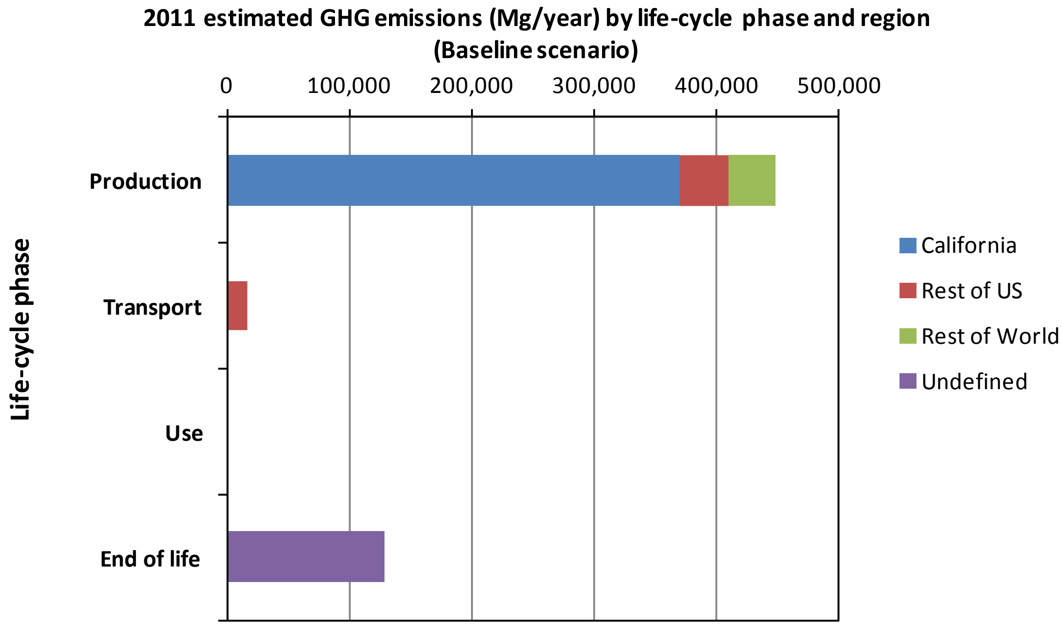
Top 10 contributing sectors to the GHG emissions footprint of product manufacture (MJ/$)



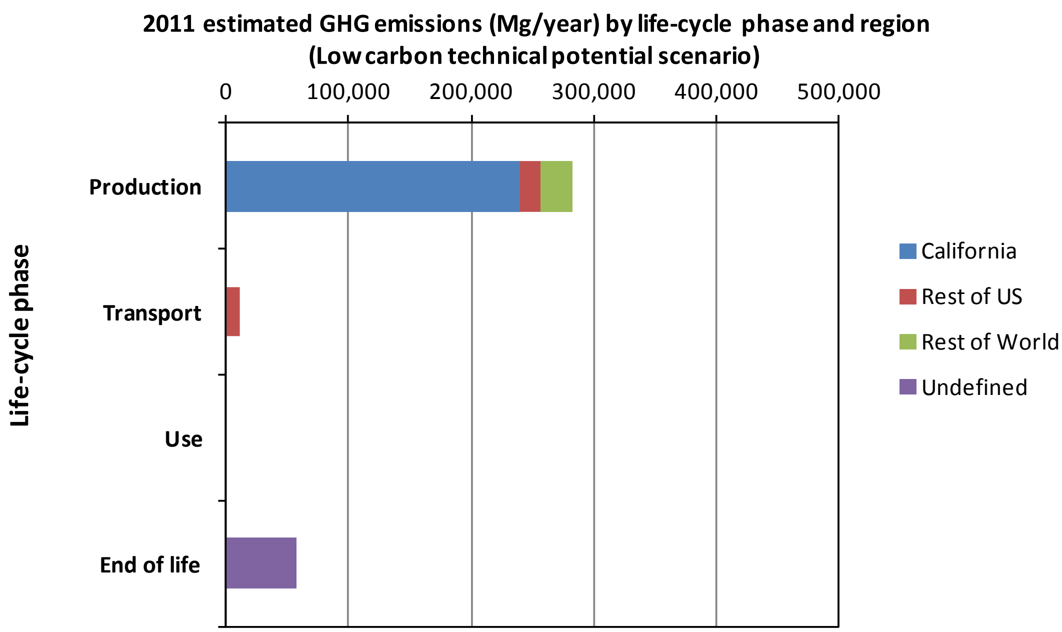
Top 15 supply chain efficiency and GHG mitigation reductions by improvement opportunity and emissions source



