Arnold Schwarzenegger
Governor

OPTIMIZATION OF PRODUCT LIFE CYCLES TO REDUCE GREENHOUSE GAS EMISSIONS IN CALIFORNIA

DRAFT

PIER FINAL PROJECT REPORT

Prepared For:
California Energy Commission
Public Interest Energy Research Program

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June 2005
CEC-500-2005-110-D

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Contract No. 500-02-004
Master Research Agreement 015-006

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Acknowledgements

This work was supported by the California Energy Commission and the California Institute for Energy and Environment (CIEE) using support from the California Energy Commission through the U.S. Department of Energy (U.S. DOE) under Contract No.DE-AC03-76SF0098. This work does not necessarily represent the views of the California Energy Commission, its employees, or the State of California. Publication of these research results does not imply CIEE endorsement of or agreement with these findings, nor that of any of its sponsors.

Some data used in this report were obtained from previous projects funded by the UC Toxic Substances Research and Training Program through its Health Effects component (2000–2002) and Investigator-Initiated Grants program (2003–2004).

The authors would like to thank H. Scott Matthews and Györgyi Cicás (Carnegie Mellon University) for generating the results in Tables 5-1, 5-5, and 5-6, and for their helpful contributions to the follow-on research described in Section 5.

The authors would also like to thank John Bennett (California Portland Cement Company) and Andy O'Hare (Portland Cement Association) for their valuable comments on the cement and concrete analyses presented in this report and Robert Parkhurst (Hewlett-Packard) for his valuable comments on the personal computer analyses presented in this report. Thanks also to Edward Vine, the Environment Exploratory Grant Program Administrator at the University of California Office of the President California Institute for Energy and the Environment, for his helpful feedback during this research project.

Please cite this report as follows:

Masanet E., L. Price, S. de la Rue du Can, R. Brown, and E. Worrell. 2005. *Optimization of Product Life Cycles to Reduce Greenhouse Gas Emissions in California.* California Energy Commission, PIER Energy-Related Environmental Research. CEC-500-2005-110.

Preface

The Public Interest Energy Research (PIER) Program supports public interest energy research and development that will help improve the quality of life in California by bringing environmentally safe, affordable, and reliable energy services and products to the marketplace.

The PIER Program, managed by the California Energy Commission (Energy Commission), annually awards up to $62 million to conduct the most promising public interest energy research by partnering with Research, Development, and Demonstration (RD&D) organizations, including individuals, businesses, utilities, and public or private research institutions.

PIER funding efforts are focused on the following RD&D program areas:

- Buildings End-Use Energy Efficiency
- Energy-Related Environmental Research
- Energy Systems Integration
- Environmentally Preferred Advanced Generation
- Industrial/Agricultural/Water End-Use Energy Efficiency
- Renewable Energy Technologies

What follows is the final report for the Optimization of Product Life Cycles to Reduce Greenhouse Gas Emissions project, contract number 500-02-004, MRA 015-006, conducted by the Lawrence Berkeley National Laboratory. The report is entitled *Optimization of Product Life Cycles to Reduce Greenhouse Gas Emissions*. This project contributes to the PIER Energy-Related Environmental Research program area.

For more information on the PIER Program, please visit the Energy Commission's website www.energy.ca.gov/pier or contact the Energy Commission at (916) 654-4628.

Abstract

Product life-cycle optimization considers the reduction of energy use and greenhouse gas (GHG) emissions associated with the production, use, and end-of-life phases of products. In this scoping study, Lawrence Berkeley National Laboratory (LBNL) evaluated the opportunities related to product life-cycle optimization in California. LBNL estimated the energy consumption and associated GHG emissions of the manufacturing, use, and end-of-life phases for 50 products produced in California. The purpose of the 50-product life-cycle analysis (LCA) was to provide a preliminary estimate of the relative GHG emissions associated with a wide range of California-manufactured products. Explorative case studies to identify opportunities for GHG emissions reduction—as well as to identify practical opportunities and policy options in California for promoting life-cycle optimization—were then conducted for PCs, and for cement and concrete. These case studies found that there are significant life-cycle GHG mitigation options (as well as a number of policy options) that could potentially lead to reduction of product life-cycle GHG emissions in California. Follow-on research developed a revised product LCA methodology to facilitate a direct comparison of the life-cycle GHG emissions of different products in California, and to more accurately estimate the GHG emissions occurring within California's borders. The revised product LCA methodology was applied to the California pharmaceutical and semiconductor sectors.

Keywords: product life-cycle optimization, life-cycle analysis, greenhouse gas mitigation, climate change, industrial ecology, computers, cement, concrete, semiconductors, pharmaceuticals

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Executive Summary

Introduction

Product life-cycle optimization considers the reduction of energy use and GHG emissions associated with the production, use, and end-of-life phases of products. In this scoping study, LBNL evaluated the opportunities related to product life-cycle optimization in California. LBNL estimated the energy consumption and associated GHG emissions of the manufacturing, use, and end-of-life phases for 50 products produced in California. Explorative case studies to identify opportunities for GHG emissions reduction were then conducted for PCs, and for cement and concrete

Interest in this project's initial results resulted in follow-on research, which developed and applied a revised product life-cycle analysis (LCA) methodology to the California pharmaceutical and semiconductor sectors.

Purpose

The purpose of the 50-product life-cycle analysis was to provide a preliminary estimate of the relative GHG emissions associated with a wide range of California-manufactured products. The purpose of the explorative case studies was to identify practical opportunities and policy options in California for promoting life-cycle optimization for PCs and for cement and concrete.

The purpose of the follow-on work was to develop a revised methodology to provide a common and meaningful basis on which to compare GHG emissions of different products in California, and to more accurately estimate the GHG emissions occurring within California's borders.

Project Objectives

The first part of this project had three objectives: (1) identify 50 products manufactured in California and estimate the associated life-cycle energy consumption and GHG emissions of these products; (2) select products for two case studies to further explore and identify opportunities for life-cycle GHG emissions reductions in California; and (3) present an initial exploration of the practical opportunities and policy options in California for promoting product life-cycle optimization for the two case studies evaluated.

The follow-on work had two objectives: (1) to develop a revised LCA methodology for products manufactured and consumed in California that would provide a common basis for comparing the GHG emissions of different products, and that would more accurately estimate the manufacturing-stage GHG emissions occurring within California's borders; and (2) to apply the revised LCA methodology to the analysis on two major California-manufactured products.

Project Outcomes

Life-Cycle Analysis of 50 Products Produced in California

LBNL began by identifying the largest manufacturing sectors in California, using information on value added and total value of shipments in California, which was then used to provide guidance for identifying a sample of major products produced in California. The 50 products chosen by LBNL span a wide cross-section of California's manufacturing output and include such diverse items as personal computers (PCs), cheese, aircraft, wine, carpet, gasoline, and paint. All products have GHG emissions associated with their production and end-of-life phase; some products also have GHG emissions associated with their use. In addition, some products can be recycled at the end of their life, thus reducing product-specific GHG emissions.

For the *product manufacturing phase*, total GHG emissions per product varied widely for the 50 products evaluated, ranging from a low of 0.1 kilograms of carbon dioxide equivalent ($kgCO_2e$), or 0.03 kilograms of carbon equivalent (kgCe) for manufacture of an aluminum can to a high of 17 million kg CO_2e (5.3 million kgCe) for manufacture of an airplane.

Twenty of the 50 products evaluated use energy, either directly or indirectly, during the *product use phase* and thus produce energy-related GHG emissions. The products include airplanes, large industrial water pumps, semiconductor process machines, and cars.

Greenhouse gas emissions from the *product end-of-life phase* were also calculated for each of the 50 products. End-of-life phase energy use and GHG emissions are highest for products such as airplanes, asphalt paving mixtures, ready-mix concrete, and hydraulic cement, due to the large fraction of these products that was assumed to be disposed of via landfill.

Next, LBNL performed two case studies—one on PCs, and one on cement and concrete—to develop more detailed estimates of product-specific opportunities for life-cycle GHG mitigation in California. The two particular case studies were chosen for several important reasons. First, PCs and cement and concrete are extremely important products to California from both an economic and environmental perspective. Second, the availability of published data on the life cycle impacts of PCs and cement and concrete made a detailed analysis more feasible than for most of the other products on the LBNL 50-product list. Third, LBNL has established contacts in the PC and cement and concrete industries, which allowed for valuable industry feedback on the case study results. Lastly, LBNL has prior experience in the environmental analysis of the products in the two case studies.

Case Study: Personal Computers

California is the nation's largest manufacturer of computer equipment. California's importance to the $47 billion per year U.S. computer industry is undeniable: 33% of U.S. value-added computer manufacturing operations occur within the state. California's "hi-tech" sector, which manufactures the semiconductors, printed circuit boards, and myriad other electronic components supporting the global computer industry, employs over 700,000 people and is the second-largest source of employment in the state.

Table ES-1 provides LBNL's estimates of life-cycle GHG emissions for PCs in California, showing that PC manufacturing is responsible for GHG emissions of 4.18 million metric tons of carbon dioxide ($MtCO_2$) or 1.14 million metric tons of carbon (MtC) per year. Emissions from use of PCs in California total 1.72 $MtCO_2$ (0.47 MtC) per year, while emissions from PC disposal and demanufacturing operations at the end-of-life phase total 0.004 $MtCO_2$ (0.001 MtC) per year. The annual life-cycle GHG emissions for PCs in California are thus estimated to total 5.9 $MtCO_2$ (or 1.6 MtC). Table ES-1 also lists potential GHG mitigation options for life-cycle emissions from PCs manufactured, used, and discarded in California, as well as the estimated technical potential for GHG emissions reduction associated with each option.

Table ES-1. Life-Cycle GHG Emissions and Mitigation Options for Personal Computers in California

Life-Cycle Phase	Annual California GHG Emissions		GHG Mitigation Options	Potential Life-Cycle GHG Emission Reduction		
	($MtCO_2$)	(MtC)		(Mt CO_2)	(MtC)	(% of Total)
Manufacture	4.18	1.14	Reduce semiconductor PFC emissions	0.26	0.07	4
			Improve clean room energy efficiency	0.72	0.19	12
Use	1.72	0.47	Maximize PC energy efficiency	0.10	0.03	2
			Increase use of power management for PC control units	0.16	0.04	3
			Maximize PC power management	0.47	0.13	8
			Switch from CRT monitors to LCDs	0.48	0.13	8
End-of-Life	0.004	0.001	Maximize PC control unit recycling	0.0005	0.0001	0.01
			Upgrade PCs to extend their useful life	0.018	0.005	0.3
Total	**5.90**	**1.61**				

Case Study: Cement and Concrete

In 2002, California produced over 11 million metric tons of cement in eight plants, making it the largest cement-producing state in the United States. In California, the cement industry employs approximately 2,000 workers and has an annual value of shipments of $1 billion. The concrete and ready-mix industries in California together directly employ almost 19,000 employees and have an annual value of shipments of around $4.1 billion.

Table ES-2 provides LBNL estimates of life-cycle GHG emissions for cement and concrete production in California. Focusing just on the cement manufacturing facility, emissions are estimated to be 9.6 $MtCO_2$ (2.6 MtC). When emissions for raw materials mining, transport, and all other associated activities are included, total cement GHG emissions are estimated to be 10.4 $MtCO_2$ (2.8 MtC). Cement is used primarily to make concrete. In making concrete, energy is used for mining of the aggregates and sand; mixing, shaping, and curing the concrete; and transporting the raw materials, cement and concrete to the construction site. These activities result in concrete production emissions of 1.4 $MtCO_2$ (0.4 MtC), and including the negligible emissions associated with disposal, bring total emissions from cement and concrete production to 11.8 $MtCO_2$ (3.2 MtC). Table ES-2 also lists potential GHG mitigation options for life-cycle GHG emissions for cement and concrete produced, used, and discarded in California, as well as the estimated technical potential for GHG emissions reduction associated with each option.

Table ES-2. Life-Cycle GHG Emissions and Mitigation Options for Cement and Concrete in California

Life-Cycle Phase	Product	Annual California GHG Emissions		GHG Mitigation Options	Potential Life-Cycle GHG Emission Reduction		
		($Mt\ CO_2$)	(MtC)		($Mt\ CO_2$)	(MtC)	(% of Total)
Manufacture	Cement plant	9.6	2.6	Energy efficiency improvements	0.68	0.19	6.0
	Cement other	0.9	0.2	Use of waste fuels	0.62	0.17	5.4
	Total cement	10.4	2.8	Blended cement	0.55	0.15	4.8
	Concrete	1.4	0.4	Limestone addition to portland cement	0.44	0.12	3.8
	Total	11.8	3.2	CemStar® (steel slags) in portland cement	0.007	0.002	0.1
Use		0.0	0.0	Fuel efficiency for heavy trucks	0.04	0.01	0.4
End-of-Life		0.018	0.005	Concrete recycling	0.004	0.001	0.032
Total		**11.8**	**3.2**				

Follow-on Research: Revised LCA Methodology

The revised LCA methodology incorporates two significant improvements over the previous approach: First, it utilizes a California-specific Economic Input-Output Life Cycle Assessment (EIO-LCA) analysis method, which is currently under development at Carnegie Mellon University (CMU), to disaggregate the manufacturing-stage GHG emissions of a given product into GHG emissions occurring within California and GHG emissions occurring outside of California. The California EIO-LCA approach thus provides a more accurate estimate of California's actual manufacturing-stage GHG emissions than the previous 50-product LCA methodology. Second, it uses the total statewide life-cycle GHG emissions attributable to a given product in California each year as the basis of comparison. This allows significantly different products to be compared based on their total annual GHG "footprint" in California, which allows different products to be ranked based on their contribution to California's annual GHG emissions.

Revised LCA Methodology Case Studies: Pharmaceuticals and Semiconductors

The LBNL research team applied the revised LCA methodology to estimate the annual life-cycle GHG emissions of two products manufactured and consumed on a large scale in California—pharmaceuticals semiconductors in computers. It was estimated that the annual life-cycle GHG emissions arising from the manufacture and disposal of pharmaceuticals (specifically, prescription and over-the-counter drugs) in California amount to roughly 2.75 $MtCO_2$. For semiconductors contained in computers, it was estimated that the annual life-cycle GHG emissions of manufacture, use, and disposal in California amount to roughly 1.5 $MtCO_2$. Both case studies also uncovered significant supply chain GHG emissions occurring outside of California to support in-state manufacturing operations.

Conclusions

The LBNL research team found that the top 20 GHG-emitting products from a list of 50 products produced in California (from a life-cycle perspective) are: airplane, large industrial water pump, semiconductor process machine, car, commercial refrigerator, gas stove and range, air conditioner, metal window, tape storage drive, PC, hydraulic cement, asphalt paving mixture, microwave oven, wooden table, semiconductor chip, ready-mix concrete, scanner, printed circuit board, tires, and bicycle.

The case study of life cycle emissions from PCs in California found that PC manufacturing in California is responsible for annual GHG emissions of 4.18 $MtCO_2$ (1.14 MtC). The annual emissions from use of PCs in California amount to 1.72 $MtCO_2$ (0.47 MtC). At the end-of-life stage, California's PCs generate another 0.004 $MtCO_2$ (0.001 MtC) per year. LBNL identified a number of opportunities for reducing GHG emissions from PC manufacture and use in California—opportunities that could potentially save over 2 $MtCO_2$ (MtC) each year.

For the cement and concrete case study, LBNL estimated that total annual GHG emissions from cement and concrete production in California are 11.8 $MtCO_2$ (3.2 MtC) LBNL also identified a number of opportunities for reducing GHG emissions from cement and concrete manufactured and used in California. Those opportunities could potentially save nearly 2 $MtCO_2$ (MtC) each year.

The revised LCA methodology developed in the follow-on research offers California an improved method for conducting LCAs for a wide range of products by providing a common basis for product-to-product comparisons and a more accurate estimate of in-state versus out-of-state product GHG emissions. The revised LCA methodology can therefore serve as a powerful screening tool for identifying specific products and product life-cycle stages in California that can be targeted for more detailed life-cycle optimization studies to reduce California's GHG "footprint," both within and outside the state.

Recommendations

Policy Recommendations: Personal Computers

LBNL's analysis has shown that significant opportunities exist for GHG reductions at each stage of the product life cycle for PCs, giving rise to the following areas for potential policy initiatives:

- Promotion of further clean room energy efficiency improvements
- Promotion of institutional policies and energy efficiency awareness campaigns that lead to more widespread usage of PC power management features, the purchase of more energy-efficient control units and displays, and the use of LCDs instead of CRT monitors
- Promotion of upgrading to extend the life of PCs for as long as possible, through the establishment of institutional policies and public awareness campaigns
- Promotion of green procurement initiatives to guide institutional purchasing decisions

Policy Recommendations: Cement and Concrete

LBNL's analysis found that opportunities exist to enhance GHG emission reduction throughout the manufacture, use, and disposition of cement and concrete through the following policy initiatives:

- Promotion of further energy efficiency improvements in cement manufacturing
- Promotion of procurement and product specifications for changes in the composition of cement (e.g., blended cement and limestone addition) by government agencies involved in construction (e.g., the California Department of Transportation)
- Promotion of the use of alternative fuels such as tires and other wastes to replace coal-burning in cement kilns by both waste management agencies and through air quality permitting of those cement plants that can safely incinerate wastes in the kiln
- Promotion by waste management agencies of the increased recycling of concrete for use in making aggregate

Future Research Recommendations: Follow-on Research

This follow-on research also identified several important areas for continued research:

- Further development and refinement of input-output based environmental models for California
- The development of comprehensive use-stage energy consumption and GHG emissions databases for major energy-consuming products used in California
- The analysis of additional products to further evaluate the revised LCA methodology
- The expansion of the revised LCA methodology to include other important environmental metrics, such as energy consumption, criteria air pollution, and solid waste generation
- The inclusion of recycling "credits" to capture the environmental benefits of materials recycling in California.
- The development of comprehensive databases on annual waste flows and recycling statistics for major products in California

Benefits to California

The methodology developed by this project can be used to assess the life-cycle GHG emissions associated with specific products manufactured in California. In addition, this work identified targeted policy recommendations for reducing product-specific life-cycle GHG emissions in California that are associated with PCs, and with cement and concrete. The case study information in this report can help policy makers assess the benefits of those policies for those two sectors.

Using such a life-cycle optimization approach to evaluate the potential for product-specific GHG emissions reductions enables California's policymakers to identify mitigation options beyond those that are more commonly recognized. This type of systematic approach provides policymakers with a wider breadth of information regarding both GHG emissions sources in California and potential GHG emissions mitigation options.

The revised LCA methodology developed in the follow-on research offers California an improved method for conducting LCAs for a wide range of products by providing a common basis for product-to-product comparisons and a more accurate estimate of in-state versus out-of-state product GHG emissions. The revised LCA methodology can therefore serve as a powerful screening tool for identifying specific products and product life-cycle stages in California that can be targeted for more detailed life-cycle optimization studies to reduce California's GHG "footprint."

1. Introduction

1.1. Background and Overview

Many opportunities exist for reducing the energy use and greenhouse gas (GHG) emissions associated with the production, use, and disposal of products in California. Reducing energy use and GHG emissions can improve the state's competitive position in a global business environment, while also addressing environmental problems such as climate change, air pollution, and waste export.

Product life-cycle optimization considers the reduction of energy use and GHG emissions associated with the production, use, and disposition of products. The product life-cycle optimization approach can identify efficiency improvements related to both direct energy use for manufacturing products and the indirect energy embodied in the products' materials. Savings of the energy consumed during product use and product end-of-life disposition can also be quantified. Policies or measures to minimize energy use and GHG emissions during the product life cycle can address energy efficiency, material substitution, recycling and re-use, product design, and procurement practices.

In this project, Lawrence Berkeley National Laboratory (LBNL) evaluated the opportunities related to product life-cycle optimization for products produced in California, given the structure of the economy, the mix of products manufactured in the state, and the opportunities to support specific policies and measures within the state. Such optimization involves evaluating the opportunities to improve the sustainability of industrial production up and down the product life cycle, from resource extraction to product manufacturing to product use and product disposition.

1.2. Project Objectives

The main body of this project had three objectives: (1) identify 50 products manufactured in California and estimate the associated life-cycle energy consumption and GHG emissions of these products; (2) select products for two case studies to further explore and identify opportunities for life-cycle GHG emissions reductions in California; and (3) present an initial exploration of the practical opportunities and policy options in California for promoting product life-cycle optimization for the two case studies evaluated.

Interest in this project's initial results led to follow-on research, the objectives of which are discussed in Section 5.

1.3. Report Organization

The report begins with a description of the project approach in Section 2, including a definition of life-cycle optimization and the methodology used. Section 3 then discusses project outcomes, including identification of major products produced in California and estimation of product life-cycle GHG emissions. Section 3 also includes two case studies to identify product-specific opportunities for potential GHG emissions reductions. Section 4 provides conclusions and recommendations. Section 5 discusses follow-on work to this research, which revised the LCA methodology presented in the other sections and applied it to two additional case study products.

2. Project Approach

2.1. Definition of Life-Cycle Optimization

Life-cycle optimization is the evaluation of environmental burdens (including energy use and GHG emissions) associated with all aspects of a product's life, in an effort to identify approaches for minimizing those burdens. Energy consumption and GHG emissions can occur during product manufacturing, product use, and at the end of a product's life. Possibilities may exist at each of these steps to reduce GHG emissions. A life-cycle optimization approach evaluates the potential to reduce GHG emissions at each of these steps, provides a comprehensive overview of the product's full life-cycle GHG emissions, and assists in identifying possible areas for emissions mitigation.

2.2. Methodology

LBNL began by identifying the largest manufacturing sectors in California. The U.S. Census *Annual Survey of Manufactures* (U.S. Census 2003) provides economic activity information for the manufacturing sectors in California. This information is provided at the 3- and 4-digit North American Industrial Classification System (NAICS)[1] level. Information on the manufacturing sector's value added and total value of shipments was used to begin to identify major products produced in California.

Using the U.S. Census data and the *2004 Directory of California Manufacturers*, LBNL identified a representative sample of 50 products manufactured in California (U.S. Census 2003; MNI 2004). For each of these 50 products, LBNL then estimated the GHG emissions arising from the manufacture, use, and end-of-life disposition of each product.

Energy-related carbon dioxide (CO_2) emissions associated with the manufacture of each product were calculated using information from the *Economic Input-Output Life-Cycle Assessment (EIO-LCA) Database* of Carnegie Mellon University's Green Design Initiative (CMU-GDI 2004). This database provides energy use and related GHG emissions, by fuel, for every million dollars (in 1997 dollars) of economic activity in a given sector. The current price of each of the 50 products was identified, converted to 1997 dollars using the Consumer Price Index (U.S. Department of Labor 2005), and input into the database to generate manufacturing life-cycle GHG emissions for each product. The EIO-LCA database uses the product price to calculate the energy use and related GHG emissions from fuels used to manufacture each product using U.S. EPA AP-42 emissions factors (U.S. EPA 1995). LBNL used the consumer price for each product considered in this analysis (i.e., the final product price paid by the consumer) instead of product price (i.e., the manufactured cost of the product), due to the general lack of product price data in the public domain. The use of

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[1] NAICS is the first-ever North American industry classification system. The system was developed by the Economic Classification Policy Committee, on behalf of the U.S. Office of Management and Budget, in cooperation with Statistics Canada and Mexico's Instituto Nacional de Estadística, Geografía e Informática to provide comparable statistics across the three countries. For the first time, government and business analysts will be able to compare directly industrial production statistics collected and published in the three North American Free Trade Agreement countries (U.S. Census 2005a).

consumer price instead of product price will likely result in an overestimation of a product's life-cycle energy consumption and GHG emissions by EIO-LCA. However, the purpose of the 50-product analysis is to provide only a rough estimate of the relative GHG emissions associated with each product, thus the uncertainty introduced by using consumer prices is acceptable, considering the preliminary nature of this analysis.

Process-related GHG emissions from product manufacturing, which are defined as the non-energy-related GHG emissions of a manufacturing process, are not included in the EIO-LCA database. Therefore, LBNL calculated process-related CO_2 emissions for cement manufacturing based on previous work which indicated that CO_2 emissions from the calcination process that took place during clinker production were roughly equal to those from fuel consumption (Worrell et al. 2001). LBNL also calculated the process-related GHG emissions arising from semiconductor manufacture in the production process for semiconductor chips and PCs. These emissions include perfluorocarbons, trifluoromethane (CHF_3), nitrogen trifluoride (NF_3), and sulfur hexafluoride (SF_6), which are collectively referred to as perfluorocompounds (PFCs). These emissions were estimated in the detailed analysis of PCs presented later in this report.

During the product use phase, only a subset of the selected products use energy either directly or indirectly, producing energy-related GHG emissions. Among the 50 products, those that use energy directly include PCs and related computer equipment, telephones, airplanes, semiconductor process machines, cars, large industrial water pumps, air conditioners, refrigerators, blood pressure monitors, stoves, and microwave ovens. For some of these products, the research team used annual energy consumption values drawn directly from the literature. Office equipment (PC and scanner) energy consumption values were from Kawamoto et al. (2001). Cordless phone consumption was from Roberson et al. (2004). For residential air conditioners, the team used average cooling consumption values for California homes from the Department of Energy's Residential Energy Consumption Survey (U.S. DOE 1999). For commercial refrigerators, the team used the annual consumption of a two-door, reach-in refrigerator from Westphalen et al. (1996). For residential ranges and microwave ovens, the team used annual consumption values reported by U.S. DOE (1998), weighted by the share of electric and gas ranges in California reported by U.S. DOE (1999).

For other devices, the research team was not able to find annual energy consumption values directly in the literature, so annual consumption estimates were derived using typical power levels from the literature and assuming plausible operating patterns (i.e., annual hours of use per year). For cellular phones, the team used power levels for plug-in chargers from Roth et al. (2002), and assumed that the charger was plugged-in continuously, with the phone handset connected and charging one-third of the time. The power level for tape storage drives[2] is drawn from Matthews (2002), with the devices assumed to be on all the time. Average energy consumption values for airplanes and cars were estimated based on data available in the literature (EEA 2004; IEA 2003; Kitou 2002; ORNL 2004; Davis and Diegel 2003; U.S. BTS 2003; U.S. DOT 2004).

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[2] Tape storage drives are used for backup storage of data in computer systems.

Products that use energy indirectly—which the research team defined as products requiring refrigeration or influencing the energy consumption within a building—include milk, cheese, meat, soft drinks, and metal windows. Metal windows influence energy consumption by increasing a building's heat loss or gain, which in turn causes the building's heating and cooling equipment to consume energy. To calculate this induced energy consumption in California residences, the team started with the annual heating and cooling energy consumption for the average California home in 1997 (24.3 GJ) (U.S. DOE 1999). The team then multiplied by the fraction of heating and cooling load due to windows—33% (Huang et al. 1999)—to estimate heating and cooling consumption attributable to the windows in an average home. Finally, the team divided by the number of windows in a typical home—8 (U.S. DOE 1999)—to estimate per-window annual energy consumption. For the products that require refrigeration, the team assumed an average refrigeration life of 10 days and allocated a percentage of the daily energy consumed by a commercial refrigerator (Westphalen et al. 1996) to each product based on the ratio of the product volume to the interior volume of the refrigerator.

For all energy-consuming products, the team multiplied the annual average energy consumption value for each product by the average product lifetime to determine the product's use phase energy consumption.

Greenhouse gas emissions associated with product use were then calculated by multiplying the lifetime energy consumption by fuel with fuel-specific emission factors (Marnay et al. 2002; IPCC 1997). For electricity, the research team used a 1999 average emission factor of 0.396 kilograms (kg) CO_2/kilowatt-hour (kWh) (0.108 kgC/kWh) for the State of California that includes electricity imports and generation of electricity for use in California by out-of-state generation facilities owned by California utilities (Marnay et al. 2002). Alternative electricity emission factors for California are available for the Western Electricity Coordinating Council, California and Southern Nevada Sub region (WECC/CNV) via the Clean Air and Climate Protection Software (CACPS) Database (STAPPA/ALAPCO and ICLEI 2003). The average electricity emission factor for 2000 to 2002 provided by this database is 0.318 kgCO_2/kWh (0.087 kgC/kWh), slightly lower than the value used. Marginal electricity emission factors, which are commonly used to calculate savings when a project or policy reduces energy use "at the margin"—or the last kWhs produced at any given time—are significantly higher than average emission factors in California, due partially to the fact that marginal electricity is often imported from coal-burning power plants in the U.S. Southwest (Marnay et al. 2002). The CACPS marginal electricity emission factors for the WECC/CNV region for 2000 to 2002 range from 0.771 kg CO_2/kWh (0.21 kgC/kWh) to 0.907kg CO_2/kwh (0.247 kgC/kWh) (STAPPA/ALAPCO and ICLEI 2003). Use of marginal electricity emission factors for these calculations would more than double the estimated GHG emissions from electricity-using devices during the product use phase.