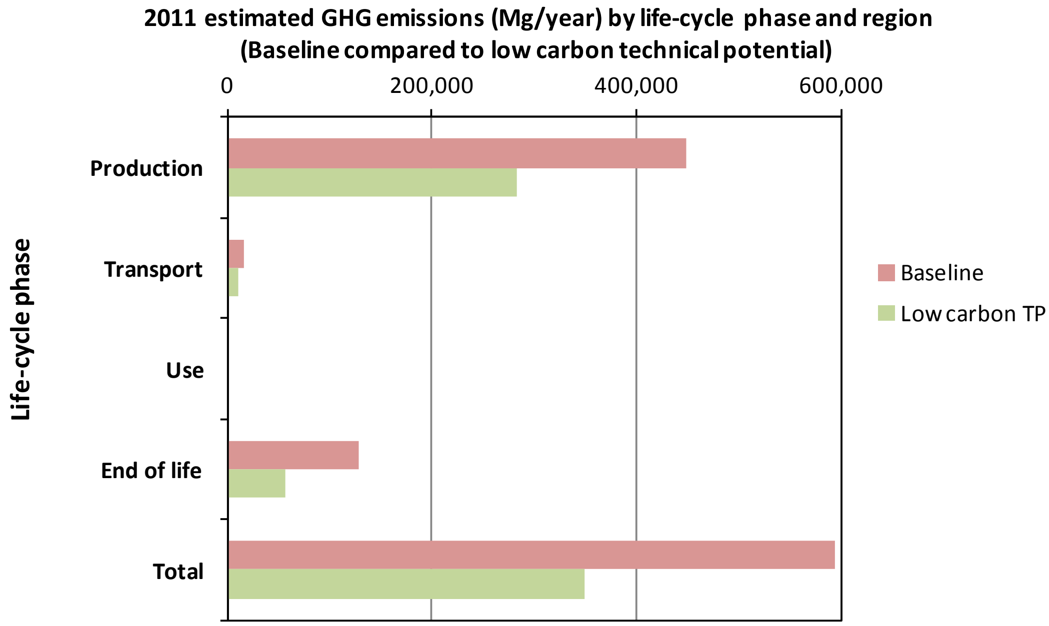
Top 10 supply chain efficiency and GHG mitigation reductions by sector and emissions source



Estimated 2011 GHG emissions by life-cycle phase and region in the baseline scenario



2011 GHG emissions by life-cycle phase and region in the low carbon technical potential scenario



Side by side comparison of baseline and low carbon technical potential scenarios

Total savings = 41%

21: Wine

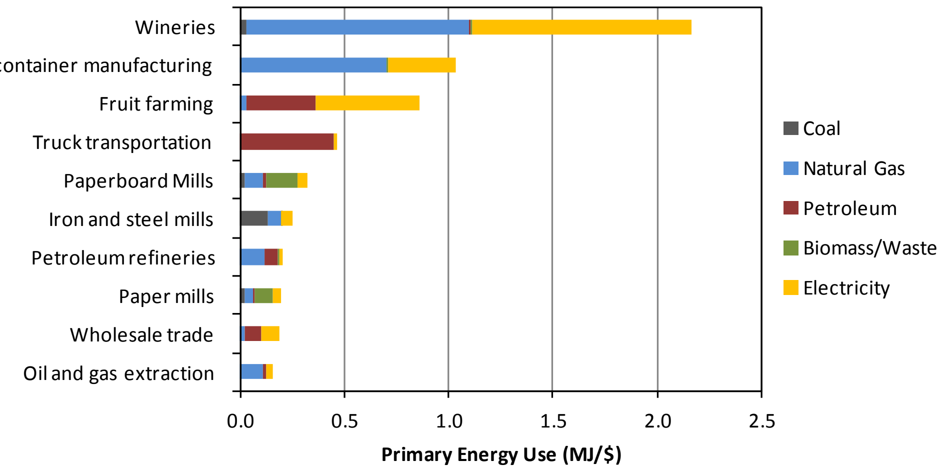
Product

Bottled wine packaged in 750ml glass bottles, with label and cork.