For the end-of-life phase, the energy consumption and GHG emissions associated with the disposal of each product were calculated using published life-cycle inventory (LCI) data for municipal solid waste collection, landfill equipment operations, and landfill gas emissions. The average consumption of diesel fuel associated with solid waste collection was assumed to be 9.1 liters/t (McDougall et al. 2001) and the average consumption of diesel fuel associated with landfill equipment was assumed to be 5.8 liters/t (Franklin Associates 1994).

The GHG emissions arising from diesel consumption were calculated using published emission factors for diesel combustion (BUWAL 1998).

The GHG emissions from landfill gas produced by the biodegradable products in this study (such as food, newspapers, and wooden furniture) were calculated using average landfill gas composition data (52.8% methane, 44.1% CO_2) and the assumption that 250 cubic meters (m^3) of landfill gas are generated for each tonne of biodegradable waste disposed (McDougall et al. 2001). Based on 2004 tipping data from California landfills available from the California Integrated Waste Management Board (CIWMB), it was estimated that 67% of solid waste is disposed at sites that employ landfill gas-to-energy (LFGTE) technology, 12% of solid waste is disposed at sites where landfill gas is flared, and 21% of solid waste is disposed at sites where landfill gas is vented to the atmosphere (CEC 2002a; CIWMB 2004a). GHG emissions from the combustion of landfill gas at LFGTE and gas flaring sites were calculated using stoichiometric combustion balances. The average energy "credit" arising from electricity generation at LFGTE facilities was assumed to be 1.5 kWh per m^3 of landfill gas captured (McDougall et al. 2001), and the associated GHG emissions "credit" was calculated by multiplying the electricity "credit" by the average electricity CO_2 emission factor of 0.396 kg CO_2/kWh (Marnay et al. 2002). The net GHG emissions attributable to landfill gas for each biodegradable product were then calculated by subtracting the GHG emissions "credit" from the GHG emissions arising from combustion and venting.

The average percentage of each product's mass that is disposed of via landfill in California was estimated based on published recycling data for each product category (CIWMB 2004b; CIWMB 2003a; CIWMB 1999; Matthews and Matthews 2003; Steel Recycling Institute 2003; U.S. EPA 2003a). Only the energy and GHG emissions of product disposal were calculated in this study; the calculation of energy and GHG "credits" associated with materials recycling requires detailed data on product materials composition and recycling processes, and was therefore deemed beyond the scope of the 50-product life-cycle analysis (LCA). As such, this 50-product analysis only covers the initial life cycle of each product and does not address the fact that some portion of the materials in each product might be recycled. The implication of excluding recycling "credits" in the 50-product LCA is that the potential benefits of recycling (i.e., the avoidance of GHG emissions realized via virgin materials substitution) are not quantitatively acknowledged for each product. However, as many recyclable materials in California are exported for recycling (CIWMB 1996), it is not clear the extent to which any recycling "credits" attributable to a given product would actually be realized in California. This report explores the issue of recycling "credits" in more detail in the case study of PCs, in Section 3.3.1.

3. Project Outcomes

LBNL calculated GHG emissions associated with the manufacture, use, and end-of-life phases of 50 products manufactured in California. LBNL then performed two case studies—one on PCs, and one on cement and concrete—to develop more detailed estimates of product-specific opportunities for life-cycle GHG mitigation in California. LBNL then evaluated the practical opportunities and policy options in California for promoting product life-cycle optimization related to the two case studies.

3.1. Identification of Major Products Produced in California

LBNL began by identifying the largest manufacturing sectors in California using the U.S. Census *Annual Survey of Manufactures* for California (U.S. Census 2003). Table 3-1 provides information on manufacturing sectors (at the NAICS 4-digit level) in California ranked by economic activity, as expressed by value added[3] and total value of shipments[4] (U.S. Census 2003). The top manufacturing sectors in California include semiconductor and other electronic component manufacturing, communications equipment manufacturing, navigational, measuring, medical, and control instruments manufacturing, aerospace product and parts manufacturing, and petroleum and coal products manufacturing.

This information on the state's manufacturing sectors was then used to provide guidance for identifying major products produced in California. Using the U.S. Census data as well as detailed information on manufacturing facilities in the state provided by the *2004 Directory of California Manufacturers*, LBNL identified 50 products manufactured in California that were generally in the top manufacturing sectors (U.S. Census 2003; MNI 2004). Table 3-2 lists the 50 products and 50 associated manufacturing sectors that were identified for this project. The products chosen span a wide cross-section of California's manufacturing output and include such diverse items as PCs, cheese, aircraft, wine, carpet, gasoline, and paint. All products chosen have GHG emissions associated with their production and end-of-life phase; some products also have GHG emissions associated with their use. In addition, some products can be recycled at the end of their life, thus reducing GHG emissions associated with landfill disposal. Issues associated with the various types of products selected are discussed further below.

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[3] Value added means "being or pertaining to something added to a product to increase its value or price." Value added is a measure of manufacturing activity that is derived by subtracting the cost of materials, supplies, containers, fuel, purchased electricity, and contract work from the value of shipments (products manufactured plus receipts for services rendered). The result of this calculation is adjusted by the addition of value added by merchandising operations (i.e., the difference between the sales value and the cost of merchandise sold without further manufacture, processing, or assembly) plus the net change in finished goods and work-in-process between the beginning- and end-of-year inventories (U.S. Census 2003).

[4] *Value of shipments* is defined as the net selling value of all products shipped and includes extensive duplication, since products of some industries are used as materials of others (U.S. Census 2003).

Table 3-1. Manufacturing in California Ranked by Value Added and Total Value of Shipments

NAICS Code	Value Added	NAICS Code	Total Value of Shipments
3344	Semiconductor and other electronic component manufacturing	3342	Communications equipment manufacturing
3342	Communications equipment manufacturing	3344	Semiconductor and other electronic component manufacturing
3345	Navigational, measuring, medical, and control instruments manufacturing	3341	Computer and peripheral equipment manufacturing
3364	Aerospace product and parts manufacturing	3241	Petroleum and coal products manufacturing
3341	Computer and peripheral equipment manufacturing	3345	Navigational, measuring, medical, and control instruments manufacturing
3254	Pharmaceutical and medicine manufacturing	3364	Aerospace product and parts manufacturing
3241	Petroleum and coal products manufacturing	3254	Pharmaceutical and medicine manufacturing
3391	Medical equipment and supplies manufacturing	3261	Plastics product manufacturing
3261	Plastics product manufacturing	3391	Medical equipment and supplies manufacturing
3231	Printing and related support activities	3152	Cut and sew apparel manufacturing
3152	Cut and sew apparel manufacturing	3114	Fruit and vegetable preserving and specialty food manufacturing
3327	Machine shops, turned product, and screw, nut, and bolt manufacturing	3231	Printing and related support activities
3114	Fruit and vegetable preserving and specialty food manufacturing	3115	Dairy product manufacturing
3399	Other miscellaneous manufacturing	3332	Industrial machinery manufacturing
3118	Bakeries and tortilla manufacturing	3399	Other miscellaneous manufacturing
3119	Other food manufacturing	3222	Converted paper product manufacturing
3323	Architectural and structural metals manufacturing	3327	Machine shops, turned product, and screw, nut, and bolt manufacturing
3332	Industrial machinery manufacturing	3119	Other food manufacturing
3222	Converted paper product manufacturing	3323	Architectural and structural metals manufacturing
3273	Cement and concrete product manufacturing	3118	Bakeries and tortilla manufacturing
3115	Dairy product manufacturing	3273	Cement and concrete product manufacturing
3329	Other fabricated metal product manufacturing	3362	Motor vehicle body and trailer manufacturing
3371	Household and institutional furniture and kitchen cabinet manufacturing	3116	Meat product manufacturing
3359	Other electrical equipment and component manufacturing	3371	Household and institutional furniture and kitchen cabinet manufacturing
3256	Soap, cleaning compound, and toilet preparation manufacturing	3359	Other electrical equipment and component manufacturing
3339	Other general-purpose machinery manufacturing	3329	Other fabricated metal product manufacturing
3363	Motor vehicle parts manufacturing	3339	Other general-purpose machinery manufacturing
3362	Motor vehicle body and trailer manufacturing	3363	Motor vehicle parts manufacturing

Source: U.S. Census 2003.
Note: Data for NAICS 3121 was withheld to avoid disclosing data of individual companies.

Table 3-1. (continued)

NAICS Code	Value Added	NAICS Code	Total Value of Shipments
3219	Other wood product manufacturing	3256	Soap, cleaning compound, and toilet preparation manufacturing
3116	Meat product manufacturing	3219	Other wood product manufacturing
3333	Commercial and service industry machinery manufacturing	3361	Motor vehicle manufacturing
3255	Paint, coating, and adhesive manufacturing	3255	Paint, coating, and adhesive manufacturing
3372	Office furniture (including fixtures) manufacturing	3333	Commercial and service industry machinery manufacturing
3328	Coating, engraving, heat treating, and allied activities	3372	Office furniture (including fixtures) manufacturing
3251	Basic chemical manufacturing	3111	Animal food manufacturing
3336	Engine, turbine, and power transmission equipment manufacturing	3112	Grain and oilseed milling
3112	Grain and oilseed milling	3324	Boiler, tank, and shipping container manufacturing
3221	Pulp, paper, and paperboard mills	3251	Basic chemical manufacturing
3346	Manufacturing and reproducing magnetic and optical media	3328	Coating, engraving, heat treating, and allied activities
3259	Other chemical product manufacturing	3321	Forging and stamping
3272	Glass and glass product manufacturing	3221	Pulp, paper, and paperboard mills
3321	Forging and stamping	3336	Engine, turbine, and power transmission equipment manufacturing
3351	Electric lighting equipment manufacturing	3351	Electric lighting equipment manufacturing
3335	Metalworking machinery manufacturing	3259	Other chemical product manufacturing
3324	Boiler, tank, and shipping container manufacturing	3272	Glass and glass product manufacturing
3353	Electrical equipment manufacturing	3346	Manufacturing and reproducing magnetic and optical media
3111	Animal food manufacturing	3211	Sawmills and wood preservation
3113	Sugar and confectionery product manufacturing	3113	Sugar and confectionery product manufacturing
3315	Foundries	3353	Electrical equipment manufacturing
3361	Motor vehicle manufacturing	3335	Metalworking machinery manufacturing
3379	Other furniture related product manufacturing	3141	Textile furnishings mills
3366	Ship and boat building	3366	Ship and boat building
3279	Other nonmetallic mineral product manufacturing	3379	Other furniture related product manufacturing
3262	Rubber product manufacturing	3262	Rubber product manufacturing
3334	Ventilation, heating, air-conditioning, and commercial refrigeration equipment manufacturing	3334	Ventilation, heating, air-conditioning, and commercial refrigeration equipment manufacturing

Source: U.S. Census 2003.
Note: Data for NAICS 3121 was withheld to avoid disclosing data of individual companies.

Table 3-1. (continued)

NAICS Code	Value Added	NAICS Code	Total Value of Shipments
3133	Textile and fabric finishing and fabric coating mills	3311	Iron and steel mills and ferroalloy manufacturing
3325	Hardware manufacturing	3279	Other nonmetallic mineral product manufacturing
3141	Textile furnishings mills	3315	Foundries
3252	Resin, synthetic rubber, and artificial and synthetic fibers and filaments manufacturing	3312	Steel product manufacturing from purchased steel
3322	Cutlery and handtool manufacturing	3133	Textile and fabric finishing and fabric coating mills
3312	Steel product manufacturing from purchased steel	3325	Hardware manufacturing
3159	Apparel accessories and other apparel manufacturing	3117	Seafood product preparation and packaging
3149	Other textile product mills	3212	Veneer, plywood, and engineered wood product manufacturing
3211	Sawmills and wood preservation	3313	Alumina and aluminum production and processing
3271	Clay product and refractory manufacturing	3252	Resin, synthetic rubber, and artificial and synthetic fibers and filaments manufacturing
3212	Veneer, plywood, and engineered wood product manufacturing	3159	Apparel accessories and other apparel manufacturing
3343	Audio and video equipment manufacturing	3314	Nonferrous metal (except aluminum) production and processing
3151	Apparel knitting mills	3149	Other textile product mills
3313	Alumina and aluminum production and processing	3151	Apparel knitting mills
3331	Agriculture, construction, and mining machinery manufacturing	3343	Audio and video equipment manufacturing
3314	Nonferrous metal (except aluminum) production and processing	3322	Cutlery and handtool manufacturing
3326	Spring and wire product manufacturing	3331	Agriculture, construction, and mining machinery manufacturing
3253	Pesticide, fertilizer, and other agricultural chemical manufacturing	3271	Clay product and refractory manufacturing
3117	Seafood product preparation and packaging	3253	Pesticide, fertilizer, and other agricultural chemical manufacturing
3311	Iron and steel mills and ferroalloy manufacturing	3326	Spring and wire product manufacturing
3132	Fabric mills	3132	Fabric mills
3169	Other leather and allied product manufacturing	3274	Lime and gypsum product manufacturing
3274	Lime and gypsum product manufacturing	3352	Household appliance manufacturing
3352	Household appliance manufacturing	3169	Other leather and allied product manufacturing
3162	Footwear manufacturing	3162	Footwear manufacturing

Source: U.S. Census 2003.
Note: Data for NAICS 3121 was withheld to avoid disclosing data of individual companies.

Table 3-2. Fifty Products Manufactured in California

Product(s)	NAICS	Manufacturing Sector
Air conditioner*	333415	AC & warm air heating & commercial/industrial refrig equip mfg
Airplane*	336411	Aircraft mfg
Aluminum can	332431	Metal can mfg
Asphalt paving mixtures	324121	Asphalt paving mixture & block mfg
Beef**	311611	Animal (except poultry) slaughtering (and packing)
Bicycle	336991	Motorcycle, bicycle, & parts mfg
Bolts, nuts, screws	33272	Turned product & screw, nut, & bolt mfg
Bread	311812	Commercial bakeries
Canned vegetables	311421	Fruit & vegetable canning
Car*	336111	Automobile mfg
Carpet	31411	Carpet & rug mills
Cellular phone*	33422	Radio & TV broadcasting & wireless communications equip mfg
Cement, hydraulic	32731	Cement mfg
Cheese**	311513	Cheese mfg
Commercial refrigerator*	333415	AC & warm air heating & commercial/industrial refrig equip mfg
Cordless telephone*	33421	Telephone apparatus mfg
Corrugated cardboard box	322211	Corrugated & solid fiber box mfg
Deodorant	32562	Toilet preparation mfg
Dress	315233	Women's & girls' cut & sew dress mfg
Flyer/coupon book	323110	Commercial lithographic printing
Gas stove/range*	335221	Household cooking appliance mfg
Gasoline	32411	Petroleum refineries
Golf club	33992	Sporting & athletic goods mfg
Home blood pressure monitor*	339112	Surgical & medical instrument mfg
Metal window**	332321	Metal window & door mfg
Microwave oven*	335221	Household cooking appliance mfg
Milk**	311511	Fluid milk mfg
Motor oil	324191	Petroleum lubricating oil & grease mfg
Newspapers	511110	Newspaper publishers
OTC drugs	325412	Pharmaceutical preparation mfg
Paint	32551	Paint & coating mfg
Pallets	32192	Wood container & pallet mfg
Personal computer*	334111	Electronic computer mfg
Plastic bag	326111	Unsupported plastics bag mfg
Plastic bottle	32616	Plastics bottle mfg
Plastic cup	326199	All other plastics product mfg
Printed circuit board*	334418	Printed circuit assembly (electronic assembly) mfg

* direct consumers of energy in use phase; ** indirect consumers of energy in use phase

Table 3-2. (continued)

Product(s)	NAICS	Manufacturing Sector
Ready-mix concrete	32732	Ready-mix concrete mfg
Recording paper roll	322299	All other converted paper product mfg
Scanner*	334119	Other computer peripheral equipment mfg
Semiconductor chip*	334413	Semiconductor & related device mfg
Semiconductor equipment*	333295	Semiconductor machinery mfg
Shoe box	322212	Folding paperboard box mfg
Soap	3256	Soap, cleaning compound, & toilet preparation mfg
Soft drink**	312111	Soft drink mfg
Tape storage drive*	334112	Computer storage device mfg
Tires	326211	Tire mfg (except retreading)
Water pump*	333911	Pump & pumping equipment mfg
Wine	31213	Wineries
Wooden table	337122	Nonupholstered wood household furniture mfg

* direct consumers of energy in use phase; ** indirect consumers of energy in use phase

3.2. Estimation of Product Life-Cycle GHG Emissions

Table 3-3 provides LBNL's estimates of product life-cycle GHG emissions for the 50 selected products. This table lists the products, the unit used, and the average price per unit in both 2004 and 1997 dollars. The table then provides the results of LBNL's calculations of GHG emissions from the manufacturing, use, and end-of-life phases for each product. The far right columns of the table provide total life-cycle energy use and GHG emissions per unit for each product.

3.2.1. GHG Emissions from Product Manufacturing

GHG emissions associated with the manufacture of each product were calculated using information from Carnegie Mellon's EIO-LCA database (CMU-GDI 2004).[5] The EIO-LCA database provides energy use and GHG emissions per dollar of economic activity for 491 U.S. commodity sectors developed by the U.S. Department of Commerce. Values for all manufacturing activities related to each commodity sector are provided. For example, for fluid milk manufacturing, the EIO-LCA database provides energy use and GHG emissions values for not only fluid milk manufacturing itself, but the related activities such as power generation and supply, nitrogenous fertilizer manufacturing, phosphatic fertilizer

[5] This database calculates GHG emissions per dollar from producing commodities or services in 491 different economic sectors in the United States, providing information on the relative impact of different types of products, materials, services, or industries with respect to resource use and emissions throughout the U.S. The entire supply chain of requirements is included, so that the effects of producing a $20,000 motor vehicle would include not only the impacts of final assembly, but also the impact from mining of metals, making electronic parts, forming windows, etc. that are needed for parts to build the car. This analysis is a form of life-cycle assessment based upon an economic input-output model of the United States, publicly available data and linear algebra calculation methods (CMU-GDI 2004).

manufacturing, grain farming, all other crop farming, natural gas distribution, real estate, petroleum refineries, and rail transportation. The limitations of the EIO-LCA method include the following: (1) its reported energy consumption data are from 1997 and will therefore not capture any sector-specific energy efficiency improvements made since that time, (2) it relies on highly aggregated data that might be representative of industrial sectors as a whole but may not accurately reflect the operating practices of any single supply chain, and (3) its reported energy consumption data represent the average energy consumed by all reporting U.S. manufacturers within a given sector and are therefore not fully representative of California's electricity mix. Despite these limitations, the EIO-LCA method is still the best available LCA methodology in the public domain for including the supply chain environmental impacts of product manufacture in the United States. Furthermore, because the purpose of the 50-product analysis is to provide only a rough estimate of the GHG emissions associated with each product, the uncertainty introduced by the EIO-LCA method is acceptable, given the preliminary nature of this analysis.

For each product, the 1997 consumer price per unit was multiplied by the EIO-LCA energy intensity value and the EIO-LCA GHG emissions intensity value to generate the total energy use and GHG emissions per unit.

As shown in Table 3-3, total energy use and GHG emissions per product varies widely for the 50 products evaluated, ranging from a low of 1 megajoule (MJ) and 0.1 kilograms of carbon dioxide equivalent (kgCO_2e) for manufacture of an aluminum can to a high of 185 million MJ and 17 million kgCO_2e for manufacture of an airplane.[6] Other products with high energy consumption and GHG emissions during the manufacturing phase include semiconductor equipment,[7] cars, large industrial water pumps,[8] asphalt paving mixtures, hydraulic cement, commercial refrigerators, tape storage drives, and PCs.

3.2.2. GHG Emissions from Product Use

Twenty of the 50 products evaluated in this project use energy, either directly or indirectly, during the product use phase and thus produce energy-related GHG emissions. Those products that use energy directly are PCs and related computer equipment, telephones, airplanes, cars, large industrial water pumps, air conditioners, semiconductor process equipment, refrigerators, blood pressure monitors, stoves, and microwave ovens. Of these, Table 3-3 shows that the largest energy consumers are airplanes, water pumps, semiconductor processing machines, cars, commercial refrigerators, gas stoves, and air

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[6] It must be noted that the per-product GHG emissions of the 50 products selected for this analysis cannot be compared directly. Each product provides a significantly different service to end users and thus the "functional unit" (i.e., the normalization basis in LCA) will vary from product to product. A revised product LCA approach, wherein the total annual life-cycle GHG emissions of each product in California are estimated to provide a normalized basis for comparison, is discussed in Section 5 of this report.

[7] The significant energy required to manufacture the semiconductor equipment is possibly due to the energy-intensive precision manufacturing processes necessary for such equipment.

[8] A material-intensive 100 hp water pump is assumed.

conditioners. Those products that use energy indirectly are milk, cheese, beef, soft drinks, and metal windows. Of these, metal windows are by far the largest indirect energy consumer. Where possible, LBNL obtained estimates for the useful life of each energy-consuming product from the published literature.[9] When such data were not available, LBNL estimated the useful life of energy-consuming products based on experience and expert advice.

3.2.3. GHG Emissions from Product End-of-Life Phase

GHG emissions from the product end-of-life phase were also calculated for each of the 50 products. As shown in Table 3-3, end-of-life phase energy use and GHG emissions are highest for products such as airplanes, asphalt paving mixtures, ready-mix concrete, and hydraulic cement, due to the large mass fraction of these products that was estimated to be disposed of via landfill. The methodology and assumptions for calculating end-of-life-phase energy consumption and GHG emissions are discussed in Section 2.2.

3.2.4. Total GHG Emissions from 50 Products Produced in California

The last column in Table 3-3 sums the total GHG emissions arising from the product manufacturing, use, and end-of-life phases for each of the 50 selected products. The total per-product life-cycle GHG emissions estimated for each product are presented graphically in Figure 3-1.

The data in Table 3-3 indicate the relative life-cycle GHG emissions of 50 major products manufactured in California. However, for many of these products it is likely that not all manufacturing operations occur within the geographical boundaries of California. For example, although the final assembly of aircraft occurs in California, the myriad raw materials and components contained in an aircraft might be produced using a vast network of domestic and international facilities. The same is true for other complex products included in the 50-product analysis, including cars, PCs, scanners, and cellular phones. Without accurate data on the supply chain structure of these 50 products— which are rarely available in the public domain—it is difficult to determine exactly what percentage of manufacturing stage GHG emissions are attributable to California for each product.[10] Similarly, at the product use phase, a significant portion of the life-cycle GHG emissions of airplanes may occur outside of California.

However, although many of the products in Table 3-3 might not be manufactured entirely in California, it is possible that the design and manufacturing decisions behind these products—which can heavily influence a product's life-cycle GHG emissions—are made by California-based businesses. Because GHG emissions have global climate implications, it is the amount of GHG emissions associated with each product—not the exact source location—

[9] Useful life data sources: airplane (U.S. FAA 2004), car (Davis and Diegel 2003), commercial refrigerator (Westphalen et al. 1996), gas stove and range and microwave oven (U.S. DOE 1998), tape storage drive (Matthews 2002).

[10] The issue of determining California-specific GHG emissions for a complex product is explored further in the case study for PCs in Section 3.3.1 and in Section 5, which presents a revised methodology for conducting the 50-product LCA that quantifies in-state GHG emissions.

that is truly important. Thus, the data in Table 3-3 help illuminate California's role in global GHG emissions, whether these emissions occur in-state or elsewhere. These data can therefore help highlight specific products for which proactive design and manufacturing changes by California businesses might lead to significant global GHG reductions.[11]

Additionally, the data in Table 3-3 could be coupled with data on the annual production and consumption volumes of each product in California to provide an approximate "footprint" of the life-cycle GHG emissions of the 50 products.[12] Such an analysis would be useful in identifying the most important products and product-specific life-cycle stages for GHG mitigation initiatives in California. The data in Table 3-3 therefore provide a crucial first step toward more comprehensive analyses.

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[11] There are many strategies for reducing the life-cycle GHG emissions of products through proactive design and manufacturing decisions that have emerged from the nascent field of "green design and manufacturing." Strategies include: reducing product mass, designing for energy efficiency, designing for recycling, minimizing manufacturing complexity, choosing bulk materials with low embodied energy, and utilizing energy-efficient manufacturing methods (Lewis et al. 2001; Graedel and Allenby 2002).

[12] A revised product LCA methodology, which estimates a product's GHG "footprint" based on annual production, consumption, and disposal data, is presented in Section 5.