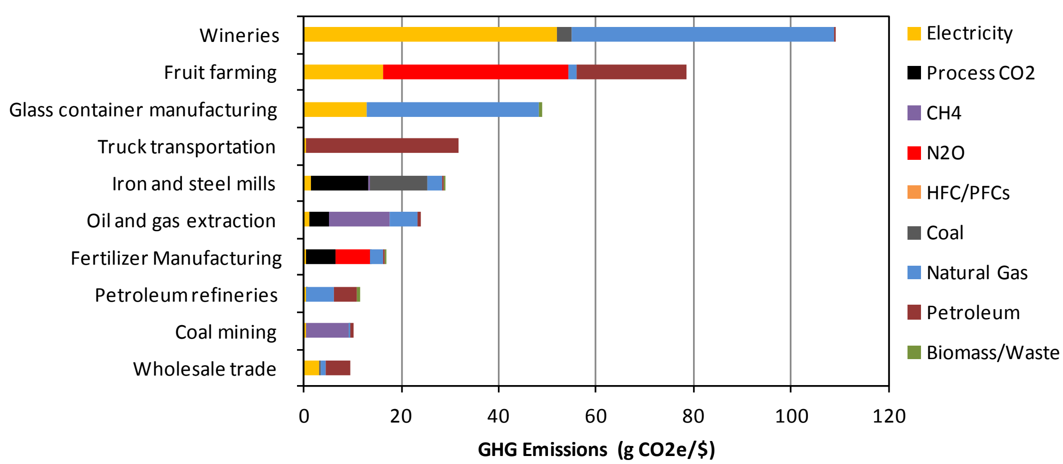
Life-cycle system description

The first step in wine production is the harvesting of grapes, after which the grapes are trucked to the winery for processing. At the winery, grapes are first de-stemmed; then they are crushed into juice and pulp in the crushing process. The next steps are fermentation and pressing. The fermentation process takes place at a controlled cool temperature for quality purposes, and represents one of the major energy uses in the typical winery. The next step is clarification, the purpose of which is to separate clear wine from spent yeasts and other solids after fermentation. The wine is then stored and aged under cool, temperature-controlled conditions in tanks or barrels. The aging process can range from a few weeks to a few years, depending on the type of wine. Finally, the wine is bottled and corked, and labels are applied (Galitsky et al. 2005).

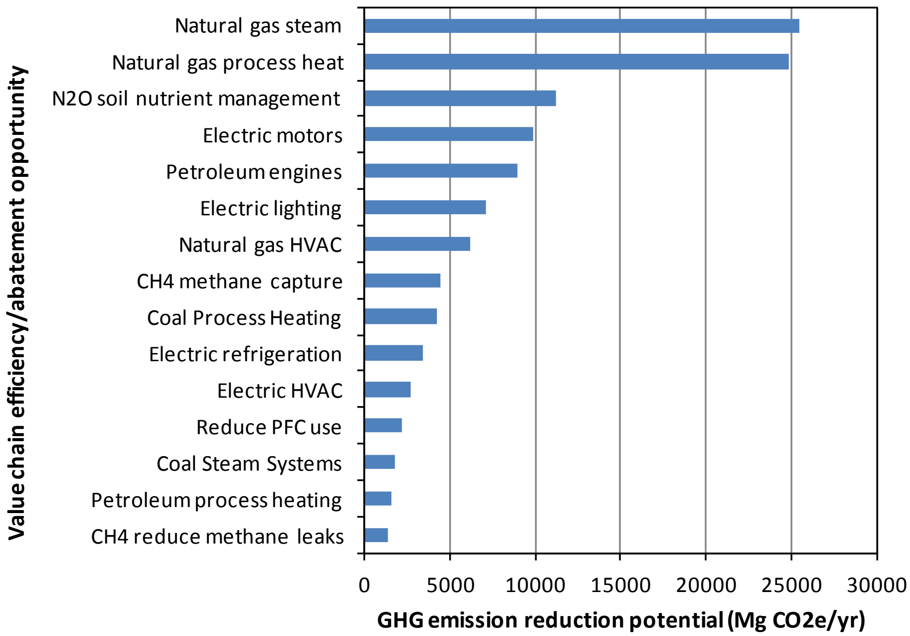
Most wine is then shipped in cases (commonly made of cardboard) to its final sales or consumption destination, which can include retail outlets, wholesalers, and food service operations. Prior to consumption many white wines are refrigerated, while most red wines are stored at room temperature. After consumption, wine bottles are either disposed of as municipal solid waste, or recycled.



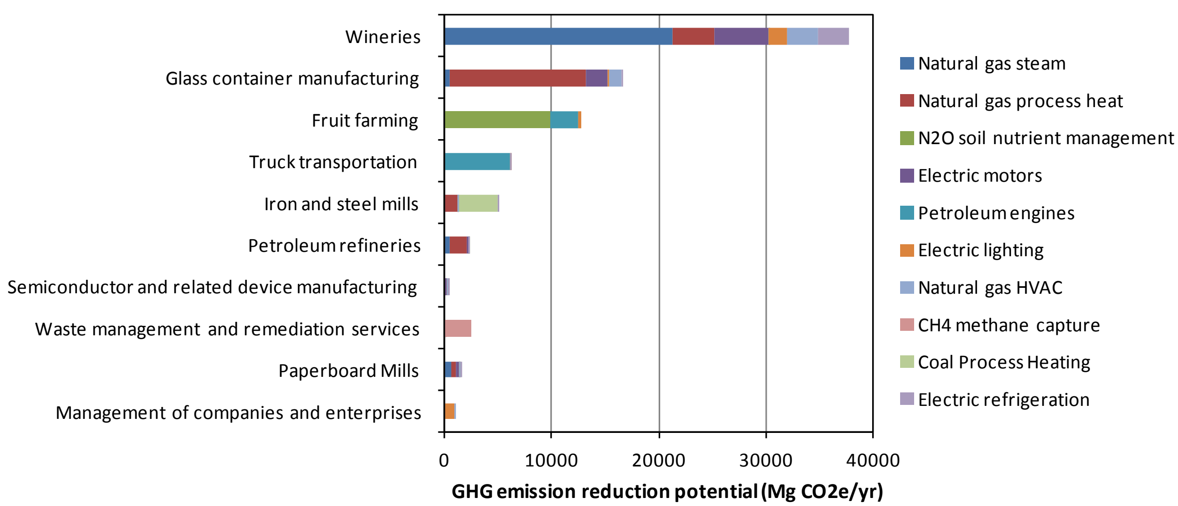
Top 10 contributing sectors to the energy footprint of product manufacture (MJ/$)