Table 3-3. Product Life-Cycle GHG Emissions for 50 Products Manufactured in California

		Avg Price ($/unit)		Manufacturing Phase				Use Phase				End-of-Life Phase		Total	
Product(s)	Unit	2004	1997	Energy Intensity (MJ/$)	Total Energy (MJ/unit)	GHG Emissions Intensity (kg CO_2e/$)	Total GHG Emissions (kg CO_2e/unit)	Energy Intensity (MJ/year)	Unit Lifetime (years)	Total Energy Usage (MJ/unit)	Total GHG Emissions (kg CO_2e/unit)	Total Energy Usage (KJ/unit)	Total GHG Emissions (kg CO_2e/unit)	Total Energy Usage (MJ/unit)	Total GHG Emissions (kg CO_2e/unit)
Airplane*	product	36,000,000.00	30,799,574.00	6.0	185,000,000	0.56	17,000,000	18,600,000	20	370,000,000	27,000,000	990,000	81	560,000,000	44,000,000
Water pump*	product	12,000.00	10,266.52	7.2	73,600	0.65	6,600	164,000	20	33,000,000	3,600,000	14,000	1.1	33,000,000	3,600,000
Semiconductor equipment*	product	2,000,000.00	1,711,087.44	7.1	12,100,000	0.62	1,100,000	330,000	5	1,700,000	180,000	29,000	2.3	14,000,000	1,200,000
Car*	product	20,000.00	17,110.87	8.6	148,000	0.73	12,000	69,400	17	1,200,000	81,000	28,000	2.3	1,300,000	94,000
Commercial refrigerator*	product	1,488.00	1,273.05	8.2	10,500	0.76	970	15,500	10	160,000	17,000	13,000	1.1	170,000	18,000
Gas stove/range*	product	899.00	769.13	9.1	6,990	0.76	580	6,750	19	130,000	7,200	13,000	1.1	140,000	7,800
Air conditioner*	product	135.00	115.50	8.2	948	0.76	88	3,800	12	46,000	5,000	5,800	0.47	47,000	5,100
Metal window**	product	103.00	88.12	10.9	960	0.89	79	767	30	23,000	1,500	2,800	0.23	24,000	1,600
Tape storage drive*	product	2,195.00	1,877.92	5.3	9,910	0.44	820	882	5	4,400	490	3,400	0.28	14,000	1,300
PC*	product	1,500.00	1,283.32	4.5	5,820	0.50	640	1,070	4	4,300	470	13,000	1.1	10,000	1,100
Cement, hydraulic	tonne	85.00	72.72	46.7	3,400	14.35	1,000					290,000	23	3,700	1,100
Asphalt paving mixtures	tonne	186.00	159.13	73.1	11,700	6.58	1,000					250,000	20	12,000	1,100
Microwave oven*	product	80.00	68.44	9.1	621	0.76	52	515	10	5,200	570	4,100	0.34	5,800	620
Wooden table	product	549.00	469.69	7.9	3,730	0.67	310					-6,500	73	3,700	390
Semiconductor chip*	product	250.00	213.89	5.9	1,260	0.51	180	165	4	660	73	5.1	<0.01	1,900	250
Ready-mix concrete	tonne	75.00	64.17	19.6	1,260	2.27	150					290,000	23	1,500	170
Scanner*	product	133.00	113.79	4.5	514	0.48	54	133	5	670	73	2,100	0.17	1,200	130
Printed circuit board*	product	160.00	136.89	4.5	618	0.48	65	118	4	470	52	260	0.02	1,100	120
Tires	product	95.00	81.28	16.7	1,360	1.20	97					3,600	0.29	1,400	98
Bicycle	product	250.00	213.89	4.4	949	0.45	96					800	0.07	950	96
Golf club	product	135.00	115.50	9.0	1,040	0.80	92					510	0.04	1,000	93
Cordless telephone*	product	55.00	47.05	3.6	167	0.34	16	151	3	450	50	530	0.04	620	66
Cellular phone*	product	50.00	42.78	4.4	189	0.39	17	65.2	3	190	21	260	0.02	380	38
Home blood pressure monitor*	product	80.00	68.44	4.7	321	0.46	32	2.35	10	21	2	570	0.05	340	34
Motor oil	gallon	7.49	6.41	63.2	405	4.39	28					940	0.08	410	28
Dress	product	35.00	29.94	9.0	269	0.79	24					96	<0.01	270	24
Paint	gallon	20.00	17.11	17.7	303	1.34	23					720	0.06	300	23
Wine	liter	15.00	12.83	8.3	107	1.06	14					130	0.01	110	14
Milk**	gallon	4.25	3.64	13.5	49	2.98	11	209	0.03	5.7	1			55	11
Pallets	product	8.00	6.84	8.3	57	0.71	5.0					-540	6	56	11
Beef**	lb	3.49	2.99	16.8	50	3.36	10	151	0.03	4.1	0.5	6	0.05	54	11
Cheese**	12 oz	3.35	2.87	13.6	39	3.05	9.0	25.1	0.03	0.7	0.1	5.4	0.04	40	9.1
Carpet	sq yd	7.65	6.54	15.5	101	1.28	8.0					270	0.02	100	8.0

Table 3-3. (continued)

Product(s)	Unit	Avg Price ($/unit) 2004	Avg Price ($/unit) 1997	Manufacturing Phase				Use Phase				End-of-Life Phase		Total	
				Energy Intensity (MJ/$)	Total Energy (MJ/unit)	GHG Emissions Intensity (kg CO_2e/$)	Total GHG Emissions (kg CO_2e/unit)	Energy Intensity (MJ/year)	Unit Lifetime (years)	Total Energy Usage (MJ/unit)	Total GHG Emissions (kg CO_2e/unit)	Total Energy Usage (KJ/unit)	Total GHG Emissions (kg CO_2e/unit)	Total Energy Usage (MJ/unit)	Total GHG Emissions (kg CO_2e/unit)
Over-the-counter drugs	box or bottle	15.00	12.83	6.4	82	0.53	7.0					55	<0.01	82	7.0
Gasoline	Gallon	2.25	1.92	25.8	50	3.07	5.9							50	5.9
Soap	product	5.99	5.12	11.6	60	0.92	4.7							60	4.7
Bread	product	3.00	2.57	8.2	21	0.83	2.1					<1	0.04	21	2.1
Canned vegetables	can	1.99	1.70	12.0	20	1.24	2.1					26	0.01	20	2.1
Deodorant	product	4.00	3.42	6.2	21	0.54	1.8					36	<0.01	21	1.8
Corrugated cardboard box	product	0.90	0.77	14.3	11	1.14	0.9					-15	0.17	11	1.1
Soft drink**	product	0.99	0.85	9.4	8	0.86	0.7	33.1	0.05	1.8	0.2	150	0.01	10	0.9
Bolts, nuts, screws	product	1.85	1.58	5.9	9	0.53	0.8					84	<0.01	9.1	0.8
Newspapers	product	0.50	0.43	4.6	2	0.37	0.2					-55	0.63	1.9	0.8
Plastic cup	product	0.79	0.68	10.4	7	0.90	0.6					18	<0.01	7	0.6
Plastic bottle	product	0.50	0.43	13.8	6	1.15	0.5					18	<0.01	6	0.5
Recording paper roll	product	0.40	0.34	12.4	4	0.97	0.3					-14	0.16	4	0.5
Shoe box	product	0.32	0.27	14.3	4	1.14	0.3					-15	0.16	4	0.5
Flyer/coupon book	product	0.55	0.47	8.2	4	0.71	0.3					29	<0.01	4	0.3
Plastic bag	product	0.21	0.18	13.5	2	1.12	0.2					5	<0.01	2	0.2
Aluminum can	product	0.09	0.08	16.3	1	1.35	0.1					45	0.04	1	0.1

* direct consumers of energy in use phase; ** indirect consumers of energy in use phase

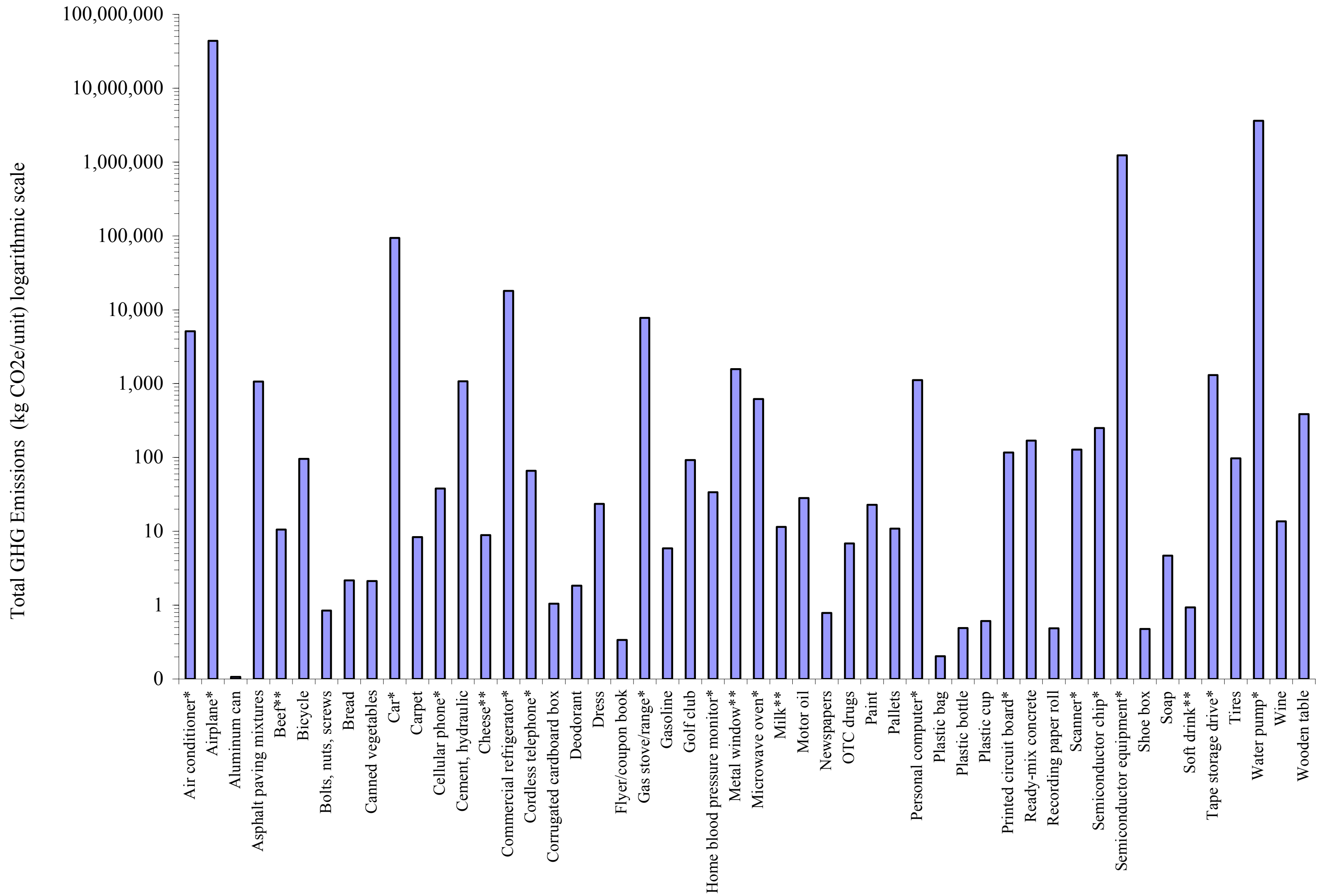

Figure 3-1. Total Life-Cycle GHG Emissions per Product for 50 Products Manufactured in California

3.3. Case Studies to Identify Product-Specific Opportunities for Life-Cycle GHG Emission Reductions

LBNL performed two case studies—one on PCs and one on cement and concrete—to develop detailed estimates of product-specific opportunities for life-cycle GHG mitigation in California. The two particular case studies were chosen for several important reasons. First, PCs and cement and concrete are extremely important products to California from both an economic and environmental perspective. Second, the availability of published data on the life cycle impacts of PCs and cement and concrete made a detailed analysis more feasible than for most of the other products on the LBNL list. Third, LBNL has established contacts in the PC and cement and concrete industries, which allowed for valuable industry feedback on the case study results. Lastly, LBNL has prior experience in the environmental analysis of the products in the two case studies—for example, LBNL has published extensive work on energy efficiency in the cement industry (Worrell et al. 2001; Coito et al. 2005)—making the two case studies feasible within the allowable timeframe and budget of this study.

3.3.1. Personal Computers

Since its debut on the marketplace in the late 1970s, the PC has become one of the most ubiquitous and indispensable products of the modern age, in households and businesses alike. In 2000, the total stock of desktop and notebook PCs in the United States was estimated at over 130 million units (Kawamoto et al. 2001), an installed base rivaling that of passenger vehicles. This total stock is expected to grow significantly as sales of PCs in the United States have increased dramatically over the last decade—from around 20 million units sold in 1994 to over 55 million units sold in 2004 (Matthews and Matthews 2003; ZDNet 2004)—due in large part to their steadily increasing affordability and to the rapid expansion and widespread use of the Internet. In light of this explosive growth, some analysts predict that the number of PCs installed in the United States may reach as many as 1 billion units by 2010 (IAER 2003).

California is the nation's largest manufacturer of computer equipment and is home to several major U.S. PC companies, including Hewlett-Packard, Apple, and Sun Microsystems. California's importance to the $47 billion per year U.S. computer industry is undeniable: 33% of the nation's value added computer manufacturing operations occur within the state (U.S. Census 2005b). California's "hi-tech" sector, which manufactures PCs and the semiconductors, printed circuit boards, and myriad other electronic components contained within them, employs over 700,000 people and is the second-largest source of employment in the state (CEC 2004).

Fittingly, Californians also consume more PCs than any other U.S. state. In 2001, 7.9 million PCs were installed in California households, nearly twice as many as in Texas, the nation's next largest consumer of PCs (U.S. DOE 2001a). A similar number of PCs is expected to be installed in California's commercial and industrial buildings (Kawamoto et al. 2001).

California's role as the top producer and consumer of PCs in the United States comes at a price, however, when energy consumption and GHG emissions are considered. The research team estimates that over 54 petajoules (PJ) of primary energy are required annually to manufacture PCs in California, representing 2.7% of the total primary energy consumed each year by California's industrial sector (U.S. DOE 2001b). The team also estimates that over

39 PJ of primary energy are consumed each year to power California's PCs during product use, which is 1.7% of the primary electrical energy consumed in California homes and commercial buildings each year (U.S. DOE 2001c, 2001d). Combined estimates for the GHG emissions occurring during PC manufacture and PC use in California each year total nearly 6 $MtCO_2e$, an amount equivalent to roughly 1.5% of California's 1999 statewide net GHG emissions (CEC 2002b).

Furthermore, the state's enormous appetite for PCs inevitably leads to high PC disposal rates: an estimated 10,000 PCs become obsolete in California each day (CAW 2004). To fuel the disposal and demanufacturing processes that handle this continuous stream of "e-waste," the research team estimates that 0.05 PJ of primary energy will be consumed and 4 kilotonnes (kt) CO_2 (1 ktC) will be emitted in California each year. These results are summarized in Table 3-4. The methodology and assumptions behind these results are detailed in the sections that follow.

Table 3-4. Annual Life-Cycle Primary Energy Consumption and GHG Emissions for PCs in California

	Primary Energy	GHG Emissions	
Life-Cycle Phase	**PJ/yr**	**$MtCO_2$/yr**	**MtC/yr**
Production	54.3	4.18	1.14
Use	39.4	1.72	0.47
End-of-Life	0.05	0.004	0.001
Total	**93.75**	**5.90**	**1.61**

The estimates summarized in Table 3-4 underscore the magnitude of the PC's contribution to California's annual energy consumption and GHG emissions. Given California's unique role as both the top producer and the top consumer of PCs in the United States, the potential benefits of life-cycle optimization for PCs are likely to be more compelling for California than for any other state.

3.3.1.1. Product Description

The typical desktop PC is comprised of a control unit, a keyboard, a mouse, and a display. The control unit is the heart of the PC, 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. The display can either be a cathode ray tube (CRT) monitor or a liquid crystal display (LCD). CRT monitors have traditionally dominated the PC market, but LCDs are becoming increasingly common in recent years as they become more affordable.

Despite the ubiquity and importance of PCs in industrialized economies, few data have been published to date that characterize the life-cycle energy consumption and GHG emissions of PC manufacturing, use, and end-of-life treatment. This lack of data is due in large part to the extremely complex nature of the PC production system, which typically involves hundreds of different manufacturing processes, raw materials, and process chemicals and is often dispersed among a vast global network of suppliers.

Table 3-5 summarizes the results of four LCA studies of PC systems that have been reported in the published literature to date. Although these studies are interesting because they shed light on the general breakdown of impacts between life-cycle stages, the results are difficult to compare because each study considers different product systems[13] and therefore arrives at different conclusions. Furthermore, none of these studies is fully transparent in its data sources or assumptions, and all studies report only aggregate data for each life-cycle stage. It is therefore impossible to disaggregate the data to pinpoint the impacts of individual processes and materials or to apply regional-specific emission factors to tailor the results to specific geographic areas such as California. This lack of transparency is a significant limitation for using these data in product life-cycle optimization analyses.

Table 3-5. Summary of Life-Cycle Analysis Data for PCs

Life-Cycle Stage	MCC 1993		Tekawa et al. 1997		Atlantic Consulting 1998		Dreier and Wagner 2000	
	Energy MJ	GHG kg CO_2e	Energy MJ	GHG kg CO_2e	Energy MJ	GHG kg CO_2e	Energy MJ	GHG kg CO_2e
Manufacturing	8,330	N/R	N/R	137	3,630	183	9,527	N/R
Use	32,760	N/R	N/R	590	10,200	448	13,500	N/R
End-of-Life	N/R	N/R	N/R	5	-98	17	-800	N/R
Total	**41,090**	**N/R**	**N/R**	**732**	**13,732**	**648**	**22,227**	**N/R**

N/R = not reported

To estimate the energy consumption and GHG emissions of PC manufacturing, use, and end-of-life treatment in California each year, the research team avoided the limitations of the published LCA data by employing a stage-by-stage approach based on a variety of data sources.

At the PC manufacturing stage, the team estimated the primary energy consumption and GHG emissions associated with the production of a generic PC control unit using LCI data provided by Williams (2003, 2004) as primary data sources. Because not all PC manufacturing operations occur in California, the team estimated California's share of manufacturing stage energy consumption and GHG emissions using macro-level economic and production data for key PC materials, components, and manufacturing processes. The details of this approach are provided in Section 3.3.1.2.1.

At the PC use stage, the research team estimated the electricity consumption of PC control units, CRT monitors, and LCDs in California using power consumption and usage pattern data from Kawamoto et al. (2001), Roberson et al. (2004), and Socolof et al. (2001). The team estimated the GHG emissions generated at the PC use stage by applying a California-specific

[13] For example, Drier and Wagner (2000) include a printer in their system description and also include the primary energy necessary for paper and toner ink production in their use stage energy calculations; Tekawa et al. (1997) do not include packaging; MCC (1993) excludes the computer housing, disk drives, power supply, keyboard, and mouse from its system definition and also assumes an intense usage scenario (24 hours per day at full power for four years).

CO_2 emission factor for electricity consumption from Marnay et al. (2002). The details of this approach are provided in Section 3.3.1.2.2.

At the end-of-life stage, the research team estimated the energy consumption and GHG emissions associated with disposing of PCs in California via landfill using LCI data from Franklin Associates (1994) and McDougall et al. (2001). The team estimated the energy consumption and GHG emissions of demanufacturing PCs for recycling in California from publicly available facility data (Fujitsu-Siemens 2001). Researchers estimated the "credits" associated with recycling the bulk materials in PC control units, CRT monitors, and LCDs using several publicly-available LCI data sources (ETH-ESU 1996; BUWAL 1998). Although the use of recycling "credits" is somewhat controversial in LCA (Boustead 2001), they serve as a convenient measure of the potential systems-level savings associated with raw materials recycling. Recycling "credits" are therefore used here as a simplifying assumption to avoid the more complex practice of allocating environmental savings across multiple (and sometimes significantly different) product life cycles. The details of this approach are provided in Section 3.3.1.2.3.

3.3.1.2. Analysis of GHG Emission Reduction Opportunities

3.3.1.2.1. Product Manufacturing Stage

Personal computers are manufactured using a vast global network of production facilities, only a fraction of which are located within the geographical boundaries of California. Although California's Silicon Valley is the birthplace of the PC and was once a global manufacturing leader for PCs and their myriad components, the outsourcing of computer manufacturing operations to China, Taiwan, and Southeast Asia over the last two decades has steadily reduced California's leadership role in the global PC manufacturing industry. However, key PC manufacturing operations are still quite active in California, including the manufacture of semiconductors and printed circuit boards (PCBs) and the assembly of finished components into final PC products. Furthermore, the massive volume of the global PC industry—169 million PCs were produced worldwide in 2003 (Gartner Dataquest 2004)—means that despite its diminished role in the global PC industry, California's annual energy consumption and GHG emissions from PC manufacturing operations are still significant.

To estimate the annual energy consumption and GHG emissions associated with PC manufacturing operations in California, the research team focused the analysis on the manufacture of PC control units. The analysis assumed that CRT monitors and LCDs are manufactured entirely overseas based on recent display market data[14] and therefore that only the manufacture of PC control units is relevant to California. It was also assumed that the energy consumption and GHG emissions of input device manufacturing (i.e., keyboard and

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[14] Williams (2003) notes that 80% of CRTs are produced in East Asia (excluding Japan). The U.S. EPA has reported that only roughly 2% of CRT monitors were produced in North America in 1998 (U.S. EPA 1998), and it appears that since then, several of the remaining U.S. CRT manufacturers have either scaled back or ceased production (Hachman 2003; Grahl 2004). South Korea produces 40% of global LCDs, followed by Japan (39%) and Taiwan (21%) (Gerardino 2001).

mouse) are minor compared to control unit manufacturing, and therefore these components were excluded from the analysis.[15]

To estimate the annual energy consumption and GHG emissions associated with PC control unit manufacturing in California, this study employed a three-stage approach. First, the research team estimated the percentage of global production that occurs in California for each of the key bulk materials, specialty materials, and manufacturing operations necessary to manufacture generic PC control units. Second, the team estimated the life-cycle energy consumption and GHG emissions associated with each bulk material, specialty material, and manufacturing process for a single generic PC control unit. Third, the team multiplied California's allocated share of the energy consumption and GHG emissions for each bulk material, specialty material, and manufacturing process by the global annual production volume of PCs (169 million) to arrive at estimates for the total annual energy consumption and GHG emissions of California's PC control unit manufacturing activities.

The research team estimated California's allocated share of global control unit production using macro-level economic and production data. Table 3-6 provides a summary of this allocation procedure. The specific bulk materials, specialty materials, and manufacturing operations listed in Table 3-6 are based on LCI data for the production of a generic PC control unit available in the published literature (Williams 2003, 2004). The allocation procedure was carried out in two phases. First, global economic and production data were used to estimate the percentage of annual production that occurs within the United States for each bulk material, specialty material, and manufacturing process. Next, the percentage of total U.S. production that occurs within California for each material/process was estimated using manufacturing valued added data for each material/process sector from the U.S. Economic Census. The resulting geographic allocation factors listed in Table 3-6 estimate the percentage of annual energy consumption and GHG emissions that are attributable to California for each material/process in the manufacture of PC control units. For illustrative purposes, non-California energy consumption and GHG emissions occurring during PC manufacture are allocated to the rest of the United States (excepting California) and to the rest of the world for each material/process in a similar fashion, via the "U.S." and "international" allocation factors listed in Table 3-6.

Although this allocation method is somewhat crude, further accuracy is precluded by the general lack of publicly available data on the production characteristics (e.g., facility locations and annual production volumes) of global supply chains for the myriad components contained in a typical PC. Thus, this study's allocation method is a simplified approach that provides only a rough approximation of the energy consumption and GHG emissions of PC control unit manufacturing operations in California.

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[15] Data from Atlantic Consulting (1998) suggest that keyboards generate only 1% of the total life-cycle GHG emissions of a PC, and thus the effect of excluding input devices is assumed to be minor.

Table 3-6. Manufacturing Stage Geographic Allocation Factors for a PC Control Unit

	Global Production[16]			Value Added[17]			Allocation Factors %			Data Source(s) [18]
	World	U.S.	% U.S.	U.S.	CA	% CA				
Bulk Material	**tons**	**tons**		**$**	**$**		**Int'l**	**U.S.**	**CA**	
Steel	9.0E+08	9.2E+07	10.2%	2.5E+10	3.3E+08	1.3	**89.8**	**10.1**	**0.1**	1,2,5
Copper	1.6E+07	1.5E+06	9.7	1.4E+09	0.0E+00	0.0	**90.3**	**9.7**	**0.0**	1,2,5
Aluminum	2.6E+07	5.6E+06	21.8	3.2E+09	9.1E+07	2.8	**78.2**	**21.2**	**0.6**	1,2,5
Plastics/Epoxy	1.7E+08	4.8E+07	28.6	1.8E+10	3.1E+08	1.7	**71.4**	**28.1**	**0.5**	3,4,6
Tin	2.8E+05	6.4E+03	2.3	2.6E+09	2.3E+08	8.9	**97.7**	**2.3**	**0.0**	1,5,7
Lead	6.4E+06	1.4E+06	21.6				**78.4**	**19.7**	**1.9**	1,5,7
Nickel	1.3E+06	2.3E+05	17.1				**82.9**	**17.1**	**0.0**	1,5,7
Silver	2.0E+04	3.6E+03	18.1				**82.0**	**16.5**	**1.6**	1,5,7
Gold	2.6E+03	2.7E+02	10.7				**89.3**	**9.8**	**1.0**	1,5,7
Specialty Materials	**$**	**$**		**$**	**$**					
Silicon wafers	--	--	38	--	--	2.0	**61.8**	**37.5**	**0.8**	8
Specialized materials	--	--	34	7.1E+09	4.9E+08	6.9	**65.7**	**31.9**	**2.4**	6,9
Manufacturing Processes	**$**	**$**		**$**	**$**					
Semiconductors	1.4E+11	5.4E+10	38	4.7E+10	9.1E+09	19.2	**61.8**	**30.9**	**7.3**	6,10
PCBs	4.3E+10	1.1E+10	26	1.0E+10	2.1E+09	21.2	**74.5**	**20.1**	**5.4**	6,11
PC Assembly	1.7E+11	3.5E+10	21	2.1E+10	6.9E+09	33.2	**78.9**	**14.1**	**7.0**	6,12

For a description of key assumptions for this table, see Appendix A.1.

Table 3-7 lists the research team's estimates for the total primary energy and GHG emissions associated with the manufacture of a single generic PC control unit.

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[16] Global production data are from 2003 for plastics and epoxy, from 2002 for all bulk materials (except plastics and epoxy), semiconductor manufacturing, and PC assembly and from 2000 for PCB manufacturing.

[17] Value added data are from 1997 for all bulk materials (except plastics and epoxy) and from 2002 for plastics and epoxy, specialized materials, semiconductor manufacturing, PCB manufacturing, and PC assembly.

[18] Data sources: (1) USGS (2003), (2) U.S. Census (2000a), (3) APME (2004), (4) APC (2004), (5) U.S. Census (2000b), (6) U.S. Census (2005b), (7) USGS (2004), (8) IC Knowledge (2004), (9) Williams (2003), (10) SIA (2004), (11) Custer (2001), (12) ElectronicsNews (2003).

Table 3-7. Total Primary Energy and GHG Emissions of PC Control Unit Manufacturing

	Amount contained (g)	Main use in control unit	Primary energy (MJ)	GHG (kg CO_2e)	Additional data sources
Bulk materials					
Steel	6050	Chassis	226.3	13.25	ETH-ESU 1996
Copper	670	Wires, PCBs	65.3	3.66	ETH-ESU 1996
Aluminum	440	Drives, PCBs	91.9	4.64	ETH-ESU 1996
Plastics	650	Chassis, HDD	27.6[19]	1.97	Boustead 1999
Epoxy	1040	PCBs	101.9	7.13	Boustead 1999
Tin	47	Solder	11.0	N/A	
Lead	27	Solder	0.5	0.03	ETH-ESU 1996
Nickel	18	Disk Drive	3.4	0.27	ETH-ESU 1996
Silver	1.4	PCBs	2.3	N/A	
Gold	0.36	PCBs	30	N/A	
		Subtotal	**560.2**	**30.95**	
Specialty materials					
Silicon wafers	--	Chips	593.2	35.93	
Specialized materials	--	Chips, PCBs	475.0	40.00	CMU-GDI 2004
		Subtotal	**1,068.2**	**75.93**	
Manufacturing					
Semiconductors	--	Chips	3,252.8	195.05	
PFC adjustment[20]	--	--	--	70.42	US EPA 2002
PCBs	--	Motherboard	453.3	29.58	
Final PC assembly	--	--	589.3	35.02	Williams 2004
		Subtotal	**4,295.4**	**330.08**	
		Total for control unit	**5,923.8**	**436.96**	

Note: Adapted from Williams (2003). See Appendix A, Section A.1, for a detailed description of the data sources and calculation methods used to develop this table.

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[19] The LCI energy data sources employed in this report for plastics and epoxy resins include both primary production energy (i.e., the energy required to produce the plastic) and feedstock energy (i.e., the energetic value of the plastic material itself). The data in Table 3-7 for plastics and epoxy resins account only for production energy.

[20] The "PFC adjustment" in Table 3-7 is added to account for process emissions of perfluorocompounds (PFCs), which are potent global warming gases widely used by the semiconductor industry in etching, cleaning, and heat transfer applications (US EPA 2004a). No publicly available data on the emissions of PFCs per wafer were found, but qualitative information suggests that the global warming potential of PFC emissions from semiconductor manufacturing are on the same order as GHG emissions from electricity consumption (Williams 2000). The research team estimated a "PFC adjustment" in Table 3-7 of 70.4 $kgCO_2e$ per control unit to account for these emissions. For further details on this calculation, see Appendix A, Section A.1.

The research team's estimates for the annual primary energy consumption and GHG emissions attributable to California for PC control unit manufacturing are listed in Table 3-8. The estimates in Table 3-8 were calculated by multiplying the data in Table 3-7 by the geographical allocation factors in Table 3-6 and assuming an annual global PC production volume of 169 million units (the total global shipments of PCs in 2003 as reported by Gartner Dataquest (2004)).