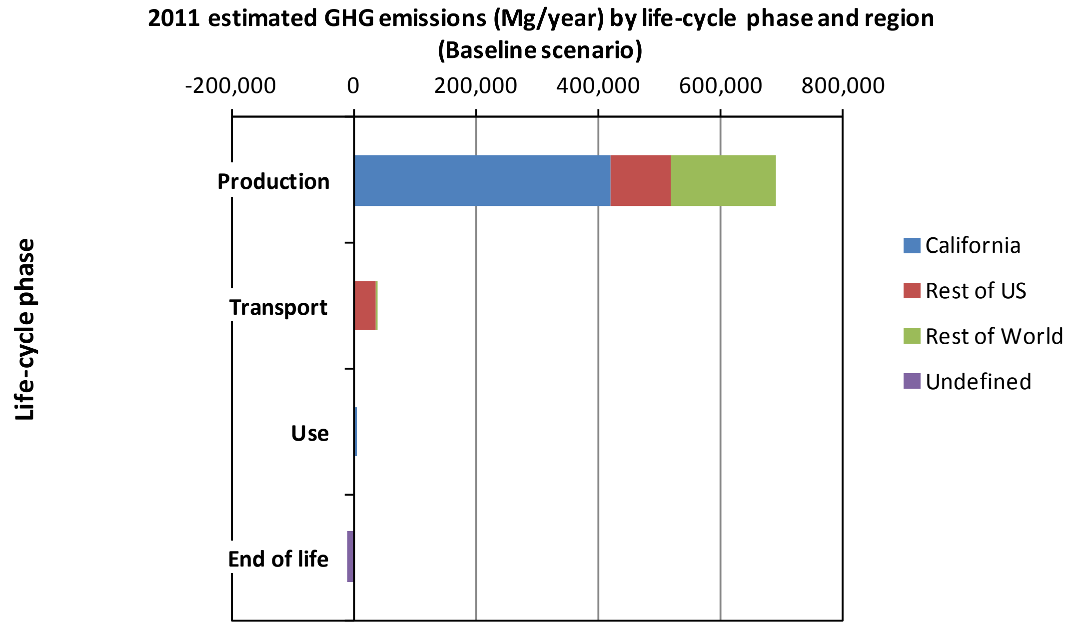
Top 10 contributing sectors to the GHG emissions footprint of product manufacture (MJ/$)



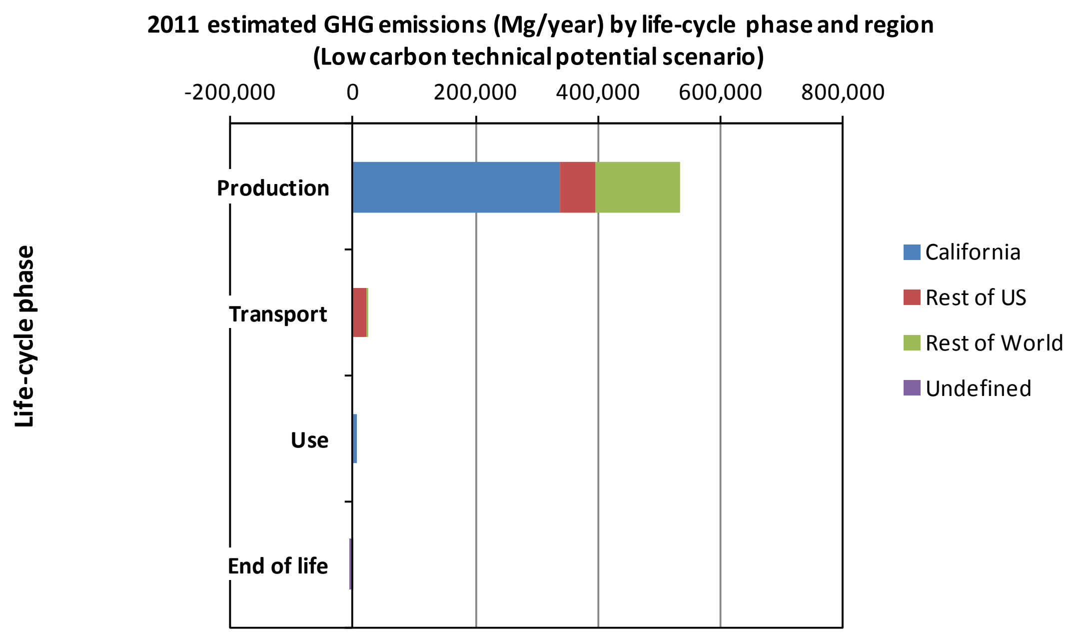
Top 15 estimated supply chain efficiency and GHG mitigation reductions by improvement opportunity and emissions source