Table 3-8. Geographic Allocations of Annual Primary Energy Consumption and GHG Emissions for PC Control Unit Manufacturing

	Primary energy consumption (PJ/yr)			**GHG emissions ($MtCO_2e/yr$)**		
	CA	**U.S.**	**Int'l**	**CA**	**U.S.**	**Int'l**
Bulk materials						
Steel	0.04	3.85	34.30	0.00	0.23	2.01
Copper	0.00	1.07	9.96	0.00	0.06	0.56
Aluminum	0.01	3.30	12.15	0.00	0.17	0.61
Plastics	0.02	1.30	3.33	0.00	0.09	0.24
Epoxy	0.09	4.84	12.30	0.01	0.34	0.86
Tin	0.00	0.04	1.80	0.00	0.00	0.00
Lead	0.002	0.02	0.06	0.00	0.00	0.00
Nickel	0.00	0.01	0.47	0.00	0.01	0.04
Silver	0.006	0.06	0.32	0.00	0.00	0.00
Gold	0.05	0.50	4.53	0.00	0.00	0.00
Subtotal	**0.22**	**14.99**	**79.22**	**0.01**	**0.90**	**3.71**
Specialized materials						
Silicon wafers	0.80	37.56	61.90	0.05	2.28	3.75
Specialized materials	1.93	25.59	52.70	0.16	2.15	4.44
Subtotal	**2.73**	**63.15**	**114.60**	**0.21**	**4.43**	**8.19**
Manufacturing processes						
Semiconductors	40.10	169.70	339.45	2.40	10.18	20.35
PFC adjustment	0	0	0	0.87	3.67	7.35
PCBs	4.13	15.40	57.02	0.27	1.00	3.72
Final PC assembly	6.96	14.03	78.51	0.41	0.83	4.67
Subtotal	**51.20**	**199.15**	**474.95**	**3.96**	**15.69**	**36.09**
Total for control unit	**54.30**	**277.60**	**669.35**	**4.18**	**21.01**	**48.60**

The estimates in Table 3-8 highlight the significant potential that might exist for energy efficiency improvements and GHG mitigation in PC control unit manufacturing in California (particularly in the production of semiconductors). Two potential measures are:

- **Reducing PFC emissions in the semiconductor fabrication process**. The research team estimates that 21% of the GHG emissions associated with control unit manufacturing in California are due to PFC emissions during semiconductor manufacture. A major voluntary initiative—the EPA's PFC Reduction and Climate Partnership for the

Semiconductor Industry (US EPA 2004a)—is underway in the United States. Participating members of the U.S. semiconductor industry have committed to reducing PFC emissions to 10% below their 1995 baseline level by 2010. If this initiative is successful, PFC emissions from California's semiconductor industry could be reduced by 30% from 2000 levels (assuming full industry participation). The research teams estimates that the potential annual savings in GHG emissions in California would be 0.26 Mt CO_2e/yr (0.07 MtC/yr) if full participation by California-based semiconductor manufacturers were achieved.[21]

- **Improving manufacturing energy efficiency**. The semiconductor manufacturing process is estimated to account for roughly 75% of control unit manufacturing energy and roughly 78% of control unit manufacturing GHG emissions in California each year and is thus a prime candidate for energy-efficiency improvements. The benefits of clean room energy-efficiency measures have been well documented: efficiency improvements to process controls and air handling, ventilation, and cooling systems can reduce clean room energy consumption by 30%–60% (Naughton 2000; CEC 1999). Assuming a conservative 30% improvement in clean room energy efficiency, the annual savings in primary energy consumption and GHG emissions in California would be around 12 PJ and 0.72 Mt CO_2e/yr (0.19 MtC/yr), respectively.

The potential savings described above are summarized in Table 3-9.

Table 3-9. Summary of Potential Annual Savings from Energy Efficiency and GHG Mitigation Opportunities in California for PC Manufacturing

Strategy	Potential Savings		
	Primary Energy (PJ/yr)	GHG Emissions ($MtCO_2e$/yr)	GHG Emissions (MtC/yr)
Reduce PFC emissions	0	0.26	0.07
Clean room energy efficiency	12	0.72	0.19

3.3.1.2.2. Product Use Stage

An estimated 7.9 million PCs were installed in California households in 2001 (U.S. DOE 2001a). A similar number of PCs is expected to be installed in California's commercial and industrial buildings (Kawamoto et al. 2001), bringing the total number of PCs in use in California to roughly 16 million.

[21] No publicly available data were found to estimate the percentage of California-based semiconductor manufacturing facilities that currently participate in the U.S. EPA's PFC Reduction and Climate Partnership for the Semiconductor Industry. A comparison of the participating companies on the Partnership's website (US EPA 2004a) and California-based semiconductor manufacturers listed in the *2004 Directory of California Manufacturers* (MNI 2004) suggests that many California-based facilities do not yet participate. The nationwide participation rate among U.S.-based semiconductor manufacturers has been estimated at 85% (Parkhurst pers. comm. 2005).

To estimate the energy consumed by these 16 million PCs each year, the research team employed the unit energy consumption (UEC) approach of Kawamoto et al. (2001), which calculates the annual electricity consumption of an electronic device based on: (1) a weekly usage pattern, (2) its power consumption in active, low, and off modes, and (3) the average utilization of power management features. The team calculated the UEC for three devices—an average PC control unit, a 17-inch CRT monitor, and a 15-inch LCD—to account for the consumption of energy by both control units and displays in California's PCs. Details of use-stage calculations are provided in Appendix A, Section A.2.

Table 3-10 lists the team's estimates for the total annual use stage primary energy consumption and GHG emissions associated with California's PCs. The use stage electricity consumed by California's PCs is converted to primary energy and GHG emissions using the conversion factors for California provided in Table A-1, in Appendix A.

Table 3-10. Estimated Use Phase Primary Energy Consumption and GHG Emissions of California's PCs

Parameter	Control Unit		CRT Monitor		LCD		Total
	Residential	Commercial	Residential	Commercial	Residential	Commercial	
Devices	8,000,000	8,000,000	6,400,000	6,400,000	1,600,000	1,600,000	
Primary energy (PJ/yr)	3.61	15.60	4.48	13.90	0.49	1.35	**39.4**
GHG emissions ($MtCO_2$/yr)	0.16	0.68	0.19	0.60	0.02	0.06	**1.71**
GHG emissions (MtC/yr)	0.04	0.19	0.05	0.16	0.01	0.02	**0.47**

The estimates in Table 3-10 suggest that commercial PCs are responsible for over 75% of the predicted use stage primary energy consumption and GHG emissions of PCs in California. Commercial PC usage could therefore represent a fruitful area for energy efficiency and GHG mitigation improvements.

The research team considered two major strategies for reducing the use stage energy consumption of PCs: (1) designing control units and displays to consume less energy during operation, and (2) increasing the utilization of power management features for control units and displays by the end user.

The design of energy-efficient PCs depends on both basic technological progress (such as the achievable transistor size of semiconductor chips) and proactive design efforts by computer equipment manufacturers (such as the use of energy-efficient power supplies). The U.S. ENERGY STAR© program has been instrumental in fostering more proactive "design for energy efficiency" efforts by PC manufacturers through its established energy standards for PC control units, monitors, and power supplies (U.S. EPA 2005a). The most energy-efficient control units certified by ENERGY STAR will consume 15 W or less in *low-power* mode (i.e., sleep mode); the most energy-efficient displays certified by ENERGY STAR will consume less than 2 W in *low-power* mode, and less than 1 W in *off* mode.

The extent to which a PC's power management features are utilized in practice depends heavily on the end user. It has been noted that "compared to power managing [control units], monitors are usually simpler, have much more energy savings potential, power manage more reliably, and are less likely to interfere with operation or network connections" (Nordman et al. 1997). This fact is reflected in the estimates for the percentage of control units utilizing power management (25%), versus the percentage of displays utilizing power management (75%), which were culled from recent data in the published literature (Nordman et al. 2000; Roberson et al. 2004).

The research team considered four potential areas for reducing the use stage energy consumption and GHG emissions of California's PCs:

- **Maximizing the operational energy efficiency of California's PC stock**. If all of California's residential and commercial PCs employed the most energy-efficient control units and displays as certified by the U.S. EPA/DOE ENERGY STAR program, the research team estimates that California's PC stock would consume 6% less electricity per year. This savings in electricity would lead to reductions of 2.4 PJ in primary energy and 0.10 $MtCO_2$ (0.03 MtC) in GHG emissions per year.

- **Increasing the utilization of power management features for PC control units**. There appears to be a significant disparity between the utilization of power management features for displays (75%) and the utilization of power management features for control units (25%), as pointed out by Nordman et al. (1997) and Roberson et al. (2004). It has been suggested by Cole (2003) that the difficulties associated with properly configuring power management features are the largest barrier to use stage energy savings for PCs. If these usability difficulties could be overcome such that the power management utilization for control units were harmonized with displays at a rate of 75%, the team estimates that California's PC stock would consume 9% less electricity per year. This savings in electricity would lead to annual savings of 3.7 PJ of primary energy and 0.16 $MtCO_2$ (0.04 MtC) of GHG emissions. Considerable work has been underway at LBNL to develop and promote an industry standard for improving the user-friendliness of PC power management features (LBNL 2005). This work recently culminated in creation of IEEE 1621 (IEEE 2005), an industry-level standard for harmonizing the power modes and power mode indicator lights of electronic products to make power management more intuitive to the user. The IEEE 1621 standard, if adopted on a widespread basis by PC manufacturers, should help to eliminate pervasive power management usability barriers for PCs such as those cited by Cole (2003).

- **Maximizing the utilization of power management features for PC control units and displays**. If 100% of California's residential and commercial PC stock employed power management for both control units and displays, the annual savings in electricity consumption would be substantial. The team estimates that for the case of 100% power management utilization, the electricity consumption of California's PC stock would be reduced by 28%, leading to annual savings of 11 PJ of primary energy and 0.50 $MtCO_2$ (0.13 MtC) of GHG emissions.

- **Switching from CRT monitors to LCDs**. The data in Table 3-10 underscore the fact that LCDs can consume significantly less energy during operation than CRT monitors. If all of California's estimated 12.8 million CRT monitors were replaced by LCDs, the team estimates that California's PC stock would consume 28% less electricity per year, leading to annual savings of 11 PJ of primary energy and 0.50 $MtCO_2$ (0.13 MtC) of GHG emissions.[22]

The estimated savings associated with each of these four measures are summarized in Table 3-11.[23]

Table 3-11. Summary of Potential Annual Savings from Energy Efficiency and GHG Mitigation Opportunities in California for PC Use

Strategy	Potential Savings		
	Primary Energy (PJ/yr)	**GHG Emissions ($MtCO_2e/yr$)**	**GHG Emissions (MtC/yr)**
Maximize PC energy efficiency	2.40	0.10	0.03
Increase control unit power management utilization	3.68	0.16	0.04
Maximize PC power management utilization	10.97	0.47	0.13
Switch from CRTs to LCDs	11.07	0.48	0.13

3.3.1.2.3. Product End-of-Life Phase

An estimated 10,000 computers become obsolete in California every day (CAW 2004), which means that each year over 3.6 million PCs are poised to enter California's solid waste stream. In 2003, California passed landmark legislation—the Electronic Waste Recycling Act of 2003[24]

[22] With respect to use stage energy efficiency, published data have clearly shown that LCDs are preferable to CRT monitors. However, the full range of environmental impacts should be considered (e.g., the use of mercury in LCD backlights) in the switch from CRT monitors to LCDs to ensure that the environmental gains associated with increased energy efficiency are not diminished by unintended environmental costs (e.g., the possibility of increased mercury pollution from LCD manufacture or disposal).

[23] It must be noted that the data in this table are only rough estimates of the potential savings available to California for each measure as they depend heavily on the baseline usage assumptions detailed in Appendix A, Section A.2. To arrive at more precise estimates, more detailed data on the characteristics of California's current PC stock would be required (e.g., total number of PCs, ratio of CRTs to LCDs, extent of power management use, power consumption attributes of various brands and models) to arrive at a more accurate baseline usage scenario. Such accuracy has been precluded in the current analysis, however, due to the general lack of such detailed data in the public domain.

[24] Senate Bill 20/Senate Bill 50 (SB 20 Sher, Chapter 526, Statutes of 2003; SB 50, Sher, Chapter 863, Statutes of 2004).

—to finance the recycling of obsolete CRT monitors, LCDs, CRT televisions, and laptop computers (CIWMB 2005a).[25] Although this legislation will ensure that CRT monitors and LCDs from California's waste PCs are recycled, PC control unit recycling is currently not included in its scope. Assuming each obsolete PC contains a control unit, California is still faced with the challenge of ensuring proper end-of-life treatment for over 3.6 million PC control units each year.

It is estimated that only 8% of PCs are currently recycled in the United States (Matthews and Matthews 2003). At this rate of recycling, over 3.3 million PC control units would be destined for California's landfills each year.

To estimate the primary energy consumption and GHG emissions of California's PCs at the end-of-life phase, this study employed the following approach. First, the research team estimated the annual energy consumption and GHG emissions of landfilling California's non-recycled PC control units using published LCI data on solid waste management processes. Second, the team estimated the energy and GHG emissions required for "demanufacturing" California's recycled CRT monitors, LCDs, and control units to generate bulk materials for recycling. Third, the team estimated the primary energy and GHG emissions "credits" associated with recycling the bulk materials from California's demanufactured CRT monitors, LCDs, and control units. Fourth, the team allocated the calculated bulk materials recycling "credits" to the geographic regions in which virgin material substitutions are likely to occur.

The geographic allocation of recycling "credits" becomes necessary based on the observation that many recyclable materials in California—including those reclaimed from demanufactured e-waste—are likely to be exported for recycling (CIWMB 1996).[26] Thus, the analysis allocates recycling "credits" in the same fashion as it allocates bulk materials production energy and GHG emissions in Section 3.3.1.2.1; both of these allocations are done in an effort to attribute environmental impacts and benefits to the geographic regions in which they are likely to occur. The details of the calculation procedure are provided in Appendix A, Section A.3.

Table 3-12 summarizes the research team's estimates for the total primary energy consumption and GHG emissions associated with landfilling, demanufacturing, and recycling PCs in California each year.

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[25] The Electronic Waste Recycling Act went into effect in California on January 1, 2005.

[26] Similarly, many of the bulk materials contained in new PCs are likely to be produced outside of California, as can be seen in Section 3.3.1.2.1.

Table 3-12. Estimated End-of-Life Phase Primary Energy Consumption and GHG Emissions of California's PCs

End-of-Life Process	Primary Energy (TJ/yr)	GHG Emissions ($ktCO_2$/yr)	GHG Emissions (ktC/yr)
Landfilling	18	1.33	0.36
Demanufacturing	32	2.89	0.79
Recycling	(6)	(0.35)	(0.09)
Total	**44**	**3.87**	**1.06**

This study considers two strategies for reducing the primary energy consumption and GHG emissions of California's PCs at the end-of-life stage:

- **Maximizing PC control unit recycling.** Although California's obsolete CRT monitors and LCDs will be recycled under California's landmark Electronic Waste Recycling Act of 2003, much opportunity still exists to recycle California's PC control units, which are not included in California's recycling legislation. If California were to recycle 100% of its estimated 3.6 million obsolete PC control units each year, the research team estimates that California would save roughly 12 TJ of primary energy and 0.50 $ktCO_2$ (0.14 ktC) annually. These savings are realized through the elimination of control unit landfilling and the corresponding increase in the recycling "credits" that are allocated to California.

- **Upgrading PCs to extend their useful life.** PC upgrading has been suggested as an effective strategy for reducing the environmental impacts associated with a PC's life cycle (Williams and Sasaki 2003). By upgrading a PC when it no longer meets the needs of its user, the purchase of a new PC can be delayed. When fewer new PCs need to be purchased within a given PC stock each year, demand will be reduced on the manufacturing processes necessary to maintain that PC stock. The reduced demand on PC manufacturing processes will in turn lead to annual savings in energy consumption and GHG emissions. The research team considered the case of upgrading 100% of California's 16 million PCs to determine an upper bound estimate on annual primary energy savings. The team estimates that upgrading 100% of California's PCs would lead to primary energy savings of nearly 300 TJ per year and GHG emissions savings of nearly 19 $ktCO_2$ (5.2 ktC) per year. The details of the calculation procedure are provided in Appendix A, Section A.3.

Table 3-13 provides estimates for California's potential annual savings in primary energy and GHG emissions associated with control unit recycling and upgrading.

Table 3-13. Summary of Potential Annual Savings from Energy Efficiency and GHG Mitigation Opportunities in California for End-of-Life PCs

Strategy	Potential Savings		
	Primary Energy (TJ/yr)	**GHG Emissions ($ktCO_2e$/yr)**	**GHG Emissions (ktC/yr)**
Maximize PC control unit recycling	12	0.50	0.14
Upgrade PCs to extend their useful life	288	18.70	5.18

3.3.1.3. Total Potential GHG Emission Reduction Opportunities for PCs

The GHG reduction opportunities for California's PCs introduced in the previous sections are summarized by life-cycle stage in Table 3-14. The percent reduction for each potential opportunity is relative to the total annual PC GHG emissions in California listed in Table 3-4 (5.9 $MtCO_2e$/yr), which includes all stages of the PC life cycle.[27]

Table 3-14. Summary of Potential Opportunities for Reducing GHG Emissions Across the PC Life Cycle in California

Life-Cycle Stage	Opportunity	Potential Life-Cycle GHG Emission Reduction		
		$MtCO_2e$	MtC	% of Total
Production	Reduce PFC emissions	0.26	0.07	4
	Clean room energy efficiency	0.72	0.19	12
Use	Maximize PC energy efficiency	0.10	0.03	2
	Increase control unit power	0.16	0.04	3
	Maximize PC power management	0.47	0.13	8
	Switch from CRTs to LCDs	0.48	0.13	8
End-of-Life	Maximize PC control unit recycling	0.0005	0.0001	0.01
	Upgrade PCs to extend their useful life	0.018	0.005	0.3

3.3.2. Cement and Concrete

In California, cement plants are found in southern California, the San Francisco Bay Area, and a single plant in northern California. In 2002, California produced over 11 million tonnes of cement in eight plants, making California the largest cement-producing state in the United States (USGS 2003). In California, the cement industry directly employs approximately 2,000 workers and has an annual value of shipments of $1 billion. The concrete and ready-mix industries in California together directly employ almost 19,000 employees and have an annual value of shipments of around $4.1 billion (O'Hare pers. comm. 2005).

3.3.2.1. Product Description

Cement is an inorganic, non-metallic substance with hydraulic binding properties and is used as a bonding agent in building materials. It is a fine powder, usually gray in color that consists of a mixture of the hydraulic cement minerals to which one or more forms of calcium

[27] The data summarized in Table 3-14 are based on secondary data from a wide range of sources—including many LCI data for which the uncertainty is not known—as well as many simplifying assumptions. Thus the data in Table 3-14 should be interpreted as more illustrative of savings potentials in California than definitive. In practice, full savings potentials are nearly impossible to realize due to a wide variety of factors, including economic constraints, institutional and organizational barriers, and behavioral inertia. Despite being rough estimates, the data in Table 3-14 are quite useful in indicating the potential order of magnitude of savings associated with each opportunity, as well as its relative effectiveness.

sulfate have been added. Mixed with water it forms a paste, which hardens due to formation of cement mineral hydrates. Cement is the binding agent in *concrete,* which is a combination of cement, mineral aggregates and water. Concrete is a key building material for a variety of applications.

Because of the importance of cement as a construction material, and the geographic abundance of the main raw materials such as limestone, cement is produced in virtually all countries. Cement plants are typically constructed in areas with substantial raw materials deposits (e.g., those that will produce for 50 years or longer). The widespread production of cement is the result of a relatively low price and high density, which limits ground transportation because of relatively high costs. Ground deliveries generally do not exceed distances beyond 150–200 kilometers (roughly 95 to 125 miles). Bulk sea transport is possible, but typically limited.

Clinker is a grey powder (after grinding) that is used as the primary component of cement. Clinker is produced through a controlled high-temperature burn in a kiln of a measured blend of calcareous rocks (usually limestone) and lesser quantities of siliceous, aluminous, and ferrous materials. The kiln feed blend (also called raw meal or raw mix) is adjusted depending on the chemical composition of the raw materials and the type of cement desired. Portland and masonry cements are the chief types produced in the United States.

Cement is used primarily to make concrete. About 75% of the cement is shipped to ready mix concrete producers, while 13% is shipped to concrete product manufacturers, and the remainder is sold to contractors and dealers (Coito 2004). Concrete may contain varying amounts of cement, depending on its application. Typically in making concrete, 10% to 15% cement is combined with 60% to 70% aggregates (gravel or crushed stone and sand) and 15% to 20% water. Concrete is used for many different applications. In California, construction of commercial buildings and road construction are the most important concrete markets.

In producing cement, CO_2 emissions are due to three sources: (1) combustion of fuels (mainly coal) in cement kilns, (2) power generation (onsite and offsite) to provide the power for the plants, and (3) calcination of limestone in the clinker-making process.

Table 3-15 provides life-cycle GHG emissions for cement and concrete manufactured and used in California. In Table 3-3 the average emission intensity for cement making in the United States is estimated to be 1047 $kgCO_2e/t$ cement, including mining and transport of raw materials (estimated at approximately 78 $kgCO_2e/t$).[28] However, the cement industry in California is more energy efficient than the U.S. average, as the penetration of the energy-intensive wet clinker-making[29] process is much higher in the United States and specific

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[28] Direct average emission intensity for the U.S. cement industry in 1997 is estimated at 969 $kgCO_2e/t$ (Worrell and Galitsky 2004), compared to the EIO-LCA value of 1047 $kgCO_2e/t$. LBNL assumes that the difference is due to mining of raw materials offsite (e.g., coal, iron ore) and transport of these materials to the cement plant which are included in the EIO-LCA value but not in the Worrell and Galitsky (2004) value.

[29] Clinker is produced in two major process routes. In the wet process the raw materials are mixed with water to grind the material to raw meal, resulting in higher energy consumption in the kiln as the

emissions are estimated to be 854 $kgCO_2$/t (233 kgC/t) cement. Including similar emissions for raw materials mining and transport, the total intensity would be 932 $kgCO_2e$/t cement, or 11% lower than the average U.S. emission intensity. Focusing just on the cement manufacturing facility, emissions in California are estimated to be 9.6 $MtCO_2$ (2.6 MtC). Carbon dioxide emissions for all aspects of cement manufacturing, including raw materials mining, transport, and all other associated activities, are estimated to be 10.4 Mt CO_2 (2.8 MtC).

Table 3-15. Life-Cycle GHG Emissions for Cement and Concrete in California

Phase	Product	California GHG Emissions (Mt CO_2)	(MtC)
Production	Cement plant	9.6	2.6
	Other cement-related	0.9	0.2
	Total cement	10.4	2.8
	Concrete	1.4	0.4
	Total	11.8	3.2
Use		0.0	0.0
End-of-Life		0.018	0.005
Total		11.8	3.2

In making concrete, electricity is used for mixing, shaping, and curing (for pre-fabricated concrete), while fuel is used in the curing and other processes. Electricity use in concrete production is estimated at 2 kWh/t concrete, and fuel use at 98 MJ/t concrete (Nisbet et al. 2002). In 2001, 11.2 million tonnes of cement were used to produce concrete (USGS 2003). Assuming 14% cement in concrete, the production of concrete in California consumes 160 terawatt-hours (TWh)/year, and 7.8 PJ fuel, resulting in CO_2 emissions of approximately 662 $ktCO_2$ (180 ktC).

Furthermore, energy is used for the mining of the aggregates and sand used in the concrete production. Assuming 86% aggregates, it is assumed that for each tonne of concrete, 14 MJ of diesel is used and 7 kWh/t (Heijningen et al. 1992) is used to mine and separate the sand and gravel. For concrete production in California, the mining and cleaning of the sand and gravel results in consumption of 1.3 PJ of diesel fuel and 620 TWh of electricity. This results in CO_2 emissions of 473 $ktCO_2$ (129 ktC).

The above figures exclude the energy use associated with transporting the raw materials (cement to the ready-mix plant and concrete to the construction site). There are no specific data available on transport distances of aggregates and concrete in California. Based on a European study, the transporting energy use for sand and aggregates is estimated at 42 MJ/t (Heijningen et al. 1992; Nisbet et al., 2002). This figure is relatively low, as transport by ship has been assumed. In California, this energy use is likely to be higher, as road transport is the primary mode of aggregate transport. This study assumes specific energy consumption of

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water needs to be evaporated. Wet kilns are an older process, and no new wet kilns have been built in the United States since the 1970s.

50 MJ/t, which results in the total energy use of 3.4 PJ and emissions of 253 $ktCO_2$ (69 ktC). Total GHG emissions from concrete are estimated to be 1.4 $MtCO_2$ (0.38 MtC).

Energy consumption during the use phase of concrete is assumed to be negligible.

Concrete can be recycled as roadfill and aggregate. The California Integrated Waste Management Board (CIWMB) estimates the total amount of concrete waste at only 400,000 tonnes annually (CIWMB 1999), suggesting that large amounts of concrete are already recycled in California. However, the research team suggests that a more detailed study is needed to analyze concrete construction and demolition waste streams in California to better understand the life cycle and flows of concrete in California. Emissions are mainly attributable to transport of the concrete to the landfilling site.

Total life-cycle GHG emissions in California for cement and concrete, including disposal (see Table 3-3), are estimated to be 11.8 $MtCO_2$ (3.2 MtC).

3.3.2.2. Analysis of GHG Emission Reduction Opportunities

3.3.2.2.1. Manufacturing: Energy Consumption

Energy is consumed in cement manufacturing for the preparation of raw materials, clinker production, and finish grinding. By far the largest proportion of energy consumed in cement manufacture consists of fuel that is used to heat the kiln. Therefore, the greatest gain in reducing energy input and related GHG emissions may come from improved fuel efficiency. The main energy-efficiency opportunities in the kiln are conversion to a more energy-efficient process (i.e., pre-calciner multi-stage pre-heater kiln), optimization of the clinker cooler, improvement of preheating efficiency, improved burners, and process control and management systems. Electricity use can be reduced through improved grinding systems, high-efficiency classifiers, high-efficiency motor systems, and process control systems. Table 3-16 provides a list of energy-efficient practices and technologies for cement production (Worrell and Galitsky 2004).

Several studies have demonstrated the existence of cost-effective potentials for energy efficiency improvement in the cement industry. The technical potential for energy efficiency improvement in California is estimated to be 22%, based on replacing the current equipment with best practice technology (Coito 2004). This would lead to a total emission reduction of 678 $ktCO_2$ (185 ktC) or 9% of the total emissions of the cement industry. In reality, the economic potential will be lower. No detailed studies on the economic potential for energy efficiency improvement in the California cement industry are available.

Table 3-16. Energy-Efficient Practices and Technologies in Cement Production

Raw Materials Preparation	
Efficient transport systems (dry process)	
Slurry blending and homogenization (wet process)	
Raw meal blending systems (dry process)	
Conversion to closed circuit wash mill (wet process)	
High-efficiency roller mills (dry cement)	
High-efficiency classifiers (dry cement)	
Fuel preparation: Roller mills	
Clinker Production (Wet)	**Clinker Production (Dry)**
Energy management and process control	Energy management and process control
Seal replacement	Seal replacement
Kiln combustion system improvements	Kiln combustion system improvements
Kiln shell heat loss reduction	Kiln shell heat loss reduction
Use of waste fuels	Use of waste fuels
Conversion to modern grate cooler	Conversion to modern grate cooler
Refractories	Refractories
Optimize grate coolers	Heat recovery for power generation
Conversion to pre-heater, pre-calciner kilns	Low pressure drop cyclones
Conversion to semi-dry kiln (slurry drier)	Optimize grate coolers
Conversion to semi-wet kiln	Addition of pre-calciner to pre-heater kiln
Efficient kiln drives	Conversion to multi-stage pre-heater kiln
Oxygen enrichment	Efficient kiln drives
	Oxygen enrichment
Finish Grinding	
Energy management and process control	
Improved grinding media (ball mills)	
High-pressure roller press	
High efficiency classifiers	
General Measures	
Preventative maintenance (insulation, compressed air system, maintenance)	
High efficiency motors	
Efficient fans with variable speed drives	
Optimization of compressed air systems	
Efficient lighting	

Source: Worrell and Galitsky (2004)

In addition to energy-efficiency improvement, the use of alternative, or waste-derived, fuels offers an opportunity for GHG emissions reduction from cement manufacture.[30] More than 90% of the energy used in the cement production is from fuels; the remainder is electricity. A significant option for reducing CO_2 emissions is to reduce the carbon content of the fuel. The

[30]Waste-derived fuels include gaseous alternative fuels (e.g., coke oven gas, refinery gas, pyrolysis gas, landfill gas), liquid alternative fuels (e.g., halogen-free spent solvents, mineral oils, distillation residues, hydraulic oils, insulating oils), and solid alternative fuels (e.g., waste wood, dried sewage sludge, plastic, agricultural residues, tires, petroleum coke, and tar).

carbon intensity of the fuels can be reduced by using lower-carbon fuels. Using waste-derived fuels has the dual benefit of both reducing the disposal of waste materials and the use of fossil fuels. Possible disadvantages include the adverse effects on the cement quality and increased emissions of harmful gases. However, most kilns in California are equipped with state-of-the-art flue gas treatment (e.g., baghouse filters) limiting the potential risks of emissions of heavy metals from waste combustion in the cement kiln.

Currently, seven cement plants in California are permitted to use scrap tires as supplemental fuel. In 2004, four of these plants (three plants did not use scrap tires in 2004) used over 71,000 tons of scrap tires (Bennet 2005) to replace coal.

Alternative fuels may contribute to the cement process as an alternative source of energy and as a source of raw material. The mineral portion of a waste can be used as raw material. For example, organic-free mineral wastes can be added to the raw meal. The organic fraction of waste can be used as a source of fuel and may be introduced directly to the burning and/or calcining zone of the kiln. Waste needs to be processed before it can be utilized in the cement kiln. Processing is required to obtain a fuel with constant quality that can easily be fired in the kiln. Fuel qualities of concern include heat content, solid content and volatile matter, viscosity, moisture content, ash constituents, and the levels of various other constituents such as sulfur, nitrogen, metals, biocides, and halogenated organics. Several systems or techniques are developed to utilize waste in the manufacture of cement. The processing system contains transport, storage, processing, and feed systems.