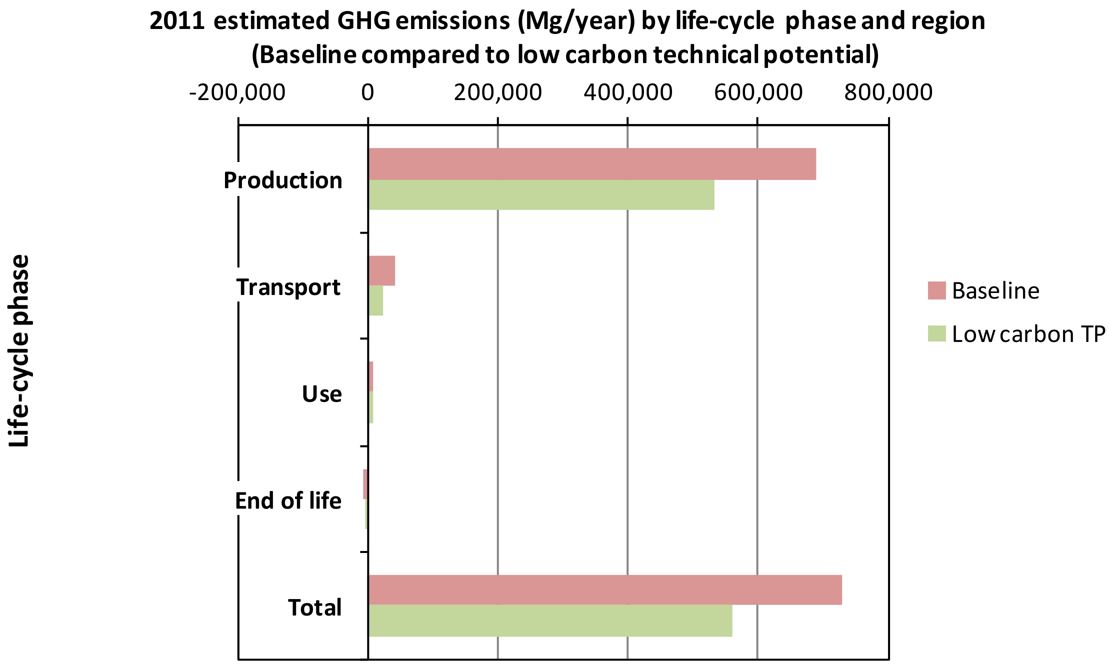
Top 10 estimated supply chain efficiency and GHG mitigation reductions by sector and emissions source



Estimated 2011 GHG emissions by life-cycle phase and region in the baseline scenario



2011 GHG emissions by life-cycle phase and region in the low carbon technical potential scenario



Side by side comparison of baseline and low carbon technical potential scenarios

Total savings = 23%

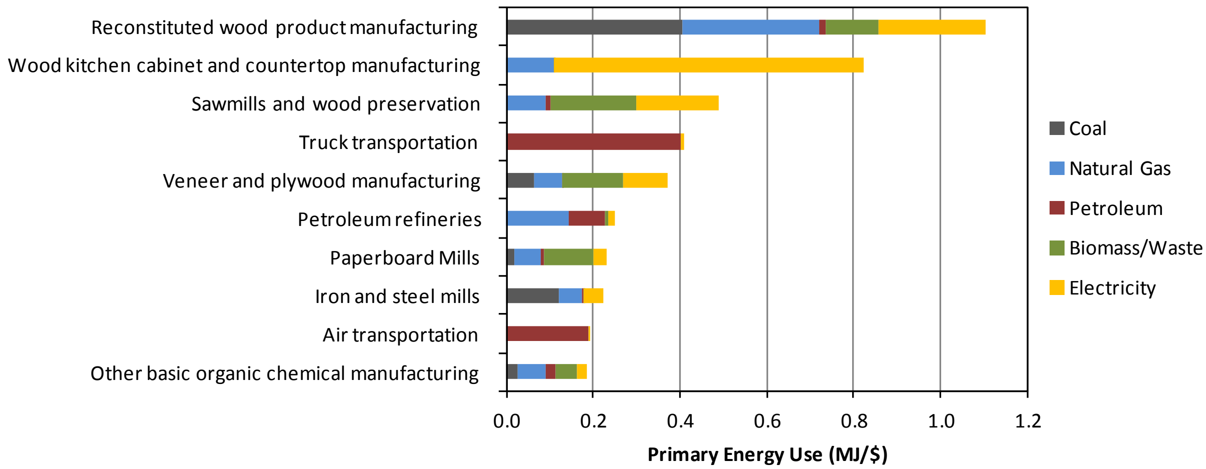
22: Wooden cabinet

Product

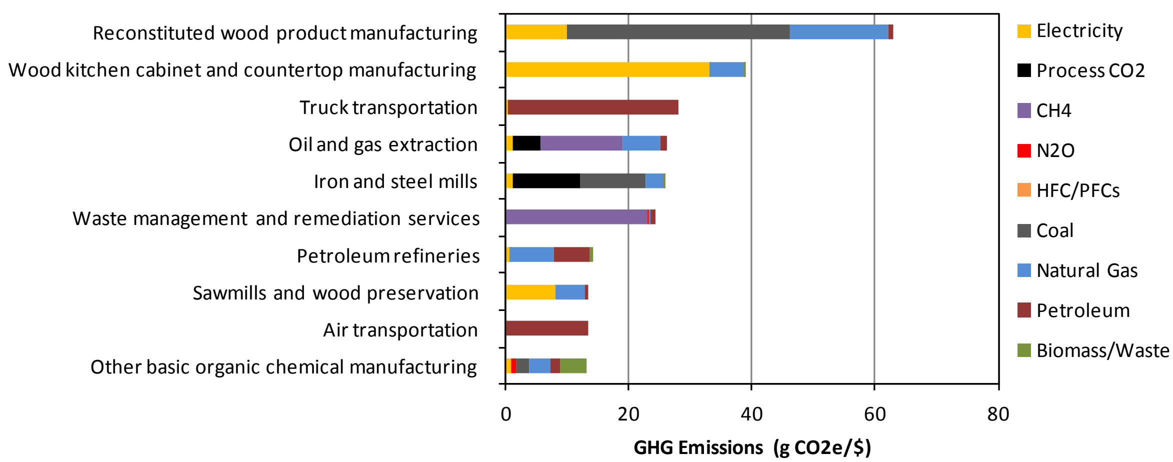
One kitchen cabinet

Life-cycle system description

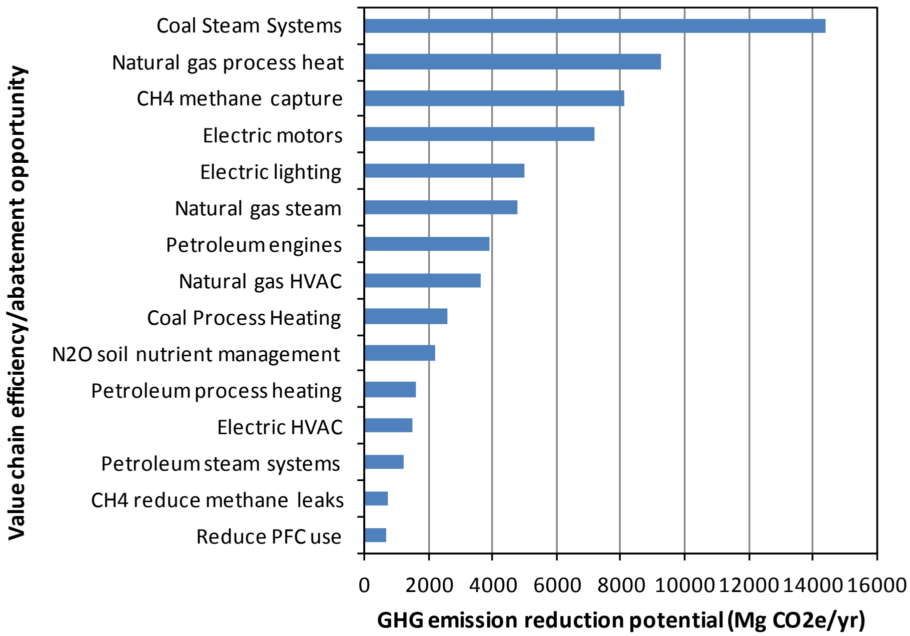
The life-cycle of a wooden cabinet begins with the harvesting of trees, which are then trucked to a primary sawmill to be cut into various dimensions, dried to remove moisture, and planed into final shape. Sawmill scraps and other wood waste can be used to make engineered boards, such as particle board, through combination with a resin binding ingredient and extrusion into final shape. Boards are then trucked to the furniture factory, where they are cut into final shapes and profiles (as needed), stained and assembled (typically with glue and mechanical fasteners), packaged, and shipped to the retailer or wholesaler. After installation in the home, a kitchen cabinet can stay in place for many years until the homeowner decides to replace it, most often for reasons of aesthetics. Discarded cabinets are typically sent to a landfill.