Following the lead taken by the cement industry in Europe, the use of waste fuels has steadily increased in the U.S. cement industry (Worrell and Galitsky 2004). The reduction in CO_2 emissions is determined by the carbon intensity of the waste fuel, the origin of the carbon (i.e., fossil or renewable), the combustion efficiency, and the alternative use of the waste (e.g., landfilling, incineration with or without energy recovery). Most waste in California is landfilled, including "mountains" of tires, while some is incinerated (mostly without energy recovery).[31] Waste tires may ignite and burn unabated, causing not only severe environmental problems, but also emitting CO_2. Although it is theoretically possible to burn a high percentage of waste (e.g., plastics with a mix of other wastes) in a kiln, generally the use of waste fuel will be limited by several factors: (1) the need to maintain perfect clinker quality (and hence the need to control composition), (2) the need to control combustion process conditions, (3) the need to control the moisture content of the waste, and (4) the limitations of existing emission controls (including permitting).

The research team assumed that, on average, a 20% replacement of fossil fuels by waste fuels is possible in kilns in California. Assuming that the carbon in the waste would have been oxidized anyway, the burning of waste will result in a reduction of 616 kt CO_2 (168 kt C).

3.3.2.2.2. Manufacturing: Use of Resources

Limestone is the main natural resource used in the manufacture of clinker and cement. *Portland cement* is the predominant cement type produced and used in California and the

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[31] Unfortunately, no statistical data were available for this study on the ways that tires are processed in California.

United States. Portland cement contains 95% clinker. Clinker is the energy-intensive production step in cement making, consuming virtually all fuels and about a third of the electricity. Also, the calcination of limestone in the clinker-making process leads to additional emissions of CO_2. In the case of California, nearly 60% of the CO_2 emissions from cement manufacture are due to calcination with the remainder from fuel and electricity use (USGS 2003). As a result, replacing clinker by less energy-intensive alternative cementitious materials, including blended cements, ground limestone, and CemStar® materials, can lead to appreciable reductions in CO_2 emissions.

Blended cements, in which cementitious alternatives (such as fly-ash from coal-fired power stations, blast furnace slag from iron production, or pozzolanic materials)[32] are inter-ground with the clinker, can significantly reduce CO_2 emissions from cement manufacturing. The use of blended cements is a particularly attractive efficiency option, because the inter-grinding of clinker with other additives not only allows for a reduction in the energy used (and CO_2 emitted) in clinker production, but results in a reduction in CO_2 emissions from calcination as well. Blended cements demonstrate a higher long-term strength, as well as improved resistance to acids and sulfates, while using waste materials for high-value applications. Short-term strength (measured after less than seven days) may be lower, although cement containing less than 30% additives will generally have setting times comparable to concrete based on portland cement.

Blended cements are very common in Europe and in numerous other countries around the world. Blast furnace and pozzolanic cements account for about 12% of total world cement production, with portland composite cement accounting for about 44% (Cembureau 1997). Blended cements reduce production costs for cement (especially energy costs), expand capacity without extensive capital costs, and reduce emissions from the kiln. In Europe, a common standard (ENV-197-1) has been developed for 25 types of cement (using different compositions for different applications). The European standard allows wider applications of additives.

In the United States, the consumption and production of blended cement is still relatively limited. The most prevalent blending materials available in the United States are fly ash and granulated blast furnace slag. Not all slag and fly ash is suitable for cement production. It is estimated that 68% of the fly ash in the United States conforms to American Society for Testing and Materials (ASTM) Standard C618 (PCA 1997). Currently, only a small part of the blast furnace slag is produced as granulated slag, while the majority is air-cooled. Air-cooled slag cannot be used for cement production. However, investments in slag processing by slag processors and cement companies could increase its production. ASTM Standards exist for different types of blended cements, i.e., C989 (slag cement), C595, and C1157. The EPA has issued procurement guidelines to support the use of blended cement in federal construction projects (U.S. EPA 2000).

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[32] A *pozzolanic material* is a siliceous and aluminous material, which in itself possesses little or no cementitious value, but will, in finely divided form and in the presence of moisture, react with calcium hydroxide to form compounds possessing cementitious properties.

A recent analysis of the U.S. situation cited an existing potential of producing 34 million tons of blended cement in 2000 using both fly ash and blast furnace slag, or 36% of U.S. capacity (PCA 1997). This analysis was based on estimates of the availability of inter-grinding materials and surveying ready-mix companies to estimate feasible market penetration. In California, there is neither iron production nor coal combustion in power plants. However, coal-fired power plants are located nearby in Nevada and Arizona. Hence, for California, the main alternative pozzolanic materials are fly ash from these coal plants and natural pozzolanes.

Blended cement produced would have, on average, a clinker/cement ratio of 65% and would result in a reduction in clinker production of 10.3 million tonnes. The reduction in clinker production corresponds to a specific fuel savings of 1.42 GJ/t. There is an increase in fuel use of 0.09 GJ/t for drying of the blast furnace slags but a corresponding energy savings of 0.2 GJ/t for reducing the need to use energy to bypass kiln exit gases to remove alkali-rich dust. Energy savings are estimated at 9 to 23 kJ/kg per percent bypass (Alsop and Post 1995). The bypass savings accrue because blended cements offer an additional advantage: the inter-ground materials lower alkali-silica reactivity (ASR), thereby allowing a reduction in energy consumption needed to remove the high-alkali-content kiln dusts. In practice, bypass savings may be minimal to avoid plugging of the preheaters, requiring a minimum amount of bypass volume. This measure, therefore, results in total fuel savings of 1.41 GJ/t blended cement. Electricity consumption, however, is expected to increase due to the added electricity consumption associated with grinding blast furnace slag (as other materials are more or less fine enough).

The cement industry in California is in a different situation than in other parts of the United States, as there are limited supplies of low-cost additives in or near the sate. This may limit the economic availability of blending materials. Therefore, the addition of ground limestone to (portland) cement is the most promising alternative on the short term to reduce the clinker content of cement in California (see below).

The costs of applying additives in cement production may vary. Capital costs are limited to extra storage capacity for the additives. However, blast furnace slag may need to be dried before use in cement production. This can be done in the grinding mill, using exhaust from the kiln, or supplemental firing, either from a gas turbine used to generate power or a supplemental air heater. The operational cost savings will depend on the purchase (including transport) costs of the additives,[33] the increased electricity costs for finer grinding, the reduced fuel costs for clinker production and electricity costs for raw material grinding and kiln drives, as well as the reduced handling and mining costs. These costs will vary by location, and would need to be assessed on the basis of individual plants. An increase in electricity consumption of 17 kWh/t (Buzzi 1997) is estimated, while an investment cost of $0.7/t cement capacity, which reflects the cost of new delivery and storage capacity (bin and weigh-feeder) is assumed.

[33] To avoid disclosing proprietary data, the USGS does not report separate value of shipments data for "cement-quality" fly ash or granulated blast furnace slag, making it impossible to estimate an average cost of the additives.

Based on the average savings for the United States, the research team estimates that the total reduction in energy use and associated emissions would be 545 kt CO_2 (148.7 ktC), assuming a replacement of 20% of the portland cement production by blended cement.

Ground limestone is another method for reducing the clinker content of cement. Ground limestone is inter-ground with clinker to produce cement, reducing the needs for clinker-making and calcination. This reduces energy use in the kiln and clinker grinding and CO_2 emissions from calcination and energy use. Addition of up to 5% limestone has shown to have no negative impacts on the performance of portland cement, while an optimized limestone cement would improve the workability slightly (Detwiler and Tennis 1996). Both Canada and Mexico have allowed the addition of limestone to portland cement, while recently the ASTM standard in the United States was adapted to also allow the use of up to 5% ground limestone in portland cement. Adding 5% limestone would reduce fuel consumption by 5% (or on average 0.3 MBtu/t clinker), power consumption for grinding by 3.3 kWh/t cement, and CO_2 emissions by almost 5%. Additional costs would be minimal (limited to material storage and distribution), while reducing kiln operation costs by 5%. This option would lead to CO_2 emission reductions of 436 ktCO_2 (119 ktC), or 4.6% of the total emissions of the California cement industry (assuming 5% limestone addition to portland cement).

The ***CemStar® process*** uses steel slag as a clinker replacement.[34] The CemStar process allows replacing 10%–15% of the clinker by electric arc furnace (EAF) slags, reducing energy needs for calcination. The slag replaces limestone (approximately 1.6 times the weight in limestone). The advantage of the CemStar process is the lack of grinding the slags, but adding them to the kiln in two-inch lumps. Depending on the location of injection, it may also save heating energy. Calcination energy is estimated to be 1.9 GJ/t clinker (Worrell et al. 2001). Because the lime in the slag is already calcined, it also reduces CO_2 emissions from calcination, and the reduced combustion energy and lower flame temperatures lead to reduced NO_X emissions (Battye et al. 2000).

Although there is no iron production in California, there is one steel plant in southern California. The slags from the plant's EAF can be used to make cement. EAFs produce between 50 and 190 kg of slag per tonne of steel (on average 116 kg/t) (U.S. DOE 1996). With an EAF production capacity in California (the TAMCO steel plant near Los Angeles),[35] annual slag production is estimated at 65 kilotonnes of slag, potentially replacing an equal amount of clinker. Hence, the overall impact on GHG emission reduction of the CemStar process in the California cement industry is limited.

For illustrative purposes alone, using a 10% injection of slags would reduce energy consumption by 0.19 GJ/t of clinker, while reducing CO_2 emissions by roughly 11%. Reductions in NO_X emissions vary by kiln type and may be between 9% and 60%, based on

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[34] The CemStar process was developed by Texas Industries in Midlothian Texas in 1994. The U.S. EPA awarded the CemStar process special recognition in 1999 as part of the ClimateWise program.

[35] The EAF at the Kingman plant in Arizona (currently owned by Nucor), just across the California border, was shut down, limiting the supply of EAF slag.

measurements at two kilns (Battye et al. 2000). Equipment costs are mainly for material handling and vary between $200,000 and $500,000 per installation. Total investments are approximately double the equipment costs. CemStar charges a royalty fee (Battye et al. 2000). Costs savings consist of increased income from additional clinker produced without increased operation and energy costs, as well as reduced iron ore purchases (as the slag provides part of the iron needs in the clinker). The iron content needs to be balanced with other iron sources such as tires and iron ore. The total annual potential emission reduction for California using nearby slag supplies is estimated to be 0.12 PJ, or 8.8 kt CO_2 (2.4 ktC).

3.3.2.2.3. Product Use

Cement and concrete are mainly used for the construction of buildings and pavement. The research team assumed that emissions produced during the use phase of concrete and cement are negligible. However, the choice of material in construction may affect the energy use and emissions of the users, even though the direct emissions during the use phase of concrete and cement are zero.

Highway road construction with concrete is initially more expensive ($80,000 versus $60,000 per mile (Gajda and VanGeem 2001)) and has higher CO_2 emissions, compared to asphalt construction due to the embodied emissions in the raw materials. However, several studies have claimed that concrete roads result in a lower resistance, and hence fuel savings (Zaniewski 1989; NRCC 2000; Gajda and VanGeem 2001). The weight of the truck results in pressure on the road, which increases the rolling resistance. This effect is more pronounced with asphalt. In practice, only experiments with heavy trucks demonstrated an impact of the pavement on fuel efficiency, as the weight of a passenger car is too small for a difference in impact.

Earlier studies (Zaniewski 1989) showed fuel savings up to 20% for heavy semi-trailer trucks when driving on a concrete versus asphalt pavement. A subsequent study by the National Research Council Canada (2000) showed fuel efficiency gains of 6%–11% depending on the surface of the road (structure, roughness), truck speed, as well as temperature. The effects of road temperature at elevated temperatures are still unclear, and may lead to higher fuel savings. A recent review of various LCA studies of pavement material by Eurobitume (the European Association of Asphalt Pavement Producers) demonstrates the difficulties in understanding the causes of the measured differences in fuel efficiency (Eurobitume 2004). This report concludes that the pavement structure is very important and that good maintenance of the road is essential for reducing fuel efficiency. It concludes that the effect of choice of pavement may be smaller and limited to 1% in fuel efficiency for heavy trucks (Eurobitume 2004).

Based on the above studies, the researcher team took a conservative approach and estimated the effect of concrete pavement on heavy truck fuel efficiency to be 2%–6%. The team estimates that in 2001, heavy trucks consumed about 528 PJ of diesel fuel on California roads (Davis and Diegel 2003; Murtishaw et al. 2005).[36] The combustion of the fuel led to emissions

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[36] Total annual fuel use for transport in California is estimated at 2950 PJ based on the California Energy Balance (Murtishaw et al. 2005). There are no data available on the share of the different modes for California. Assuming that the distribution in California is similar to that of the United States, the

of 10.78 $MtCO_2$ (2.9 MtC). Assuming that 20% of the highways would be changed to concrete pavement, and would be well maintained, a CO_2 emission reduction between 12 and 635 ktC per year would be possible. This would be partially offset by the slightly higher emissions due to concrete road construction. The team assumed a conservative net reduction of CO_2 emissions of 43 $ktCO_2$ (12 ktC).

A further advantage of concrete pavements may be the reduction of the urban heat island effect. Commonly used asphalt pavement absorbs more sunlight than pavements with a higher albedo. The absorption of the sunlight leads to increased temperatures of the pavement, which contributes to the so-called *urban heat island effect.* Replacing the asphalt with concrete (or other pavements with a high albedo) in urban areas will reduce the temperature of the pavement and in the area, and hence reduce the energy used for cooling of buildings. The total energy use savings would depend on the albedo of the new pavement and the local weather and climate conditions. A study of the replacement of pavements by a pavement with a higher albedo in Los Angeles has found a net $15 million reduction in energy costs for space conditioning (LBNL 2000), which would also lead to a reduction in air pollution.

Finally, a lighter road surface increases visibility at night (Gajda and VanGeem 2001), which may reduce the need for artificial lighting of approximately $600 per mile and year (based on a study performed in the mid-1980s in Chicago). The research team did not include the potential benefits of reduced lighting in the use stage analysis of cement and concrete. More research may provide better insights in the potential benefits in California and update the findings based on today's lighting technology.

Insulated concrete houses have a higher thermal mass than wood frame houses. The higher insulation and thermal mass may lead to increased fuel savings over the lifetime of the house. The lifetime energy used for heating and cooling a house dwarfs the embodied energy investment in the construction materials. A life-cycle inventory of an insulating concrete form house and a wood frame house in different climate zones showed reductions in CO_2 emissions over a 100-year lifetime of a house (Marceau et al. 2002). The emission reductions depend strongly on the climate in which the house is located. A (partial) life-cycle inventory (Marceau et al. 2002) showed savings of 4% in a hot dry climate (Phoenix, Arizona) to 9% in colder wet climate (Seattle, Washington). Unfortunately, the study did not include California locations.

Many life-cycle studies have been performed on buildings (e.g., Eriksson 2000; Goverse et al. 2002; Marceau and VanGeem 2002a; Marceau and VanGeem 2002b; Gajda and VanGeem 2000a; Gajda and VanGeem 2000b), and the results vary depending on the system boundaries used and assumptions made.

As the life-cycle emissions are dominated by the emissions during the use phase of the house, it is hard to provide a good estimate of the actual CO_2 emission reductions of concrete house construction in California without further study. Also, many other regulations have to

research team assumed that 17.9% of total transportation fuel use is by heavy trucks on highways (Davis and Diegel 2003).

be met when building houses in California. Without further investigation of these, it is not possible to make a full assessment of the GHG impact of concrete house construction. The research team recommends further research into this area.

3.3.2.2.4. Product End-of-Life

When roads (after 30–40 years) or buildings (after 20–100 years) are replaced, large amounts of concrete are produced as waste. Emissions at the end-of-life phase of concrete are due to the energy used in demolition, transport, and for grinding (in case of recycling) or landfilling. Based on the end-of-life GHG emissions calculation method detailed in Section 2.2, the emissions in the end-of-life phase are estimated to be 41 $kgCO_2e/t$ concrete. Typically, the concrete is landfilled, ground, and used as roadbed; or it can be re-used as aggregate. The latter will reduce the need for mining of aggregates, while reducing the need for landfills. Concrete recycling as filler has been used since the mid 1980s in the United States, but application as concrete aggregate is more recent. Recycled Concrete Aggregate (RCA) is allowed in California and is permitted for specific applications as a supporting layer. Initially, RCA was limited to 50%, but Caltrans allows up to 100% in specific applications. The City of San Francisco recently approved the use of RCA as aggregate concrete in curbs, gutter, sidewalk, and streetbase. Caltrans and other agencies continue to review the use of RCA as concrete aggregate.

A 1999 study by the CIWMB analyzed the state's waste composition. The study found that 400,000 tonnes of concrete waste were generated annually (CIWMB 1999). This is a very low estimate, when compared to the estimated use of 73 million tonnes in 2001 in California and to the annual U.S. concrete recycling rates of 90 million tonnes or more (U.S. EPA 2003b).

The potential for CO_2 emission reduction is determined by the energy used for crushing and separating the aggregate from other materials (e.g., steel). A study by the U.S. EPA (2003b) estimated the net benefit of concrete recycling as aggregate at 2.1 kgC/t recycled concrete.

Assuming that all 400,000 tonnes can be recycled, the total emission reduction from concrete recycling is nearly 4 kt CO_2 (1 ktC).

3.3.2.3. Total Potential GHG Emission Reduction

The analysis of the life cycle of cement and concrete has demonstrated that there are GHG emission reduction opportunities in every phase of the life cycle. Table 3-17 summarizes the emission reductions per phase in the life cycle of cement and concrete. Total GHG emission reductions are also shown as a share of total cement and concrete life-cycle emissions of 11.8 Mt CO_2 (3.1 MtC).

The largest potential reductions are found in improved energy efficiency in cement making, efficient use of waste fuels in cement kilns, and the production of blended cement (or addition of limestone to portland cement). While the latter two options are in the production phase of cement, the opportunities consist of the closing of life cycles of other waste material streams.

The production of blended cement and limestone addition partially compete for the same applications, and the potentials may partially overlap. Similarly, energy efficiency improvement may lower the emission reduction potential of the use of waste fuels and vice versa.

Table 3-17. Summary of Potential Opportunities for Reducing GHG Emissions Across the Cement and Concrete Life Cycle[37]

Phase	Opportunity	Potential Life-Cycle GHG Emission Reduction		
		$MtCO_2$	MtC	% of Total
Production	Energy efficiency improvement	0.68	0.19	6.0
	Waste fuels	0.62	0.17	5.4
	Blended cement	0.55	0.15	4.8
	Limestone addition to portland cement	0.44	0.12	3.8
	CemStar® (steel slags) in portland cement	0.007	0.002	0.1
Use	Fuel efficiency of heavy trucks	0.04	0.01	0.4
End-of-Life	Concrete recycling	0.004	0.001	0.03

3.4. Exploration of the Practical Opportunities and Policies in California for Promoting Life-Cycle Optimization

3.4.1. Personal Computers

As both the top producer and consumer of PCs in the United States, California stands to reap the greatest rewards from GHG emissions reductions across the manufacturing, use, and end-of-life stages of the PC product life cycle. The analysis in Section 3.3.1 has shown that significant opportunities for GHG reductions exist at each life-cycle stage.

In the production stage, the greatest estimated GHG reductions in California are attributable to **clean room energy efficiency** initiatives, such as improvements to clean room air handling systems, chillers, recirculation fans, and process controls. There are many documented case studies that demonstrate how clean room energy efficiency initiatives can lead to facility-level energy savings of 30%–60% (Naughton 2000; U.S. EPA 1997; U.S. EPA 1998; CEC 1999; LBNL 2004). However, it has been estimated that many clean room energy efficiency opportunities in the United States are currently unrealized due to several factors: (1) extremely compressed production cycles that may leave little time for efficiency improvements, (2) worries of affecting production reliability through the introduction of new technologies, (3) the fact that energy costs might only represent a small percentage of total production costs, and (4) lack of awareness of the benefits of energy efficiency improvements (Robertson 1996). Policy efforts to publicize success stories that quantify the financial benefits

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[37] The data summarized in Table 3-17 are based on secondary data from a wide range of sources— including many LCI data for which the uncertainty is not known— and on many simplifying assumptions. Thus the data in Table 3-17 should be interpreted as more illustrative of savings potentials in California than as definitive. Despite being rough estimates, the data in Table 3-17 are quite useful in indicating the potential order of magnitude of savings associated with each opportunity, and their relative effectiveness.

of clean room energy efficiency improvements—for example, those listed on LBNL's Energy Efficient Clean Room Information Site (LBNL 2004)—and to promote the adoption of facility environmental management standards (such as ISO 14000) should help to overcome this inertia.

There are also appreciable GHG reductions to be realized in California through the continued **reduction of PFC emissions in semiconductor manufacturing**. Much work is occurring on this front through the U.S. EPA's PFC Reduction/Climate Partnership for the Semiconductor Industry, a voluntary initiative that aims to reduce U.S. PFC emissions from semiconductor manufacturing by roughly 30% by 2010 (US EPA 2004a). This effort is being carried out in partnership with the World Semiconductor Council and is thus global in scope. Because this GHG reduction program is being carried out in a highly coordinated fashion at the national level, it is not discussed further in this report. Nevertheless, for such voluntary initiatives to be successful, a high rate of industry participation is necessary. Based on a review of the EPA's PFC Reduction/Climate Partnership website, it does not appear that all California-based semiconductor manufacturers have joined the initiative, which suggests that efforts to encourage or provide incentives for greater participation in California might be worthwhile.

Increasing the utilization of power management features during the use phase of the PC can lead to significant reductions in GHG emissions. One of the most significant opportunities for GHG reduction is the full-scale utilization (100%) of power management features across California's installed PC base. Power management features are virtually ubiquitous in today's PCs due to the widespread success of such initiatives as the U.S. EPA's ENERGY STAR program, the Swiss E2000 program, and the Swedish Nutek program (Cole 2003). However, power management technology cannot reach its full potential if PC users do not fully utilize this feature. Its potential will also be reduced if PC users leave their PC on overnight or during extended periods of nonuse. These barriers might be overcome through the deployment of facility "switch-off" policies (Tiller (1994) found that energy consumption could be reduced by putting stickers on monitors reminding users to turn PCs off at night) and campaigns to ensure that PC users enable power management features in PCs that do not do so automatically. Roughly 75% of the electricity consumed by PCs in California during the product use stage is attributable to commercial PCs. Thus, power management campaigns aimed at California commercial buildings could be particularly effective (see Box 1). Improved power management could also come through the promotion and institutional procurement of IEEE 1621 power management compliant PCs (LBNL 2005), as discussed below.

Box 1: Hewlett-Packard Saves 7.8 GWh Annually Through Employee Monitor Setting

The U.S. EPA has launched an initiative called the Million Monitor Drive (U.S. EPA 2005b) aimed at businesses, institutes of higher education, and government offices. Participants in the Million Monitor Drive pledge to implement power management features on monitors within their facilities, and commit to activating power management features on a specific percentage of monitors by a specific target date.

California-based Hewlett-Packard, a member of the Million Monitor Drive, has demonstrated the significant potential for energy and cost savings associated with monitor power management. Hewlett-Packard implemented a PC monitor power management setting initiative in all global facilities, whereby employee monitors would be shut off automatically after 20 minutes of inactivity (Hewlett-Packard 2005). This effort is expected to save 7.8 GWh of electricity each year, leading to $0.5 million in annual cost savings (Parkhurst pers. comm. 2005).

Two additional strategies for reducing the use stage GHG emissions of California's PCs are: (1) the **promotion of energy-efficient PC control units and displays** (e.g., those that are ENERGY STAR certified), and (2) the **encouragement of the use of LCDs,** because of their superiority in energy efficiency compared to CRT monitors.[38] Significant savings in GHG emissions could be realized under both strategies. Given that PCs in commercial buildings consume the majority of PC use stage energy in California, energy efficiency campaigns aimed at California commercial buildings could be very effective. Both strategies could be integrated seamlessly with the promotion of power management in commercial facilities to comprise a comprehensive, multi-pronged energy efficiency campaign aimed at California commercial buildings.

At the end-of-life stage, there are appreciable GHG reduction opportunities associated with **PC control unit recycling**. A major barrier to PC recycling in the United States to date has been the lack of a widespread, accessible, and economically viable e-waste recycling infrastructure. In California, however, the recently enacted Electronic Waste Recycling Act of 2003 is removing this barrier by requiring retailers to collect a recycling fee from consumers at the point of sale for new electronics products, which is used to support a statewide electronics recycling system administered by the California Integrated Waste Management Board (CIWMB 2003b). However, this legislation currently only covers CRT monitors, laptops, televisions, and LCDs, and does not include PC control units as do similar e-waste recycling laws in the European Union and Japan (Kuehr 2003). To realize the full GHG emissions potential of PC recycling, policy mechanisms for ensuring PC control unit recycling in California (such as the addition of PC control units to California's e-waste recycling regulation or the establishment of "take-back" incentives for manufacturers) should be considered.

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[38] As noted in Section 3.3.1.2.2, the full environmental implications of switching from CRT monitors to LCDs should be assessed and managed to avoid any unforeseen environmental costs.

Significant GHG emission reductions could also be realized by delaying disposal/recycling for as long as possible by **extending the life of PCs through upgrading**. PC upgrading could be promoted through educational campaigns to highlight its environmental benefits, through institutionalized upgrade policies in office environments, and by promoting PCs that are designed for ease of upgrading through green procurement policies (discussed below).

Lastly, **promoting the "green design" of PCs through institutional green procurement policies** might also have a significant impact. Large institutional buyers of computer equipment, including corporations and government agencies, are increasingly employing green procurement standards in an effort to reduce the environmental footprint of daily operations. One promising approach is for institutional buyers to require eco-label certification for all purchased PCs for such eco-labels as the U.S. EPA/DOE ENERGY STAR, the European Union's Flower, Germany's Blue Angel, and Scandinavia's independent TCO-99 eco-label (Saied and Velasquez 2003). Criteria for these eco-labels include designing PCs for ease of upgrading, designing for energy efficiency, and designing for ease of recycling. Recently, the U.S. EPA has funded the development of a comprehensive decision-support tool called "EPEAT: Electronic Products Environmental Assessment Tool" (EPEAT 2004), which is designed to provide institutional buyers of electronics with a host of green procurement metrics for informing purchasing decisions. Another strategy would be for institutional buyers to purchase only IEEE 1621-compliant PCs (LBNL 2005), thereby helping to improve institution-wide utilization of PC power management features. Green procurement initiatives reward PC manufacturers that pursue green design initiatives by giving them preferential purchasing status. The adoption of green procurement policies by California businesses and government agencies (an enormous market for PCs) could go a long way in promoting PCs that are upgradeable, energy efficient, and recyclable—design attributes that the estimates in this report suggest would lead to significant California GHG reductions.

3.4.2. Cement and Concrete

The case study of cement and concrete showed that there are opportunities for GHG emission reduction throughout the life cycle, albeit with widely varying potentials for emission reduction. Similarly, several policy options exist that can help reduce emissions throughout the life cycle.

Energy efficiency represents the most important source of potential emission reductions in the life cycle of cement. Energy efficiency improvement in the cement industry has been supported by various energy policies. Several federal agencies (e.g., U.S. DOE, U.S. EPA), state agencies (e.g., California Energy Commission), and utilities affect energy efficiency in the cement industry through existing programs (for a recent overview of industrial energy policies in various countries, see Galitsky et al. 2004).

This study's analysis has demonstrated that the use of **waste fuels** can have a large impact on GHG emissions from the cement industry, if these wastes were otherwise incinerated without energy recovery. The use of waste fuels is primarily covered by the environmental permitting process, although incentives and technology development policies can help to reduce waste problems and reduce the environmental impact of waste processing.

A kiln in the United States can only burn hazardous waste fuels with a suitable permit provided by the regulatory branch of the EPA. Currently, 25 kilns at 14 plants in the United States have permits for incinerating hazardous chemical wastes (U.S. EPA 2004b), while a number of kilns have permits to partially use alternative fuels, including tires. Permitting (especially for hazardous wastes) is often hampered by local objections to this option. Even so, a cement kiln is a very efficient way to destroy hazardous wastes, destroying 99.9999% of the toxic compounds, and emitting virtually no toxic pollutants (Worrell and Galitsky 2004).