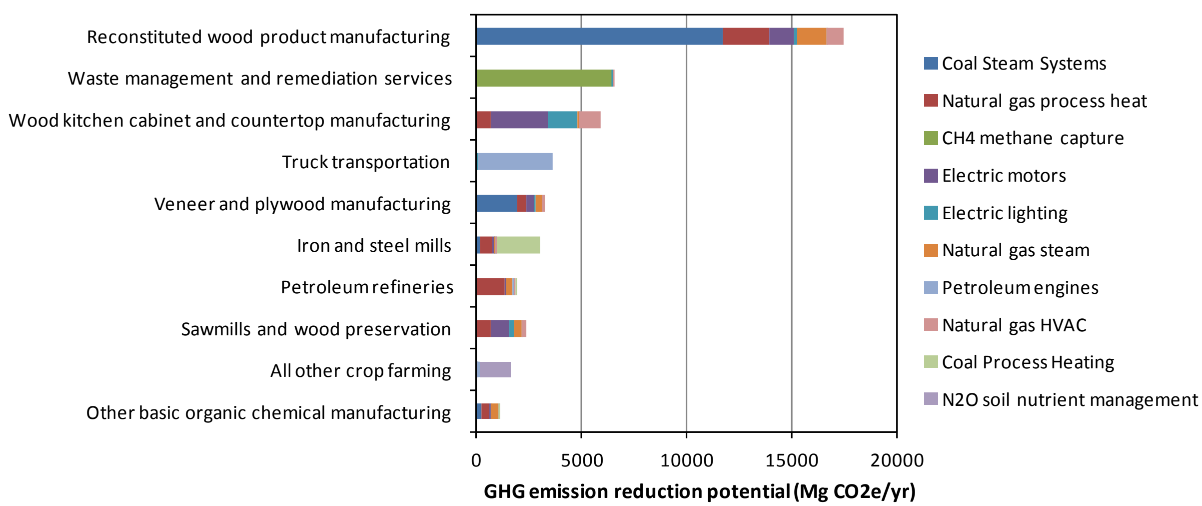
Top 10 contributing sectors to the energy footprint of product manufacture (MJ/$)



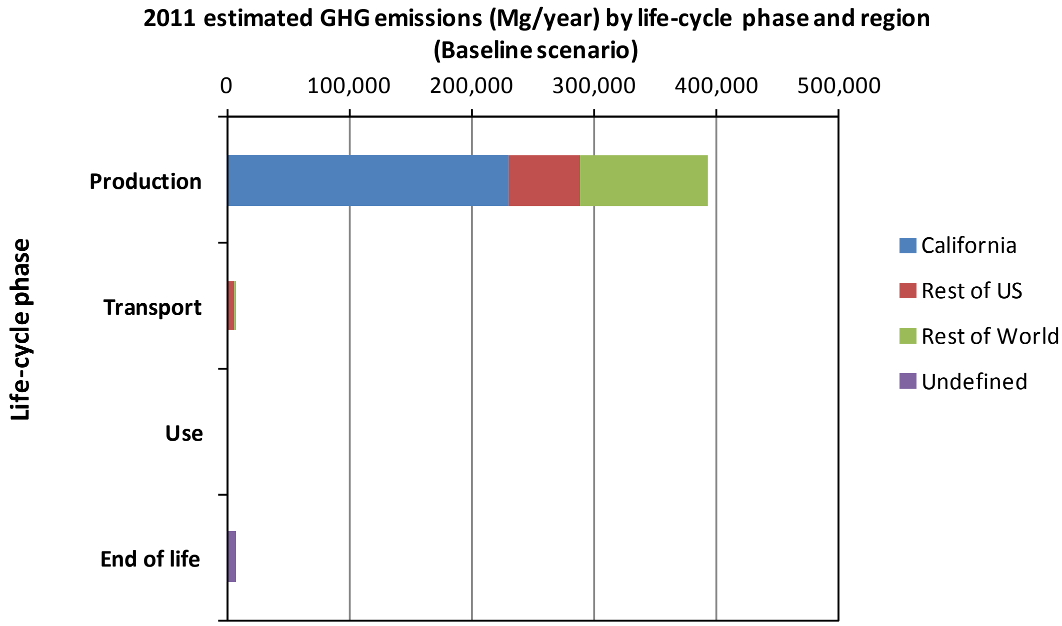
Top 10 contributing sectors to the GHG emissions footprint of product manufacture (MJ/$)



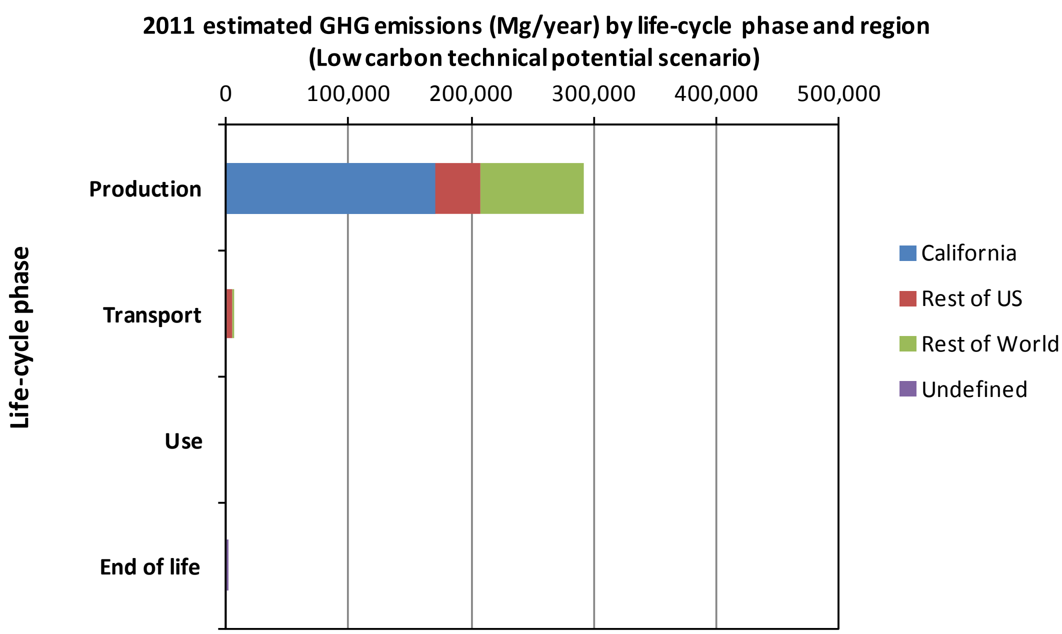
Estimated supply chain efficiency and GHG mitigation reductions by improvement opportunity and emissions source



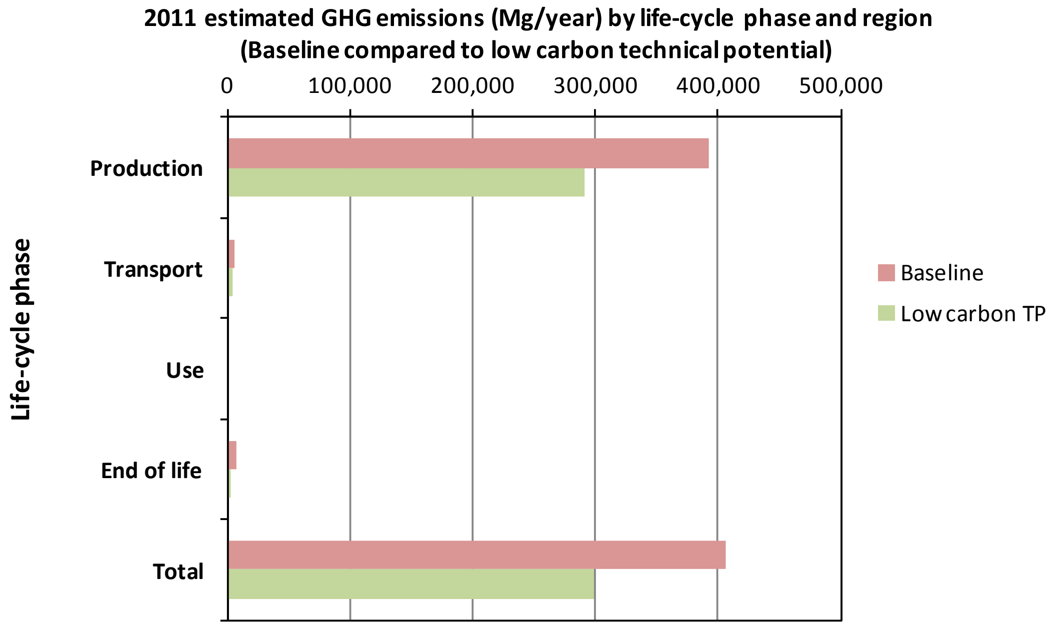
Estimated supply chain efficiency and GHG mitigation reductions by sector and emissions source



Estimated 2011 GHG emissions by life-cycle phase and region in the baseline scenario



2011 GHG emissions by life-cycle phase and region in the low carbon technical potential scenario



Side by side comparison of baseline and low carbon technical potential scenarios

Total savings = 27%