The U.S. cement producers that use waste fuels are organized in the Cement Kiln Recycling Coalition (CKRC). The CKRC builds on the work by the European cement producer associations. The European cement industry has taken a strong lead in the use of waste or alternative fuels. This effort has been the result of a strong emphasis by the European cement industry on the use of alternative fuels (Cembureau 1997), as well as on the need for environmentally sustainable options to treat the waste flows in Europe (landfilling is to be abandoned in Europe as a result of European Commission policies). Also, the Basel Convention makes it impossible to export hazardous waste outside of the European Union.

In Germany, incineration of waste plastics is recognized as "thermal recycling" due to the high energy recovery rate. Plastic waste collected that cannot be recycled (e.g., mixed plastic) is now used as fuel in cement kilns and blast furnaces.

Waste sludge can also be used as a fuel in cement kilns. The City of Zürich (Switzerland) was looking for ways to treat the sewage sludge of the municipal wastewater treatment plants. In partnership with a local cement plant and an engineering company, a highly efficient gas cleanup system was developed to make sure that none of the compounds contained in the sludge would be emitted. This has led to a cost-effective solution for both the City of Zürich and the cement maker (Terry 2004).

Several states in the United States are now looking for innovative ways to deal with the mounting problem of discarded tires (US EPA 1999; Terry 2004). States, including California, are working with the cement industry. In 2003, about 53 million tires (Blumenthal 2004), or 18% of the annual volume of waste tires, were combusted in cement kilns, making the cement industry the largest user of tires. In the United States, 43 cement plants are permitted for the combustion of tires, and 15 cement plants in various states have expressed interest or are requesting permits to burn waste tires (PCA 2005). In California, 31 million tires are discarded annually. The CIWMB is charged with administering the processing of waste tires. In 1999, 4.1 million tires were used as fuel in three Californian cement kilns, and research on the environmental impacts is underway. Seven plants in California are permitted to burn scrap tires: California Portland (Colton and Mojave); Lehigh (Redding); Mitsubishi (Lucerne Valley); National (Lebec); TXI (Oro Grande); and Cemex (Victorville) (PCA 2005).

Given that only 13% of the tires discarded annually in California are currently used as fuel in the cement industry, there is still considerable potential for further use. Besides the annual flow of discarded tires, there are various sites throughout California that store millions of scrap tires. Following the European example, permitting of the use of tires or tire-derived fuel in more cement kilns could be considered if the plants have adequate flue gas treatment.

Cement kilns in California currently typically burn coal as a fuel. These kilns are the predominant consumers of coal in the state, producing GHG emissions from coal of approximately 1 MtC per year (Murtishaw et al. 2005). Thus, further evaluation of the opportunities to increase the use of waste fuels as a replacement for fossil fuels burned in California's cement industry should be considered. Waste fuel use in the cement industry has typically been addressed by air quality regulation and waste management agencies. Including the effects on GHG emissions in the evaluation of the permits, as well as waste management policy in the state, may accelerate the adoption of permits for cement kilns to burn waste fuels in a environmentally sound and safe manner.

Changes in the **composition of cement** (e.g., blended cement and limestone addition) also offer a substantial potential to reduce GHG emissions from the cement life cycle. The main policies in supporting the production of alternative compositions of cement are standards, specifications, and purchasing requirements or preferences.

Although there are ASTM standards for cements using blast furnace slag or fly ash, the use of blended cements is much lower in the United States, compared to most European countries. An analysis of clinker and cement production in 1995 (Worrell et al. 2001) showed that the clinker content of cement in Europe is about 80% on average, whereas in the United States, this is equal to 88%, resulting in a much higher carbon intensity of cement making in the United States.[39] The slow penetration of blended cement is due to a combination of a lack of suitable standards and product specification practices for procurement.

Both the Portland Cement Association and individual cement companies actively support the introduction of blended cement in the United States. However, progress on the introduction of blended cement in the United States and California is still slow. A recent proposal by the Portland Cement Association to change the ASTM standard to allow cement with up to 5% ground limestone still to be classified as portland cement has been approved. However, this innovation has not yet been approved by important users, such as Caltrans. In many European countries, the addition of limestone is possible: European cement standards (ENV 197-1) allow portland-limestone cement with 6%–35% limestone. In addition, both Canada and Mexico permit addition of up to 5% limestone in portland cement (O'Hare pers. comm. 2005).

Many agencies and constructors still mandate the use of portland cement for projects, instead of using a performance-based metric. Blended cement needs a slightly longer time to set, but the ultimate strength is higher than portland cement. State agencies, e.g., Caltrans, develop the procurement specifications for many large projects. By changing the specifications to a performance-based standard that allows the use of blended cement, portland cement would not be the automatic cement of choice.

Recently, the City of Berkeley, California, announced their preference for the use of blended cement in city construction projects (see Box 2) (ICLEI 2003). The University of California at

[39] There are no data available on the clinker content of cement consumed in California; data on interstate traffic of clinker and cement were not available. However, the clinker-to-cement ratio in California cement production has varied between 0.95 and 1.03 over the past years.

Berkeley has used cement with high fly-ash content in the foundation for the seismic retrofit of two buildings on its campus (Maclay 2000).

The advantage of the CemStar process is that it does not need any changes in standards or specifications, because it produces portland cement.

Box 2: Berkeley First U.S. City to Adopt Blended Cement Policy

In December 2002, the City of Berkeley City Council adopted a resolution requiring that, wherever technically appropriate, procurement of cement will specify the use of blended cement used in City buildings and other construction. The resolution also instructs the City Manager to work with local concrete delivery and manufacturing facilities to assure the availability of blended cement. The resolution was developed by the Berkeley Energy Commission and supported by the Public Works Commission. Berkeley commissioners are members of the public, appointed by the City Council and Mayor.

Cement produced through conventional methods uses virgin materials and emits greenhouse gas emissions that result from burning fossil fuels in firing lime as well as in the chemical process that changes lime to clinker. The cement industry's heavy reliance on coal leads to particularly high emission levels of CO_2, nitrous oxide, and sulfur, among other pollutants. Alternative cement production methods producing "blended cement" can substantially reduce greenhouse gas emissions and use of virgin materials. Blended cement uses binding materials such as fly ash that do not require a firing process. Using such material to produce cement helps keep the material out of landfills.

Local governments routinely purchase cement to use as the binding agent in concrete for buildings and other construction. Through cement procurement policies, jurisdictions can promote the goal of reducing emissions of greenhouse gases from buildings and operations. Local governments can also influence full communities' emissions by supporting the market for blended cement. The EPA recommends revising procurement specifications to require that contract specifications for construction projects or products allow for the use of coal fly ash or ground granulated blast furnace (GGBF) slag, when technically appropriate. (Quoted directly from ICLEI 2003.)

On the basis of the potential reduction in GHG emissions due to the use of alternative cements, the research team recommends that state agencies and cities use limestone and blended cements (where available) in appropriate projects. The example of the City of Berkeley could be used to develop a more mature market for blended cement. Adaptation of procurement specifications should preferably be implemented through performance-based standards and specifications that allow the use of blended or limestone cement.

Use Phase. Because of the uncertainties in the estimates for GHG emission reductions due to the use of cement and concrete instead of traditional materials, the team recommends further research before determining any specific policy recommendations (see Section 4.4).

Disposal of concrete is a major issue due to the volume of the demolition and construction materials waste flow. In California, the CIWMB already supports the recycling of concrete as

RCA (through the California Senate Bill SB1374).[40] Caltrans and other agencies also support further studies to the use of RCA in road construction.

Given that most concrete is recycled as roadfill, policies may focus on high-quality applications along a material quality "cascade." For example, it is better to use recycled concrete as aggregate, before using it as base material or road fill. However, given the current volume of concrete in the solid waste stream in California, the potential for GHG emission reduction of this option is limited.

The research team recommends further investigation of high-quality uses of recycled concrete, as well as an analysis of the construction and demolition waste streams to investigate the use of recycled concrete. This may help to identify the most appropriate use of recycled concrete from the perspective of GHG emission reduction and resource efficiency.

[40] Senate Bill 1374 (SB 1374, Kuehl, Chapter 501, Statutes of 2002).

4. Conclusions and Recommendations

4.1. Conclusions

LBNL estimated GHG emissions for 50 products over their life cycle, which included emissions related to the manufacture, use, and end-of-life of these products. LBNL then performed two case studies—one on PCs and one on cement and concrete—to develop more detailed estimates of product-specific opportunities for life-cycle GHG mitigation in California.

4.1.1. Case Study: Personal Computers

The assessment estimated that PC manufacturing in California is responsible for annual GHG emissions of 4.18 $MtCO_2$ (1.14 MtC). Annual emissions from the use of PCs in California were estimated to be 1.72 $MtCO_2$ (0.47 MtC). At the end-of-life stage, California's PCs were estimated to generate another 0.004 $MtCO_2$ (0.001 MtC) per year due to landfilling and demanufacturing operations.

LBNL identified a number of opportunities for reducing GHG emissions from PC manufacture and use in California. Reduction of PFC emissions during the semiconductor manufacturing process to 30% below 2000 levels could result in savings of 0.26 $MtCO_2e$ (0.07 tC). Increasing clean room energy efficiency by 30% could lead to reductions of 0.72 $MtCO_2e$ (0.19 MtC). Maximizing the energy efficiency of California's PCs to the highest current ENERGY STAR standards could lead to reductions of 0.10 $MtCO_2e$ (0.03 MtC). Enabling computer power management levels for PC control units above the current average of 25% could result in savings of 0.16 $MtCO_2e$ (0.04 MtC); maximizing the use of power management for all of California's PC components could result in savings of 0.47 $MtCO_2e$ (0.13 MtC). Switching from CRT monitors to more energy-efficient LCDs could lead to reductions of 0.48 $MtCO_2e$ (0.13 MtC). At the product end-of-life phase, the recycling of PC control units could save 500 tCO_2e (136 tC), and upgrading PCs rather than discarding them could potentially lead to savings of 0.018 $MtCO_2e$ (0.005 MtC).

4.1.2. Case Study: Cement and Concrete

LBNL estimated that emissions from cement manufacturing facilities in California are 9.6 $MtCO_2$ (2.6 MtC). When emissions for raw materials mining, transport, and all other associated activities are included, total cement GHG emissions are estimated to be 10.4 $MtCO_2$ (2.8 MtC). Cement is used primarily to make concrete. In making concrete, energy is used for mining of the aggregates and sand, mixing, shaping and curing the concrete, and transporting the raw materials, cement, and concrete to the construction site. These activities result in concrete production emissions of 1.4 $MtCO_2$ (0.4 MtC), and including the negligible emissions associated with product end-of-life, bring total emissions from cement and concrete production to 11.8 $MtCO_2$ (3.2 MtC).

LBNL identified potential GHG mitigation options for life-cycle GHG emissions for cement and concrete produced and used in California. The technical potential for energy efficiency improvement for cement manufacturing in California is estimated to be 22%, based on replacing the current equipment with best practice technology, leading to a total emission reduction of 0.68 $MtCO_2$ (0.19 MtC). The use of alternative, or waste-derived, fuels could reduce emissions by an estimated 0.62 $MtCO_2$ (0.17 MtC). The use of blended cements, in which cementious alternatives (such as fly-ash from coal fired power stations, blast furnace slag from iron production, or pozzolanic materials) are inter-ground with the clinker, is estimated to reduce emissions by 0.55 $MtCO_2$ (0.15 MtC). Ground limestone inter-ground with clinker to produce cement, reducing the needs for clinker-making and calcinations, leading to CO_2 emission reductions of 0.44 $MtCO_2$ (0.12 MtC). The CemStar process, which allows replacing 10%–15% of the clinker by EAF slags, reduces energy needs for calcination, and associated emissions are estimated to be reduced by 0.007 $MtCO_2$ (0.002 MtC). The reduced resistance of concrete highway pavement may lead to increased fuel efficiency of heavy trucks, with potential emission reduction estimated to be 0.04 $MtCO_2$ (0.01 MtC).

4.2. Recommendations

Policies to reduce life-cycle GHG emissions can be implemented at the local, state, and national levels. Of the policies discussed for the two case studies, only those related to energy efficiency improvement are currently reflected in the policies of the state and federal government. Augmented state or local level support for these policies could bring additional GHG emissions reductions. The other GHG emission reduction opportunities identified are addressed by policy areas that are often the jurisdiction of state or local agencies, providing the opportunity for policy initiatives by the state to integrate GHG emission reduction into these areas.

LBNL's analysis has shown that significant opportunities exist for GHG reductions at each stage of the product life cycle for PCs, giving rise to the following areas for potential policy initiatives:

- Promotion of further clean room energy efficiency improvements
- Promotion of institutional policies and awareness campaigns that promote more widespread usage of PC power management features, the purchase of energy-efficient PC control units and displays, and the use of LCDs instead of CRT monitors
- Promotion of upgrading to extend the life of PCs for as long as possible before recycling or disposal, through the establishment of institutional policies and public awareness campaigns
- Promotion of green procurement initiatives to guide institutional purchasing decisions

Combined, these policy areas can lead to substantial reductions in GHG emissions from PCs across their entire life cycle. Additionally, each of these policy areas can be addressed by leveraging and promoting existing initiatives in both the public and private sectors.

LBNL's analysis found that important opportunities for GHG emission reduction for cement and concrete are found in energy efficiency improvement, production of alternative cement types, use of waste as fuel, and the use of concrete for highway construction. On the basis of the cement and concrete case study, LBNL found that the following areas provide opportunities for potential policy initiatives:

- Promotion of further energy efficiency improvements in cement manufacturing

- Promotion of procurement and product specifications for changes in the composition of cement (e.g. limestone addition to portland cement, blended cement) by government agencies involved in construction (e.g., Caltrans)
- Promotion of the use of alternative fuels such as tires and other wastes to replace coal-burning in cement kilns by both waste management agencies and through air quality permitting of those cement plants that can safely incinerate wastes in the kiln
- Promotion by waste management agencies of the increased recycling of concrete for use in making aggregate

Combined, these policies can have a large impact on reducing GHG emissions, especially when combined with energy-efficiency improvement efforts. Furthermore, these options are in many cases very cost-effective, providing further initiative for all stakeholders involved.

4.3. Benefits to California

The product life-cycle optimization methodology employed in this project can be used to assess the GHG emissions associated with specific products manufactured and consumed in California. In addition, this work identified targeted policy recommendations for reducing product-specific life-cycle GHG emissions in California that are associated with PCs, and with cement and concrete. The case study information in this report can help policy makers assess the benefits of those policies for those two sectors.

Using such a life-cycle optimization approach to evaluate the potential for product-specific GHG emissions reductions can enable California's policymakers to identify mitigation options beyond those that are more commonly recognized. This type of systematic approach provides policymakers with a wider breadth of information regarding both GHG emissions sources in California and potential GHG emissions mitigation options.

Regarding the two case studies evaluated in detail, the report identified a number of life-cycle GHG emissions reduction options that could be pursued in California and provided estimates of annual potential GHG emissions reductions if these options were implemented.

LBNL identified mitigation options (reducing PFC emissions and improving clean room energy efficiency) for the manufacture of PCs in California, which could lead to annual GHG emissions reductions of approximately 1 Mt CO_2 (0.27 MtC) if completely implemented. During the use phase, GHG emissions reduction potentials of around 1 Mt CO_2 (0.27 MtC) were identified for PC control units and displays. Upgrading California's PCs and maximizing the recycling of PC control units could reduce GHG emissions by another 0.02 $MtCO_2$ (0.005 MtC) per year. This represents a total maximum technical GHG emissions reduction potential for California of over 2 $MtCO_2$ (0.55 MtC) per year.

LBNL identified mitigation options (energy efficiency improvements, use of waste fuels, and blended cements) for the manufacture of cement and concrete in California, which could lead to GHG emissions reductions of approximately 1.85 $MtCO_2$ (0.5 MtC) if completely implemented. During the use phase, improved fuel efficiency for heavy trucks could reduce emissions by 0.04 $MtC0_2$ (0.01 MtC). Increased concrete recycling during the end-of-life phase could reduce GHG emissions by 0.004 $MtCO_2$ (0.001 MtC). These options represent a total maximum technical GHG emissions reduction potential for California of nearly 2 $MtCO_2$ (0.55 MtC) per year.

Combined, the mitigation options identified in these two case studies estimated a total technical potential of nearly 4 $MtCO_2$, which is about 1% of California's 1999 net GHG emissions of 398 $MtCO_2$ (California Energy Commission 2002b).

The fact that such opportunities for reducing energy use and GHG emissions associated with producing, using, and disposing of the products in the two case studies still exist in California indicates that there are considerable economic and energy losses. Conducting life-cycle optimization evaluations for other California products could undoubtedly identify many other potential options for reducing GHG emissions. To the extent that these potential reductions are related to inefficient use of materials or energy, reducing this waste can be an important driver for improving competitiveness in a global business environment, while also addressing environmental problems such as climate change, air pollution, and waste export.

4.4. Further Research

This study identified several areas that are recommended for future research. This research was not possible within the context and resources available for this particular project. Future research in these areas may help to determine the costs and benefits to California of certain opportunities, and will help stakeholders to improve their understanding and decision-making capabilities.

4.4.1. General

This report identified the need for better data regarding actual production, consumption, energy use, and GHG emissions in California. For example, although the *Annual Survey of Manufactures* for California provides economic activity information and the *2004 Directory of California Manufacturers* lists businesses that manufacture various products in California, LBNL was not able to identify a source that provides actual manufacturing output information, by product, for the state. Such a source would improve the accuracy of the list of 50 products manufactured in California. Further, the data that LBNL relied upon for the life-cycle analysis from the EIO-LCA database are from 1997, and the latest published inventory of California GHG emissions is from 1999.

In comparing the emissions from the 50 identified products, LBNL used a "per product" metric, which meant that significantly different products were compared with no normalization. LBNL considered normalizing using an economic metric (per price of each product) or a physical metric (per tonne of each product), but neither of these metrics improved the comparability between products. Another approach could be to multiply the per unit emissions by the number of units produced in California (if that information is available) to give more information about the relative importance of the production of each product in the state.

In addition to research to improve data on manufacturing in California, the scope of this project only allowed for two detailed product case studies. In order to more clearly understand the implications of reducing GHG emissions of products manufactured in California, it would be beneficial to investigate the potential emissions reductions and associated policies for a larger number of products.

Only the energy and GHG emissions of product disposal were calculated in this study; the calculation of energy and GHG "credits" associated with materials recycling requires detailed data on product materials composition and recycling processes, and was therefore deemed beyond the scope of the 50-product analysis. Such further analysis would provide more detailed estimates of the actual life-cycle GHG emissions.

Finally, the analysis of the costs and savings associated with the implementation of the suggested GHG mitigation options was not included in the scope of this report, but is recommended for further research.

4.4.2. Computers

To estimate the GHG emissions attributable to California from the manufacture of PCs, the research team employed an energy and emissions allocation procedure based on macro-level economic and production data. Although this allocation procedure was somewhat crude, further accuracy was precluded in this analysis by a general lack of more detailed data on the geographic dispersion of the myriad production facilities that comprise global PC manufacturing supply chains. The research team recommends more detailed studies to determine exactly which PC components are manufactured in California and at what annual volumes these components are manufactured in California relative to global production. Such studies would provide a more accurate picture of California's role in global PC manufacturing supply chains, which would lead to more accurate estimates of the annual GHG emissions of California's PC manufacturing industry. Similarly, more detailed studies are needed to understand exactly where the raw materials in California's PCs are recycled to better allocate PC recycling "credits" to California.

At the PC use stage, the research team provided only rough estimates of the potential savings available to California for each GHG mitigation measure as the estimated savings depend heavily on the baseline usage assumptions detailed in Appendix A, Section A.2. These baseline assumptions were derived from published data from multiple studies over multiple years and thus bear an appreciable degree of uncertainty. The research team recommends recommend further studies to arrive at more precise estimates of the total number of PCs in use in California, the energy consumption profile of California's PCs, the extent of PC power management utilization in California homes and commercial buildings, and the number of LCDs versus CRT monitors contained in California's PC stock. A better understanding of the attributes of California's PC stock would lead to more accurate estimates of the reductions achievable through the GHG mitigation measures proposed in Table 3-11.

The research team also recommends further study on the current degree of implementation for all of the GHG mitigation measures proposed in the PC case study. A better understanding of the current saturation level of each measure would lead to more accurate estimates of the potential "untapped" GHG reductions that remain feasible in California at each stage of the PC life cycle.

Lastly, the GHG reductions estimates for PCs were based on static baseline assumptions at each life-cycle stage that are subject to change over time. For example, California's relative contribution to the global PC manufacturing industry, the size of California's PC stock, and the energy efficiency of PCs could change significantly over the next several years (to name a just a few of the important baseline assumptions). The team recommends expansion of the PC case study to include forecasting analyses, which would project the potential GHG reductions associated with each measure over time under evolving baseline conditions. Such forecasting analyses would help identify the GHG mitigation measures that are most robust in the face of an uncertain future.

4.4.3. Cement and Concrete

In the cement and concrete case study, various uncertainties were found in the data that warrant further research. Besides further research into improving the data to refine the study, the research team also recommends further study of the cement and concrete markets to assess the implementation of the GHG reduction measures identified.

In the cement and concrete study, the team used estimated emission factors for waste derived fuels. Although it may be interesting to further investigate California-specific emission factors for these fuels, the impact on the current emission estimates is likely to be small, because waste fuels were (in 2002) only 4% of all fuels consumed. However, in the future, this may become more important, as the use of waste fuels is likely to increase.

Although several studies showed potential GHG emission reductions from the use of cement and concrete in pavement and housing construction, data were not available to provide a clear estimate of these savings for California. To be able to make a more accurate estimate, more research is needed, specifically focused on California.

The use of concrete pavement may be beneficial for heavy truck fuel efficiency, as found by one study. Most work in this area refers back to a single Canadian study on this issue. Original research into the fuel efficiency effects typically found on California highways is essential to assess the potential impacts on Californian GHG emissions. The positive impacts are likely to be large, warranting increased attention to this issue.

As the life-cycle emissions are dominated by the emissions during the use phase of the house, it is hard to provide a good estimate of the actual CO_2 emission reductions of concrete house construction in California without further study. Also, many other regulations have to be met when building houses in California. Without further investigation of these issues, it is not possible to make a full assessment of the GHG impact of concrete house construction. The research team recommends further research into this area.

Concrete can be recycled as roadfill and aggregate. The CIWMB estimates the total amount of concrete waste at only 442,000 tons (CIWMB 1999), suggesting that large amounts of concrete are already recycled. The research team suggests that a more detailed study is needed to analyze the construction and demolition waste streams in California, to better understand the life cycle and flows of concrete in California. Emissions are mainly attributable to the transport of the concrete to the landfilling site.

Finally, the research team recommends further investigation of high-quality uses of recycled concrete, as well as an analysis of the construction and demolition waste streams to investigate the use of recycled concrete. This research may help to identify the most appropriate use of recycled concrete from the perspective of GHG emission reduction and resource efficiency.

5. Follow-on Work: Development and Implementation of a Revised LCA Methodology

5.1. Introduction

This section presents the results of follow-on research that was conducted subsequent to the completion of the project discussed in the previous sections. The purpose of the follow-on research was to develop and apply a revised methodology for conducting the 50-product LCA presented in Sections 2.2 and 3.2.

As noted in Sections 3.2.4 and 4.4.1, there were two primary limitations to the 50-product LCA. First, it was conducted on a "per product" basis, which meant that significantly different products had to be compared without normalization. The per-product analysis method was chosen to ensure that LCA results could be obtained for all products using currently available data while also accommodating project time and budget constraints. However, as a consequence, the 50-product LCA did not allow for a ranking of the selected products by total annual life-cycle GHG emissions in California, which is a more meaningful basis for comparison than per-product GHG emissions.

Second, it was assumed that all manufacturing-stage GHG emissions were emitted within the State of California. This assumption was necessary because of the lack of data on the geographic breakdown of manufacturing-stage GHG emissions in publicly available LCA databases.[41] As discussed in Section 3.2.4, it is likely that for many of the fifty products not all manufacturing operations occur within California's boundaries. Therefore, the results of the 50-product LCA were likely to overestimate the actual manufacturing-stage GHG emissions occurring in the state for many of the selected products.

In the follow-on research described in this section, LBNL teamed with researchers from the Green Design Initiative at Carnegie Mellon University (CMU) to develop and explore a revised approach for conducting product LCAs for the State of California that would address the above limitations. H. Scott Matthews and Györgyi Cicás from CMU were instrumental in providing both results and feedback for this section of the report.

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[41] As discussed in Section 2.2, the 50-product LCA employed Carnegie Mellon's Economic Input-Output Life-Cycle Assessment (EIO-LCA) database to estimate the manufacturing-stage emissions of each product on the LBNL list. The EIO-LCA database currently estimates the aggregate U.S. GHG emissions arising from the manufacture of a given product but does not provide further geographic specificity.

5.2. Research Objectives

There were two primary objectives to the follow-on research:

1. To develop a revised LCA methodology for products manufactured, consumed, and discarded in California that would:

   (a) provide a common and meaningful basis on which to compare the GHG emissions of different products in California, and

   (b) more accurately estimate the GHG emissions occurring within the borders of California during the manufacture of different products.

2. To apply the revised LCA methodology to the analysis on two major California-manufactured products.

5.3. Revised LCA Methodology

Table 5-1 summarizes the revised product LCA methodology that was developed in the follow-on research to meet Objective 1. It compares the revised LCA methodology to the 50-product LCA methodology described in Section 2.2. The revised LCA methodology incorporates two significant improvements over the previous approach.

First, it utilizes a California-specific Economic Input-Output Life Cycle Assessment (EIO-LCA) analysis method, which is currently under development at CMU, to disaggregate the manufacturing-stage GHG emissions of a given product into GHG emissions occurring within California and GHG emissions occurring outside of California. The California EIO-LCA approach thus provides a more accurate estimate of California's actual manufacturing-stage GHG emissions than the previous 50-product LCA methodology.

Second, the revised LCA methodology uses the total statewide life-cycle GHG emissions attributable to a given product in California each year as the basis of comparison.[42] This revision allows significantly different products to be compared based on their total annual GHG "footprint" in California, which allows different products to be ranked based on their contribution to California's annual GHG emissions.

[42] Taking the example of a personal computer (PC), the annual statewide life-cycle GHG emissions would be the sum of GHG emissions arising from all PC manufacturing, all PC use, and all PC disposal in California each year.

Table 5-1. LCA Methodology Comparison

	LCA methodology employed in the initial project	Revised LCA methodology
Product comparison basis	Life-cycle GHG emissions of a single product	Total life-cycle GHG emissions attributable to all manufacturing, use, and disposal of a product in California each year
Manufacturing stage analysis	U.S EIO-LCA based on the price of a single product; assumes all manufacturing-stage GHG emissions occur inside of California	California EIO-LCA based on the total manufacturing value of shipments of a product each year; provides breakdown of manufacturing-stage GHG emissions occurring both inside and outside of California
Use stage analysis	GHG emissions arising from the use of a single product over its useful life	GHG emissions arising from the total statewide use of a product in California each year
End-of-life stage analysis	GHG emissions arising from the collection and landfill disposal of a single product	GHG emissions arising from the total statewide collection and disposal of a product in California each year

The California EIO-LCA approach is based on 1997 EIO-LCA models for the United States (CMU-GDI 2004) and for the Far West Region of the United States (Hendrickson and Matthews 2005; Hendrickson et al. 2005) previously developed by CMU.[43] In this follow-on research, CMU developed a preliminary economic input-output (I-O) matrix for the California economy to estimate sector-level I-O exchanges between California and the rest of the United States. This preliminary I-O matrix for California was derived from the 1997 U.S. Industry Benchmark I-O table (U.S. BEA 1999), using economic multipliers based on California's 1997 value added contribution to each U.S. I-O sector (U.S. Census 2000a, 2000b). The I-O matrix for California was then coupled with GHG emissions and fuel use factors for California manufacturing sectors (U.S. Department of Energy 2001f, California Energy Commission 2002b). Based on this approach, the California EIO-LCA model estimates the amount of supply chain GHG emissions occurring both in-state and out-of-state per dollar of economic output from a given California manufacturing sector.

Under the revised LCA methodology, the manufacturing-stage GHG emissions of a given product are estimated by performing a California EIO-LCA analysis based on California's total value of shipments of that product. Total value of shipments data for California manufacturing sectors—classified by North American Industry Classification System (NAICS) code—are available from the U.S. Economic Census (U.S. Census 2005c).[44] The result is an estimate of the full supply chain GHG emissions arising from the total

[43] 1997 is the most recent year for which economic input-output data are available in the EIO-LCA database.

[44] The most recent year for which U.S. Economic Census data are available for California manufacturing sectors is 2002.

manufacturing output of a given California product sector each year, both within California and outside of California.

The use-stage GHG emissions of a given product are calculated using an approach similar to the 50-product LCA employed in the initial project. Namely, a product's use-stage GHG emissions are calculated based on the typical annual energy consumption of a single product. However, under the revised LCA approach, the total annual GHG emissions arising from the use of that product in California are calculated. This is done by multiplying the use-stage GHG emissions of a single product by the estimated total number of that product in use in California each year. Data sources for estimating the use-stage GHG emissions of various products are discussed in Section 2.2. Data sources for estimating the total number of products in use in California include the U.S. Department of Energy's Residential Energy Consumption Survey (U.S. Department of Energy 2001g) for home appliances and electronics and the U.S. Bureau of Labor Statistics' Consumer Expenditure Survey (U.S. Bureau of Labor Statistics 2002) for such products as food, drugs, and clothing.

The end-of-life-stage GHG emissions of a given product are also estimated using an approach similar to the 50-product LCA, which calculated the GHG emissions arising from the collection and disposal of a single product.[45] Under the revised LCA approach, total statewide GHG emissions arising from product disposal are calculated by multiplying the GHG emissions from the disposal of a single product by the estimated total number of that product discarded in California each year.

5.4. Research Outcomes

To meet Objective 2, the revised LCA methodology described in the previous section was applied to estimate the annual life-cycle GHG emissions of two products manufactured and consumed on a large scale in California. To aid in the selection of two products for interesting case studies, CMU provided preliminary estimates for the top GHG-emitting manufacturing sub-sectors in California based on the California EIO-LCA approach. Table 5-2 lists the top 10 GHG-producing manufacturing sub-sectors in California identified through this process, sorted by U.S. Bureau of Economic Analysis I-O commodity code.[46]

[45] The methodology for calculating the GHG emissions arising from product collection and disposal is discussed in Section 2.2.

[46] As described in Section 2.2, the EIO-LCA approach quantifies not only the GHG emissions produced by a given manufacturing sector, but also the total GHG emissions produced throughout the entire supply chain necessary to support that manufacturing sector.

Table 5-2. Top 10 GHG-Emitting Manufacturing Sectors

I-O Code	Description
311612	Meat processed from carcasses
3221A0	Paper and paperboard mills
322210	Paperboard container manufacturing
324110	Petroleum refineries
325110	Petrochemical manufacturing
325180	Other basic inorganic chemical manufacturing
325190	Other basic organic chemical manufacturing
325400	Pharmaceutical and medicine manufacturing
334413	Semiconductors and related device manufacturing
336300	Motor vehicle parts manufacturing

Table 5-3 lists the top 10 California manufacturing sub-sectors (by 6-digit NAICS code) based on value-added contribution to the total value-added output of California's manufacturing sector in 1997 (U.S. Census 2000c).[47]

Table 5-3. Top 10 Value-Added Manufacturing Sub-Sectors in California (by 6-Digit NAICS Code)

NAICS Code	Description	1997 Value Added ($1,000)
334111	Electronic computer manufacturing	11,463,543
334413	Semiconductor & related device manufacturing	10,988,501
334210	Telephone apparatus manufacturing	9,372,296
334220	Radio & TV broadcasting & wireless communications equipment manufacturing	5,386,202
324110	Petroleum refineries	4,999,046
336414	Guided missile & space vehicle manufacturing	4,864,244
334511	Search, detection, navigation, & guidance instrument manufacturing	4,647,113
326199	All other plastics product manufacturing	3,779,699
334515	Electricity measuring & testing instrument manufacturing	3,706,634
325412	Pharmaceutical preparation manufacturing	3,233,222

Source: U.S. Census 2000.

[47] As of this writing, full manufacturing sub-sector data were not yet available for California from the 2002 U.S. Economic Census. Thus, 1997 data were used to rank California's top manufacturing sub-sectors.

Based on the data in Tables 5-2 and 5-3, semiconductors and pharmaceuticals were chosen as the case study products for the revised LCA methodology, because of their importance to California from both and economic and GHG emissions perspective. For semiconductors, it was decided to focus specifically on semiconductors contained in computers (i.e., microprocessors, memory chips, logic gates, and graphics chips), which represent roughly 45% of global semiconductor sales (Turley 2003). This choice was made so that the results of the revised LCA methodology could be compared to the detailed estimates that were made for semiconductors in the personal computer (PC) case study of Section 3.3.1.

5.4.1. Manufacturing Stage Analysis

Table 5-4 lists 2002 value of shipments data for NAICS sub-sectors 325412 (Pharmaceutical preparation manufacturing)[48] and 334413 (Semiconductor and related device manufacturing) in California, the latest year for which data are available (U.S. Census 2005d, 2005e). Based on the value of shipments data in Table 5-4, the total annual GHG emissions arising from the economic output of each NAICS sub-sector in California were estimated using the California EIO-LCA approach.

Table 5-4. 2002 Value of Shipments for California Pharmaceutical and Semiconductor Manufacturing

NAICS Code	Description	California 2002 Value of Shipments ($1000)
325412	Pharmaceutical preparation manufacturing	7,204,836
334413	Semiconductor and related device manufacturing	11,936,938

Sources: U.S. Census 2005b, 2005c.

Table 5-5 lists the estimated total annual supply chain GHG emissions (i.e., the "total for all sectors") arising from the manufacture of pharmaceuticals in California in units of $MtCO_2e$. The estimated GHG emissions in Table 5-5 are divided into supply chain GHG emissions occurring within California and supply chain GHG emissions occurring outside of California. Table 5-5 lists the top 10 I-O commodity sectors (out of 491 total I-O commodity sectors) that contribute to the total annual manufacturing-stage supply chain GHG emissions for pharmaceuticals, both within and outside the state.

[48] NAICS sub-sector 325412 (Pharmaceutical preparation manufacturing) includes all finished over-the-counter and prescription pharmaceuticals for internal use (except biological), including anesthetics, antibiotics, cold remedies, insulin, mouthwashes, penicillin, and vitamins, among other common pharmaceutical products.

Table 5-5. Estimated Annual GHG Emissions from Pharmaceutical Manufacturing in California

Rank	I-O Code	Description	Annual GHG Emissions ($MtCO_2e/yr$) Inside CA	Outside CA	Total
		Total for all sectors	**1.92**	**0.82**	**2.74**
1	325400	Pharmaceutical and medicine manufacturing	0.55	0.24	0.79
2	221100	Power generation and supply	0.24	0.27	0.52
3	481000	Air transportation	0.15	0.01	0.16
4	420000	Wholesale trade	0.10	0.01	0.11
5	221200	Natural gas distribution	0.09	0.04	0.13
6	562000	Waste management and remediation services	0.08	0.00	0.08
7	533000	Lessors of nonfinancial intangible assets	0.05	0.00	0.05
8	324110	Petroleum refineries	0.05	0.02	0.07
9	492000	Couriers and messengers	0.04	0.02	0.05
10	550000	Management of companies and enterprises	0.03	0.00	0.03

Table 5-5 shows that the estimated supply chain GHG emissions arising from pharmaceutical manufacturing in California each year total 2.74 $MtCO_2e$. An estimated 1.92 $MtCO_2e$ are emitted within California each year; an additional 0.82 $MtCO_2e$ are estimated to be generated each year outside of California to support the state's pharmaceutical manufacturing operations. Although the results in Table 5-5 are only rough estimates,[49] they are useful in illuminating the order of magnitude of the GHG emissions generated by California's pharmaceutical manufacturing operations (the total estimated in-state GHG emissions are roughly 0.5% of California's 1999 net GHG emissions (California Energy Commission 2002b)). The results in Table 5-5 are also useful in illuminating the potential GHG "footprint" of California's pharmaceutical manufacturing operations outside the State's borders.

Table 5-6 lists the estimated total annual supply chain GHG emissions (i.e., the "total for all sectors") arising from the manufacture of semiconductors in California. As for pharmaceuticals, GHG emissions estimates in Table 5-6 are divided into supply chain GHG emissions occurring within California and supply chain GHG emissions occurring outside of California. The top 10 I-O commodity sectors contributing to the total annual manufacturing-stage supply chain GHG emissions for semiconductors are also listed.

[49] The limitations of the EIO-LCA approach discussed in Section 3.2.1 also apply to the California EIO-LCA approach.

Table 5-6. Estimated Annual GHG Emissions from Semiconductor Manufacturing in California

Rank	I-O Code	Description	Annual GHG Emissions ($MtCO_2e/yr$)		
			Inside CA	**Outside CA**	**Total**
		Total for all sectors	**1.58**	**0.36**	**1.93**
1	221100	Power generation and supply	0.15	0.17	0.32
2	481000	Air transportation	0.09	0.01	0.10
3	334413	Semiconductors and related device manufacturing	0.06	0.03	0.09
4	420000	Wholesale trade	0.05	0.01	0.05
5	325120	Industrial gas manufacturing	0.04	0.02	0.06
6	221200	Natural gas distribution	0.04	0.02	0.05
7	325180	Other basic inorganic chemical manufacturing	0.04	0.02	0.05
8	562000	Waste management and remediation services	0.03	0.00	0.03
9	324110	Petroleum refineries	0.02	0.01	0.03
10	492000	Couriers and messengers	0.02	0.01	0.02

Table 5-6 shows that the estimated supply chain GHG emissions arising from semiconductor manufacturing in California each year total roughly 2 $MtCO_2e$. Roughly 1.6 $MtCO_2e$ are emitted within California each year;[50] an additional 0.36 $MtCO_2e$ of GHG emissions are estimated to be generated each year outside of California to support the state's semiconductor manufacturing operations. As is true for the data in Table 5-5, the data in Table 5-6 are rough estimates but are nonetheless useful in illuminating the total annual GHG "footprint" of California's semiconductor manufacturing operations, both within and outside the state.

Table 5-7 summarizes the estimated statewide manufacturing-stage GHG emissions arising from the manufacture of pharmaceuticals and computer semiconductors in California each year, based on the revised LCA methodology. The estimated total manufacturing-stage GHG emissions for computer semiconductors was obtained by multiplying the in-state GHG emissions for all semiconductors in Table 5-6 (1.58 $MtCO_2e$/year) by 44%, which is the

[50] The in-state GHG emissions of semiconductor manufacture estimated by the California EIO-LCA approach totaled 0.74 MtCO2e. However, this estimate only includes the energy-related GHG emissions of the semiconductor manufacturing sector. As discussed in Section 3.3.1.2, there are significant process-related GHG emissions associated with semiconductor manufacture, due to the release of perfluorocompounds (PFCs). The Energy Commission estimates that 1999 releases of PFCs from California's semiconductor industry totaled 0.84 $MtCO_2e$ (California Energy Commission 2002b). The Energy Commission's estimate of GHG emissions from PFCs (0.84 MtCO2e) was added to the energy-related in-state GHG emissions estimated by the California EIO-LCA approach (0.74 MtCO2e) to bring the total estimated in-state GHG emissions to 1.58 $MtCO_2e$/year in Table 5-6.

average percentage of all global semiconductors (based on sales) that are consumed by the computer industry (Turley 2003). By multiplying by 44%, the percentage of annual in-state GHG emissions from semiconductor manufacturing that is attributable to computer semiconductors is estimated.[51]

Table 5-7. Estimated Annual In-State GHG Emissions Arising from the Manufacture of Pharmaceuticals and Computer Semiconductors in California

Product	Manufacturing-stage GHG emissions within California ($MtCO_2e/yr$)
Pharmaceuticals	1.92
Computer semiconductors	0.70

It is interesting to compare the result for computer semiconductors in Table 5-7 with the estimates for the annual in-state GHG emissions arising from the manufacture of computer semiconductors calculated in Section 3.3.1.2.1. As shown in Table 3-8, the PC case study analysis estimated that 3.27 $MtCO_2e$ are emitted each year in California (due to electricity consumption, fuel consumption, and PFC emissions in the manufacturing process for computer chips), which is roughly 4.5 times greater than the estimate of 0.70 $MtCO_2e$/year in Table 5-7. Both estimates involve significant uncertainties; the estimates in Table 3-8 were derived using process-based energy consumption data from the mid-1990s from the United States and Japan (Williams 2003) and a rough geographic allocation method, while the estimate in Table 5-7 is based on aggregate 1997 U.S. I-O tables and is subject to the limitations of the EIO-LCA methodology discussed in Section 3.2.1.

Furthermore, the estimate in Table 5-7 includes the full in-state supply chain GHG emissions of semiconductor manufacture, while the estimates in Table 3-8 only include the GHG emissions of semiconductor manufacturing facilities themselves (suggesting an even greater difference in results).

Because no publicly available data characterizing the current energy consumption and/or GHG emissions of California's semiconductor manufacturing facilities could be found, it was difficult to determine which method—the process-based method or the I-O based method—provides the most accurate estimate. Further research is warranted to compile more up-to-date data on California's semiconductor manufacturing operations. Thus, the estimates in Table 3-8 and 5-7 are useful in providing a preliminary range but should be interpreted as approximate.

[51] Recall from Section 5.4 that only semiconductors used in computers (i.e., microprocessors, memory chips, logic gates, and graphics chips)—not semiconductors for all end uses—were chosen as a case study for the revised LCA methodology.

5.4.2. Use Stage Analysis

Data from the Centers for Medicare & Medicaid Services (a federal agency within the U.S. Department of Health and Human Services) show that Californians consumed an estimated $13.5 billion worth of prescription and over-the-counter pharmaceuticals in 2000 (Centers for Medicare & Medicaid Services 2002). Although there are some prescriptions and over-the-counter pharmaceuticals that may require refrigeration—and would therefore consume energy indirectly during the product use stage—it was assumed that such pharmaceuticals represent only a small fraction of annual sales in California and therefore that use-stage GHG emissions of pharmaceuticals are negligible.

As discussed in Section 3.3.1.2.2, it was estimated that California's 16 million PCs generate a total of 1.71 $MtCO_2e$ per year during the product use stage. Data in the published literature suggest that roughly 40%–60% of a PC's power consumption is attributable to its semiconductors, which include the microprocessor, memory, and graphics cards (Chinn et al. 2003; Cole 2003; Lorch and Smith 1999). Assuming that an average of 50% of the power consumed by PCs in California is attributable to semiconductors, it was estimated that 0.86 $MtCO_2e$ of GHG emissions are generated at the use stage for computer semiconductors in California each year.

Table 5-8 summarizes the use-stage results for pharmaceuticals and computer semiconductors.

Table 5-8. Estimated Annual In-State GHG Emissions Arising from the Use of Pharmaceuticals and Computer Semiconductors in California

Product	Use-stage GHG emissions within California ($MtCO_2e$/yr)
Pharmaceuticals	N/A
Computer semiconductors	0.86

5.4.3. End-of-Life Stage Analysis

The end-of-life GHG emissions associated with pharmaceuticals in California are attributable to the collection and disposal of product packaging waste. Because no publicly available data could be found on the composition and mass of the packaging materials contained in the myriad pharmaceutical products consumed in California each year, CMU's online EIO-LCA database was employed to estimate the amount of paper and plastic packaging discarded from California's annual $13.5 billion in pharmaceutical purchases.[52] Table 5-9 lists the estimated total mass of paper and plastic discarded each year in California attributable to pharmaceutical packaging disposal.

[52] The EIO-LCA online database and methodology are discussed in Section 2.2.

Table 5-9. Estimated Annual Mass of Paper and Plastic Discarded in California from Pharmaceutical Packaging

Material	Mass discarded in California (kt/year)
Paper	15
Plastic	36

To arrive at the estimates in Table 5-9, California's total 2000 pharmaceutical spending ($13.5 billion) was first converted to 1997 dollars ($12.6 billion).[53] Next, the EIO-LCA database was employed to estimate that $12.6 billion worth of economic output from the "Pharmaceutical and medicine manufacturing" sector in 1997 would require roughly $46 million in plastic resins (specifically for the manufacture of plastic bottles and packaging films and sheets) and roughly $5 million in paperboard (specifically for the manufacture of coated and laminated paper and packaging materials). Next, assuming an average price for packaging resins in 1997 of $1.30/kg[54] (Plastics Technology 2005), it was estimated that 36,000 tonnes of plastics are discarded from pharmaceutical packaging in California each year. Finally, assuming an average price for paperboard in 1997 of $0.34/kg[55] (Foex Indexes 2005), it was estimated that 15,000 tonnes of paper are discarded from pharmaceutical packaging in California each year.

Table 5-10 presents the estimated total GHG emissions arising from the collection and disposal of pharmaceutical packaging in California each year. It was assumed that 31% of paper waste and 5% of plastics waste would be recycled, based on recent estimates of California solid waste recycling rates (California Integrated Waste Management Board 2005b, 2005c). The results in Table 5-10 are based on the end-of-life GHG emissions estimation methodology described in Section 2.2, and include the GHG emissions of collection, landfilling, and landfill gas generation (for paper waste).

Table 5-10. Estimated Annual GHG Emissions from Pharmaceutical Packaging Disposal in California

Material	GHG emissions from disposal ($ktCO_2e$/year)
Paper	11.6
Plastic	1.6
Total	**13.2**

[53] The most recent year of economic input-output data in the EIO-LCA database is 1997.

[54] The May 2005 market prices for polypropylene, high-density polyethylene, and polyethylene terephthalate resins were averaged and converted to 1997 dollars, to arrive at the estimate of $1.30/kg.

[55] The May 2005 market prices for packaging paperboard were averaged and converted to 1997 dollars, to arrive at the estimate of $0.34/kg.

The end-of-life GHG emissions associated with the disposal of computer semiconductors in California are estimated based on the analysis of PC control unit disposal presented in Section 3.3.1.2.3. It was assumed in Section 3.3.1.2.3 that roughly 3.3 million PC control units will be landfilled each year in California. It was further estimated that the total mass of semiconductor components contained in a typical PC control unit is approximately 300 grams.[56] Therefore, the total estimated mass of computer semiconductors landfilled each year in California is roughly 1,000 tonnes. Based on the GHG emissions estimation methodology outlined in Section 2.2, the collection and disposal of 1,000 tonnes of computer semiconductor waste would generate approximately 50 tonnes of CO_2e per year.

Table 5-11 summarizes the end-of-life-stage results for pharmaceuticals and computer semiconductors.

Table 5-11. Estimated Annual In-State GHG Emissions Arising from the Disposal of Pharmaceuticals and Computer Semiconductors in California

Product	End-of-life-stage GHG emissions within California ($ktCO_2e/yr$)
Pharmaceuticals	13.2
Computer semiconductors	0.05

5.4.4. Total Life-Cycle GHG Emissions

Figure 5-1 summarizes the total estimated statewide GHG emissions arising from the manufacture, use, and disposal of pharmaceuticals and computer semiconductors in California each year. Using the revised LCA methodology based on the California EIO-LCA approach, it is possible in Figure 5-1 to disaggregate manufacturing-stage GHG emissions into GHG emissions occurring within California and GHG emissions occurring outside California. It is also possible to compare each product side-by-side, based on its annual life-cycle contribution to California's GHG "footprint," both inside and outside the state.

[56] A teardown analysis of a Pentium 200 MHz PC control unit performed by the research team, which found that the mass of the microprocessor was roughly 20 grams, the total mass of the memory cards was roughly 70 grams, and the mass of the graphics card was roughly 200 grams.

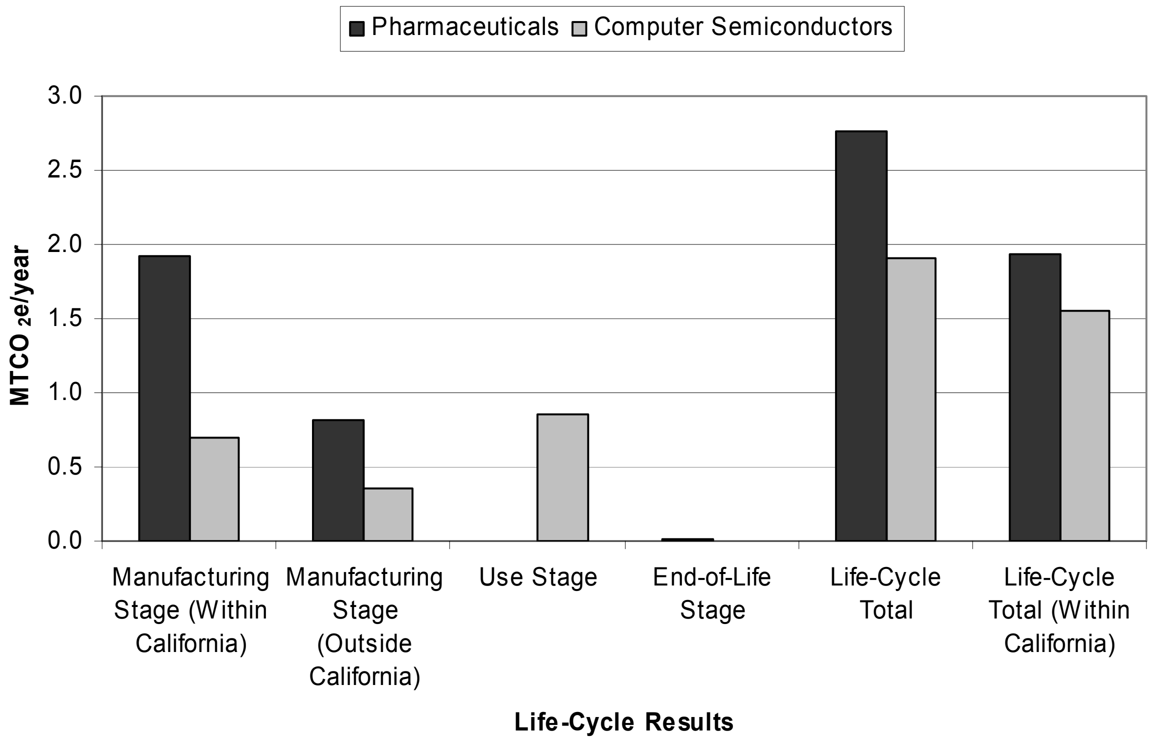

Figure 5-1. Estimated Annual Life-Cycle GHG Emissions for Pharmaceuticals and Computer Semiconductors in California

5.5. Conclusions and Recommendations

The revised LCA methodology presented in this section incorporates two significant improvements over the 50-product LCA methodology employed in the initial project.

First, the revised LCA methodology uses the total statewide life-cycle GHG emissions attributable to a given product in California each year as the basis of comparison. This approach allows products to be compared based on their total annual GHG "footprint," which will allow significantly different products in California—such as pharmaceuticals and semiconductors in Figure 5-1—to be compared on a common and meaningful basis. The revised LCA methodology can therefore be employed to perform life-cycle GHG evaluations on a wide range of products in California, to compare products based on their contribution to California's annual GHG "footprint," and to identify specific products—or specific life-cycle stages of products—for further study based on the magnitude of their annual GHG emissions.

Second, the revised LCA methodology makes use of ongoing work by CMU to develop a California EIO-LCA approach, which provides a more accurate method of estimating the manufacturing-stage GHG emissions that occur within the boundaries of the state. The California EIO-LCA approach also illuminates the extent to which the annual GHG "footprint" of California's manufacturing operations extends beyond the state's boundaries.

The follow-on research described in this section addresses two key recommendations for future research identified in the initial project: (1) development of an alternative product comparison basis, and (2) more accurate modeling of in-state manufacturing-stage GHG emissions.

By addressing these two recommendations, the revised LCA methodology offers California an improved method for conducting product LCAs for a wide range of products by providing a common basis for product-to-product comparisons and a more accurate estimate of in-state versus out-of-state product GHG emissions. The revised LCA methodology can therefore serve as a powerful screening tool for identifying specific products and product life-cycle stages in California that can be targeted for more detailed life-cycle optimization studies to reduce California's GHG "footprint."

This follow-on research also identified several important areas for continued research:

- Further development and refinement of the California EIO-LCA approach and other input-output based environmental models for California, such as the Environmental Dynamic Revenue Allocation Model (EDRAM) for California (Berck 2005) and the BEAR input-output environmental model for California (currently under development at UC Berkeley and Mills College) (Hanemann pers. comm. 2005), to improve the accuracy of manufacturing-stage GHG emissions estimates.
- The development of comprehensive use-stage energy consumption and GHG emissions databases for major energy-consuming products used in California. The development of such databases would simplify the use-stage analysis of energy-consuming products by providing a centralized resource for current data on the installed base and energy use characteristics of appliances, automobiles, office equipment, etc. in California. This follow-on research found that while such data are generally available in the public domain (e.g., through the data sources cited in Section 5.3), the data were typically spread over many years and were of varying comprehensiveness and quality. An up-to-date, centralized data clearinghouse would help ensure that the most recent and accurate data were used in California product LCA efforts.
- The analysis of additional products to further evaluate the revised LCA methodology (time and budget constraints limited the scope of this follow-on research to two case studies).
- The expansion of the revised LCA methodology to include other important environmental metrics, such as energy consumption, criteria air pollution, and solid waste generation (GHG emissions are only one product-related environmental problem to be analyzed and managed in the State of California).
- The inclusion of recycling "credits" to capture the environmental benefits of materials recycling in California.
- The development of comprehensive databases on annual waste flows and recycling statistics for major products in California. While data on waste streams and recycling rates for selected materials streams (e.g., paper, plastic, and aluminum) were found in this follow-on research, the data typically only applied to a few select products (e.g.,

newspapers and plastic bottles), were scattered over many years, and were often based on national rather than California statistics. A centralized clearinghouse of the most recent and accurate data on the annual volumes, material composition, and recycling rates of waste from various products consumed in California would simplify the end-of-life-stage analysis and would aid in calculating reasonably accurate recycling "credits."

6. References

Alsop, P. A. and J. W. Post. 1995. *The Cement Plant Operations Handbook*, (First edition). Tradeship Publications Ltd., Dorking: UK.

American Plastics Council (APC). 2004. "Seesaw Economic Recovery Leaves 2003 Plastics Volumes Flat," http://www.apcnewsmedia.com/site/tertiary.asp?TRACKID=&VID=2&CID=377&DID=1741&PSID=APC, accessed January 31, 2005.

Association of Plastics Manufacturers in Europe (APME). 2004. *Plastics: An Analysis of Plastics Consumption and Recovery in Europe*. Brussels. http://www.apme.org/media/Public_documents/20040813_143314/2002_20022003.pdf, accessed January 31, 2005.

Atlantic Consulting. 1998. *LCA Study of the Product Group Personal Computers in the EU Ecolabel Scheme*. March 1998.

Basel Action Network (BAN). 2004. *CRT Glass Recycling Survey Results*. http://www.ban.org/Library/crt_survey_2004.pdf, accessed January 31, 2005.

Battye, R., S. Walsh, and J. Lee-Greco. 2000. *NO_X Control Technologies for the Cement Industry*. Prepared for U.S. Environmental Protection Agency, Triangle Park, North Carolina.

Bennet, J. 2005. Personal communication. John Bennet, California Portland Cement, May 2005.

Berck, J. 2005. Environmental Dynamic Revenue Allocation Model (EDRAM). http://are.berkeley.edu/~peter/Research/DRAM03B/edram03.htm.

Bundesamt für Umwelt, Wald und Landshaft (BUWAL). 1998. *Life Cycle Inventories for Packagings*. Swiss Agency for the Environment, Forests, and Landscape. Environmental Series Number 250/II. CH-3003, Berne.

Blumenthal, M. 2004. "Scrap Tires Fuel U.S. Cement Industry." *Cement Americas* July 2004.

Boustead I. 1999. "Ecoprofiles of plastics and related intermediates." Association of Plastics Manufacturers in Europe, Brussels.

Boustead I. 2001. "Who Gets the Credit?" Association of Plastics Manufacturers in Europe, Brussels. www.apme.org/media/public_documents/20020221_104944/WHOGETSTHECREDITS.pdf, accessed January 31, 2005.

Buzzi, S. 1997. "Die Horomill® - Eine Neue Mühle für die Feinzerkleinerung," *ZKG International* 3 50: 127–138.

Californians Against Waste (CAW). 2004. ”E-Waste Background,“ www.cawrecycles.org/Ewaste/background%20e-waste.html, accessed January 31, 2005.

California Energy Commission (CEC). 1999. *Industrial Energy Efficiency: Clean Rooms and Laboratories for High-Technology Industries*. Sacramento, Calif.: CEC.

California Energy Commission (CEC). 2002a. *Landfill Gas-to-Energy Potential in California*. Report 500-02-041V1. Sacramento, Calif.: CEC. September 2002. http://www.energy.ca.gov/pier/final_project_reports/500-02-041v1.html.

California Energy Commission (CEC). 2002b. *Inventory of California Greenhouse Gas Emissions and Sinks: 1990–1999*. Sacramento, Calif. Publication #600-02-001F-ES.

California Energy Commission (CEC). 2004. *Computer and Electronics Industry Profile*. http://www.energy.ca.gov/pier/iaw/industry/comp.html, accessed October 26, 2004.

California Integrated Waste Management Board (CIWMB). 1996. *Market Status Report: Secondary Materials Export Markets*. www.ciwmb.ca.gov/Markets/StatusRpts/Exports.htm.

California Integrated Waste Management Board (CIWMB). 1999. *1999 Solid Waste Characterization Study.* http://www.ciwmb.ca.gov/WasteChar/Study1999/.

California Integrated Waste Management Board (CIWMB). 2003a. *Plastics White Paper: Optimizing Plastics Use, Recycling, and Disposal in California.* May 2003.

California Integrated Waste Management Board (CIWMB). 2003b. *Electronic Waste Recycling Act of 2003*. http://www.ciwmb.ca.gov/Electronics/Act2003/.

California Integrated Waste Management Board (CIWMB). 2004a. *Solid Waste Information System (SWIS) Database.* http://www.ciwmb.ca.gov/SWIS/, accessed October 27, 2004.

California Integrated Waste Management Board (CIWMB). 2004b. *Tire Recycling and Market Development.* http://www.ciwmb.ca.gov/Tires/Recycling/.

California Integrated Waste Management Board (CIWMB). 2005a. *Electronic Waste Recycling Act of 2003 (SB 20).* http://www.ciwmb.ca.gov/Electronics/Act2003/, accessed January 31, 2005.

California Integrated Waste Management Board. 2005b. “Paper Recovery (Collection /Processing).” Sacramento, California: CIWMB. www.ciwmb.ca.gov/Paper/Recovery/.

California Integrated Waste Management Board. 2005c. “Plastics Information and Resources.” Sacramento, California: CIWMB. http://www.ciwmb.ca.gov/Plastic/.

Carnegie Mellon University Green Design Initiative (CMU-GDI). 2004. *Economic Input-Output Life-Cycle Assessment (EIO-LCA) Database.* http://www.eiolca.net/.

Cembureau. 1997. *Best Available Techniques for the Cement Industry*. Brussels: Cembureau.

Centers for Medicare & Medicaid Services. 2002. *State Health Accounts: Tables by State*. Baltimore, Maryland. http://www.cms.hhs.gov/statistics/nhe/#state.

Chinn, G., S. Desai, E. DiStefano, K. Ravichandran, and S. Thakkar. 2003. "Mobile PC Platforms Enabled with Intel Centrino Mobile Technology," *Intel Technology Journal* May 2003.

Coito, F. 2004 (draft). *Industrial Case Study: The Cement Industry*. Prepared by KEMA Xenergy with assistance from Lawrence Berkeley National Laboratory. Oakland, Calif.

Coito, F., E. Worrell, L. Price, R. Friedman, and F. Powell. 2005. "Case Study of the California Cement Industry," forthcoming in Proceedings of the American Council for an Energy Efficient Economy's 2005 Summer Study on Energy Efficiency in Industry. Washington, D.C.: ACEEE.

Cole D. 2003. Energy Consumption and Personal Computers. In *Computers and the Environment: Understanding and Managing their Impacts,* Kuehr R. and E. Williams (eds.). Kluwer Academic Publishers: Dordrecht, Netherlands.

Custer W. 2001. "Market Outlook—Sept. 2001," *Circuitree* BNP Media, Troy, Mich. http://www.circuitree.com/CDA/ArticleInformation/features/BNP__Features__Item/0,2133,62531,00.html, accessed January 31, 2005.

Davis, S. C., and S. W. Diegel. 2003. *Transportation Energy Databook: Edition 23.* Oak Ridge National Laboratory, Oak Ridge, Tenn.

Detwiler, R. J., and P. D. Tennis. 1996. *The Use of Limestone in Portland Cement: A State-of-the-Art Review.* Skokie, Ill: Portland Cement Association.

Dreier T., and U. Wagner. 2000. "Life Cycle Analysis of a Personal Computer and Peripheral Equipment," *Proceedings of Electronics Goes Green 2000*+ Berlin.

ElectronicsNews.com.au. 2003. "PC shipments down to single-digit growth." http://www.electronicsnews.com.au/articles/c4/0c0169c4.asp, accessed January 31, 2005.

Electronic Products Environmental Assessment Tool (EPEAT). 2004. http://www.epeat.net/index.htm, accessed November 10, 2004.

Eriksson, P-E. 2000. *Comparative LCA's for Wood Construction and Other Construction Methods – Energy Use and GHG Emissions.* Swedish Wood Association: Sweden.

ETH Laboratorium für Energiesysteme (ETH-ESU). 1996. *Oekoinventare von Energiesystemen: Grundlagen für den ökologischen Vergleich von Energiesystemen und den Einbezug von Energiesystemen in Oekobilanzen für die Schweiz (3. überarbeitete Auflage) - Schlussbericht*, Zurich, Switzerland.

Eurobitume. 2004. *Environmental Impacts and Fuel Efficiency of Road Pavements*. Brussels, Belgium: Eurobitume.

European Environment Agency (EEA). 2004. *EMEP/CORINAIR Emission Inventory Guidebook - 3rd edition*. Copenhagen, Denmark: EEA. http://reports.eea.eu.int/EMEPCORINAIR4/en.

Foex Indexes. 2005. PIX Index for Pulp, Paper, and Packaging. http://www.foex.fi/.

Franklin Associates. 1994. *The Role of Recycling in Integrated Solid Waste Management to the Year 2000.* Report Prepared for Keep America Beautiful, Stamford, Conn.

Fujitsu-Siemens. 2001. *Simplified Environmental Report, 2001*. Paderborn, Germany.

Gajda, J., and M. VanGeem. 2000a. Energy Use in Residential Housing: A Comparison of Insulating Concrete Form and Wood Frame Walls. Skokie, Ill: Portland Cement Association (R&D Serial No. 2415).

Gajda, J., and M. VanGeem. 2000b. Energy Use in Residential Housing: A Comparison of Lightweight Concrete Masonry and Wood Frame Walls. Skokie, Ill: Portland Cement Association (R&D Serial No. 2439).

Gajda, J. W., and M. G. VanGeem. 2001. A Comparison of Six Environmental Impacts of Portland Cement Concrete and Asphalt Cement Concrete Pavements. Skokie, Ill: Portland Cement Association (R&D Serial No. 2068).

Galitsky, C., L. Price, and E. Worrell. 2004. *Energy Efficiency Programs and Policies in the Industrial Sector in Industrialized Countries.* Berkeley: Lawrence Berkeley National Laboratory (LBNL-54068).

Gartner Dataquest. 2001. "Gartner Dataquest Says Worldwide PC Market Growth Was Less Than 15 Percent in 2000." www4.gartner.com/5_about/press_room/pr20010119a.html, accessed January 31, 2005.

Gartner Dataquest. 2004. "PC shipments up 10.9% in Q4 2003." www.itfacts.biz/index.php?id=P558, accessed January 31, 2005.

Gerardino K. 2001. "Japan sees falling LCD sales." *EE Times*, September 28, 2001.

Goverse, T., M. P. Hekkert, P. Groenewegen, E. Worrell, and R. E. H. M. Smits. 2002. "Wood Innovation in the Residential Construction Sector; Opportunities and Constraints." *Resources, Conservation and Recycling* 1 (34): 53–74.

Grahl, C. L. 2004. "Glass Market Overview: Prospects Look Promising for the Glass Industry." Ceramicindustry.com. http://www.ceramicindustry.com/CDA/ArticleInformation/features/BNP__Features__Item/0,2710,134913,00.html, accessed January 31, 2005.

Graedel G., and B. Allenby. 2002. *Industrial Ecology*. 2nd ed. Prentice Hall: Upper Saddle River, New Jersey.

Hachman, M. 2003. "Sony To Phase Out Smaller CRTs." Extremetech.com. http://www.extremetech.com/article2/0,3973,841366,00.asp, accessed January 31, 2005.

Hanemann, M. Professor, Department of Agricultural and Resource Economics and Policy, University of California, Berkeley. 2005. Personal communication.

Heijiningen, R. J. J. van, J. F. M. de Castro, E. Worrell, and J. H. O. Hazewinkel. 1992. *More Energy Figures in Relation to Prevention and Reuse of Waste Streams.* Utrecht, The Netherlands: Novem (in Dutch).

Hendrickson, C.T., and H.S. Matthews. 2005. *Assessment Tool and Visualization for Regional Supply Chain Impacts*. Carnegie Mellon University, Pittsburgh, Pennsylvania. http://www.ce.cmu.edu/GreenDesign/gd/Research/REIO-LCA_onepager.pdf.

Hendrickson, C. T., L. B. Lave, H. S. Matthews, A. Horvath, S. Joshi, F. C. McMichael, H. MacLean, G. Cicas, D. Matthews and J. Bergerson. 2005. *Environmental Life Cycle Assessment of Goods and Services: An Input-Output Approach*, Resources for the Future: Washington, DC; forthcoming

Hewlett-Packard. 2005. 2005 *HP Global Citizenship Report*. Palo Alto, California. http://www.hp.com/hpinfo/globalcitizenship/gcreport/index.html.

Huang, J., J. Hanford, and F. Yang. 1999. *Residential Heating and Cooling Loads Component Analysis.* Berkeley, Calif.: Lawrence Berkeley National Laboratory. LBNL-44636.

IC Knowledge. 2004. *2004 IC Economics: Chapter 3 - Capital and Materials Spending.* www.icknowledge.com/our_products/Chapter%203.pdf, accessed January 31, 2005.

Institute of Electrical and Electronics Engineers (IEEE). 2005. *Standard for User Interface Elements in Power Control of Electronic Devices Employed in Office/Consumer Environment (IEEE 1621).* http://eetd.lbl.gov/Controls/1621/1621index.html, accessed February 10, 2005.

Intergovernmental Panel on Climate Change (IPCC). 1997. *Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories.* www.ipcc-nggip.iges.or.jp/public/gl/invs1.htm.

International Association of Electronics Recyclers (IAER). 2003. *IAER Electronics Recycling Industry Report.* Albany, New York.

International Council for Local Environmental Initiatives (ICLEI). 2003. *Hot Off the Press: News from ICLEI's Cities for Climate Protection Campaign – US.* www.iclei.org/us/hotp/hotp_summer_03.pdf.

International Energy Agency (IEA). 2003. *CO_2 Emissions from Fuel Combustion, 1971–2001*. Paris: IEA.

Kawamoto K., J. G. Koomey, B. Nordman, R. E. Brown, M. A. Piette, M. Ting, and A. K. Meier. 2001. *Electricity Used by Office Equipment and Network Equipment in the U.S.: Detailed Report and Appendices*. Lawrence Berkeley National Laboratory. LBNL-45917.

Kitou, Erasmia. 2002. *Air Pollution Assessment of Telework*. Ph.D. Thesis, Civil Engineering, UC Berkeley. May 2002.

Kuehr R. 2003. Managing PCs Through Policy: Review and Ways to Extend Lifespan. In *Computers and the Environment: Understanding and Managing their Impact.* Kuehr R. and E. Williams (eds.). Kluwer Academic Publishers: Dordrecht, The Netherlands.

Lawrence Berkeley National Laboratory (LBNL). 2000. "Cool Pavements." http://eetd.lbl.gov/HeatIsland/Pavements/.

Lawrence Berkeley National Laboratory (LBNL). 2004. "Energy Efficient Cleanroom Information Site." http://ateam.lbl.gov/cleanroom/technical.html, accessed November 5, 2004.

Lawrence Berkeley National Laboratory (LBNL). 2005. "Power Management Controls." http://eetd.lbl.gov/Controls/index.html, accessed February 10, 2005.

Lewis H., J. Gertsakis, T. Grant, N. Morelli, and A. Sweatman. 2001. *Design and Environment: A Global Guide to Designing Greener Goods*. Greenleaf Publishing: Sheffield, UK.

Lorch, J. R., and A. J. Smith. 1999. "Energy Consumption of Apple Macintosh Computers," *IEEE Micro* 18(6): 54–63.

Maclay, K. 2000. *UC Berkeley Retrofits take "Green" Approach to Concrete Foundations*. Press Release UC Berkeley, December 27, 2000.

Manufacturer's News, Inc. (MNI). 2004. *2004 Directory of California Manufacturers*. Evanston, Ill: NMI.

Marceau, M. L., and M. G. VanGeem. 2002a. Life Cycle Assessment of an Insulating Concrete Form House Compared to a Wood Frame House. Skokie, Ill: Portland Cement Association (R&D Serial No. 2571).

Marceau, M. L., and M. G. VanGeem. 2002b. Life Cycle Assessment of an Lightweight Concrete Form Masonry Compared to a Wood Frame House. Skokie, Ill: Portland Cement Association (R&D Serial No. 2573).

Marceau, M. L., J. Gajda, M. G. VanGeem, and M. A. Nisbet. 2002. Partial Environmental Life Cycle Inventory of an Insulating Concrete Form House Compared to a Wood Frame House. (PCA R&D Serial No.2464). Skokie, Ill: Portland Cement Association.

Marnay, C., D. Fisher, S. Murtishaw, A. Phadke, L. Price, and J. Sathaye. 2002. *Estimating Carbon Dioxide Emissions Factors for the California Electric Power Sector.* Berkeley, Calif.: Lawrence Berkeley National Laboratory (LBNL-49945).

Matthews S. H. 2002. "Use versus Manufacture Life Cycle Electricity and Environmental Impacts for Computer Tape Drives." *Resources Conservation & Recycling* 36 (2002): 187–196.

Matthews S. H., and D. H. Matthews. 2003. Information Technology Products and the Environment. In *Computers and the Environment: Understanding and Managing their Impacts.* Kuehr R., and E. Williams (eds.). Kluwer Academic Publishers: Dordrecht, The Netherlands.

McDougall F., P. White, M. Franke, and P. Hindle. 2001. *Integrated Solid Waste Management: A Life Cycle Inventory.* Blackwell Science: Oxford, UK.

McKendrick D. 1998. "Dispersed Concentration: Industry Location and Globalization in Hard Disk Drives." The Information Storage Industry Center, University of California, San Diego. http://isic.ucsd.edu/papers/dispersedconcentration/index.shtml, accessed January 31, 2005.

Microelectronics and Computer Technology Corporation (MCC). 1993. *Environmental Consciousness: A Strategic Competitiveness Issue for the Electronics and Computer Industry*. Austin, Texas: MCC.

Murtishaw, S., L. Price, S. de la Rue du Can, E. Masanet, E. Worrell, and J. Sathaye. 2005. *Development of Energy Balances for the State of California.* Lawrence Berkeley National Laboratory for the California Energy Commission, PIER Energy-Related Environmental Research:. CEC-500-2005-068.

National Research Council Canada (NRCC). 2000. *Effect of Pavement Surface Type on Fuel Consumption.* Ottawa, Ontario: National Research Council Canada.

Naughton P. 2000. "Energy Savings Turn into Cash Flow Savings." *Semiconductor Fabtech* 15 (3): 69–74.

Nisbet, M. A., M. L. Marceau, and M. G. VanGeem. 2002. Environmental Life Cycle Inventory of Portland Cement Concrete. Skokie, Ill: Portland Cement Association.

Nordman B., M. Piette, K. Kinney, and C. Webber. 1997. *User Guide to Power Management for PCs and Monitors.* Berkeley, Calif.: Lawrence Berkeley National Laboratory (LBNL-39466).

Nordman B., A. Meier, and M. A. Piette. 2000. *PC and Monitor Night Status: Power Management Enabling and Manual Turn-Off*. Berkeley, Calif.: Lawrence Berkeley National Laboratory (LBNL-46099).

Oak Ridge National Laboratory (ORNL). 2004. *Conversion Factors, 2004*. http://bioenergy.ornl.gov/papers/misc/energy_conv.html, accessed 21-Oct-2004.

O'Hare, A., Vice President, Portland Cement Association. 2005. Personal communication.

Parkhurst, R. Global Environmental Program Manager, Hewlett-Packard. 2005. Personal Communication.

Plastics Technology. 2005. Resin Pricing – May 2005. http://www.plasticstechnology.com/articles/200506rprice.html.

Portland Cement Association (PCA). 1997. *Blended Cement Potential Study*. Skokie, Ill.: Portland Cement Association.

Portland Cement Association (PCA). 2005. *Tire-Derived Fuel*. Skokie, Ill. and Washington, D.C.: PCA.

Roberson, J. A., C. A. Webber, M. C. McWhinney, R. E. Brown, M. J. Pinckard, and J. F. Busch. 2004. *After-hours Power Status of Office Equipment and Energy Use of Miscellaneous Plug-Load Equipment*. Berkeley, Calif.: Lawrence Berkeley National Laboratory. LBNL-53729-Revised. http://enduse.lbl.gov/Projects/OffEqpt.html.

Robertson C. 1996. *Opportunities for Energy Efficiency in the Northwest Microelectronics Industry*. Northwest Power Planning Council, Portland, Oregon.

Roth, K., F. Goldstein, and J. Kleinman. 2002. *Energy Consumption by Office and Telecommunications Equipment in Commercial Buildings—Volume I: Energy Consumption Baseline.* Washington, D.C.: Prepared by Arthur D. Little for the U.S. Department of Energy. A.D. Little Reference no. 72895-00. January. http://www.eren.doe.gov/buildings/documents.

Saied M., and G. T. Velasquez. 2003. PCs and Consumers: A Look at Green Demand, Use, and Disposal. In *Computers and the Environment: Understanding and Managing their Impacts.* Kuehr R., and E. Williams (eds.). Kluwer Academic Publishers: Dordrecht, The Netherlands.

Semiconductor Industry Association (SIA). 2004. *Total Semiconductor World Market Sales & Shares*. San Jose, California. https://www.sia-online.org/downloads/market_shares_94-present.pdf, accessed January 31, 2005.

Socolof M. L., J. G. Overly, L. E. Kincaid, and J. R. Geibig. 2001. *Desktop Computer Displays: A Life-Cycle Assessment*. Prepared for the U.S. Environmental Protection Agency, EPA-744-R-01-004a.

STAPPA/ALAPCO and ICLEI. 2003. *Clean Air and Climate Protection Software*. Produced by Torrie Smith Associates, Inc. for the State and Territorial Air Pollution Prevention Administrators and Association of Local Air Pollution Control Officials and the International Council for Local Environmental Initiatives.

Steel Recycling Institute. 2003. *Statistics on Overall Steel Recycling Rate*. Pittsburg, Penn.

Tekawa M., S. Miyamoto, and A. Inaba 1997. "Life Cycle Assessment: An Approach to Environmentally Friendly PCs." *Proceedings of the 1997 IEEE International Symposium on Electronics and the Environment*. May.

Terry, M. S. 2004. "Future Trends in Cement Manufacturing." In *Innovations in Portland Cement Manufacturing*. J. I. Bhatty, F. M. Miller, and S. H. Kosmatka. Skokie, Ill.: Portland Cement Association. 1255–1273.

Tiller T., and G. Newsham. 1994. "The Energy Consumption of Desktop Computers: Measurement and Savings Potential." *IEEE Transactions on Industry Applications*. 30 (4): 1065–1070.

Toto D. 2003. "Monitoring the future: CRT glass recyclers speculate on the future of glass-to-glass recycling markets - Commodity Focus - Cathode ray tubes." *Recycling Today* December, 2003. www.findarticles.com/p/articles/mi_m0KWH/is_12_41/ai_111896642, accessed January 31, 2005.

Turley, J. 2003. *The Essential Guide to Semiconductors*. Prentice Hall, Upper Saddle River, New Jersey.

United States Bureau of Economic Analysis (BEA). 1999. 1997 Benchmark Input-Output Accounts. Washington, D.C.: BEA.

United States Bureau of Labor Statistics. 2002. *Consumer Expenditure Survey 2002*. Washington, D.C.: BLS.

United States Bureau of Transportation Statistics (BTS). 2003. *Air Carrier Profile of Reporting U.S. Carriers*. http://www.bts.gov/publications/national_transportation_statistics/2002/html/table_air_carrier_profile.html.

United States Census. 2000a. *1997 Economic Census, General Summary: Manufacturing*. Washington, D.C.: U.S. Census EC97M31S-GS.

United States Census. 2000b. *1997 Economic Census, California: Manufacturing*. Washington, D.C.: U.S. Census EC97M31A-CA.

United States Census. 2000c. *1997 U.S. Economic Census: Geographic Area Series, California*. Washington, D.C. http://www.census.gov/epcd/ec97/ca/CA000_31.HTM.

United States Census. 2003. *Annual Survey of Manufactures: Geographical Statistics: 2001*. Washington, D.C.: U.S. Census. http://www.census.gov/prod/2003pubs/m01as-3.pdf and http://www.census.gov/mcd/asmdata/2001/ca06.htm.

United States Census. 2005a. *Development of NAICS: Background.* www.census.gov/epcd/www/naicsdev.htm.

United States Census. 2005b. *2002 Economic Census, Industry Series Reports: Manufacturing.* Washington, D.C.: U.S. Census. www.census.gov/econ/census02/guide/INDRPT31.HTM, accessed January 31, 2005.

United States Department of Energy (DOE), Office of Industrial Technologies. 1996. *Energy and Environmental Profile of the U.S. Iron and Steel Industry*. Washington, D.C.: U.S.DOE-OIT.

United States Department of Energy (DOE). 1998. *Technical Support Document: Energy Conservation Standards for Consumer Products: Cooking Products*. Washington, D.C.: DOE.

United States Department of Energy (DOE), Energy Information Administration. 1999. *Household Energy Consumption and Expenditures 1997: Total Energy Consumption in U.S. Households by Four Most Populated States.* Washington, D.C.: EIA.

United States Department of Energy (DOE), Energy Information Agency. 2001a. *Residential Energy Consumption Survey 2001.* Table HC7-7A. Washington, D.C.: EIA. http://www.eia.doe.gov/emeu/recs/recs2001/hc_pdf/homeofc_equip/hc7-7a_4popstates2001.pdf, accessed October 29, 2004.

United States Department of Energy (DOE), Energy Information Administration. 2001b. *Industrial Sector Energy Consumption Estimates, 1960–2001*, California. Washington, D.C.: EIA. www.eia.doe.gov/emeu/states/sep_use/ind/use_ind_ca.html, accessed January 31, 2005.

United States Department of Energy (DOE), Energy Information Administration. 2001c. *Residential Sector Energy Consumption Estimates, 1960–2001*, California. Washington, D.C.: EIA. www.eia.doe.gov/emeu/states/sep_use/res/use_res_ca.html, accessed January 31, 2005.

United States Department of Energy (DOE), Energy Information Administration. 2001d. *Commercial Sector Energy Consumption Estimates, 1960–2001,* California. Washington, D.C.: EIA.
www.eia.doe.gov/emeu/states/sep_use/com/use_com_ca.html, accessed January 31, 2005.

United States Department of Energy (DOE), Energy Information Administration. 2001f. *Individual State Data.* Washington, D.C.: EIA. http://www.eia.doe.gov/emeu/states/main_ca.html.

United States Department of Energy (DOE), Energy Information Agency. 2001g. *Residential Energy Consumption Survey 2001.* Washington, D.C.: EIA.

United States Department of Energy (DOE), Energy Information Agency. 2004. *Net Generation by Energy Source: Electric Utilities, 1990 through June 2004*. Washington, D.C.: EIA. www.eia.doe.gov/cneaf/electricity/epm/table1_2.html, accessed October 29, 2004.

United States Department of Labor, Bureau of Labor Statistics. 2005. Inflation Calculator. Washington, D.C.: BLS. www.bls.gov/, accessed February 12, 2005.

United States Department of Transportation (DOT), Federal Highway Administration. 2004. *Highway Statistics*. www.fhwa.dot.gov/policy/ohim/hs02/mv1.htm, accessed 21-Oct-2004.

United States Environmental Protection Agency (EPA). 1995. *Compilation of Air Pollutant Emission Factors AP-42, Fifth Edition, Volume I: Stationary Point and Area Sources*. Washington, D.C.: U.S. EPA.

United States Environmental Protection Agency (EPA). 1997. *Proceedings of the Semiconductor Energy Efficiency Opportunities Workshop*. San Jose, California, November 13–14. Washington, D.C.: U.S. EPA.

United States Environmental Protection Agency (EPA). 1998. *Computer Display Industry and Technology Profile.* Washington, D.C.: U.S. EPA. www.epa.gov/oppt/dfe/pubs/comp-dic/tech_reports/index.htm, accessed January 31, 2005.

United States Environmental Protection Agency (EPA). 1999. *State Scrap Tire Programs. A Quick Reference Guide, 1999 Update*. Washington, D.C.: U.S. EPA.

United States Environmental Protection Agency (EPA). 2000. *2000 Buy Recycled Products: Construction Products.* U.S. EPA, Washington, D.C. www.epa.gov/epaoswer/non-hw/procure/pdf/const-00.pdf.

United States Environmental Protection Agency (EPA). 2002. *In Brief: The U.S. Greenhouse Gas Inventory.* U.S. EPA, Washington, D.C. www.epa.gov/semiconductor-pfc/awards/ghgbrochure.pdf, accessed November 2, 2004.

United States Environmental Protection Agency (EPA). 2003a. *Municipal Solid Waste in the United States: 2001 Facts and Figures.* Washington, D.C.: U.S. EPA.

United States Environmental Protection Agency (EPA). 2003b. *Background Document for Life-Cycle Greenhouse Gas Emission Factors for Clay Brick Reuse and Concrete Recycling*. Washington, D.C.: U.S. EPA.

United States Environmental Protection Agency (EPA). 2004a. *PFC Reduction/Climate Partnership for the Semiconductor Industry.* www.epa.gov/semiconductor-pfc/index.html.

United States Environmental Protection Agency (EPA). 2004b. *Proposed Standards for Hazardous Air Pollutants for Hazardous Waste Combustors.* www.epa.gov/epaoswer/hazwaste/combust/newmact/hazmact.htm#California, accessed March 23, 2005.

United States Environmental Protection Agency (EPA). Energy Star Program. 2005a. *Energy Star Product Specifications, Eligibility Criteria, & Partner Commitments.* www.energystar.gov/index.cfm?c=product_specs.pt_product_specs, accessed January 31, 2005.

United States Environmental Protection Agency (EPA). Energy Star Program. 2005b. *Million Monitor Drive.* http://www.energystar.gov/index.cfm?c=pm_mmd.pr_pm_mmd, accessed June 20, 2005.

United States Federal Aviation Administration (FAA). 2004. "Average Aircraft Age," Washington, D.C.: FAA. http://av-info.faa.gov/GetFleetAge.asp, accessed November 12, 2004.

United States Geological Survey (USGS). 2003. *2002 Minerals Yearbook*. Reston, Virginia: USGS. http://minerals.usgs.gov/minerals/pubs/myb.html.

United States Geological Survey (USGS). 2004. *Flow Studies for Recycling Metal Commodities in the United States*. Reston, Virginia: USGS. http://pubs.usgs.gov/circ/2004/1196am/c1196a-m.pdf.

Westphalen, D., R. A. Zogg, A. F. Varone, and M. A. Foran. 1996. *Energy Savings Potential for Commercial Refrigeration Equipment.* Cambridge, Mass.: Prepared by Arthur D. Little, Inc. for the Building Equipment Division, Office of Building Technologies, US Department of Energy. ADL Reference No. 46230. June. www.eere.energy.gov/buildings/info/documents/pdfs/comm_refridg_equip.pdf.

Williams E. 2000. Global Production Chains and Sustainability: *The Case of High-Purity Silicon and its Applications in Information Technology and Renewable Energy*. United Nations University, Tokyo.

Williams E., R. U. Ayres, and M. Heller. 2002. "The 1.7 Kilogram Microchip: Energy and Material Use in the Production of Semiconductor Devices," *Environmental Science and Technology* 36 (24): 504–5510.

Williams E. 2003. "Environmental Impacts in the Production of Personal Computers." In *Computers and the Environment: Understanding and Managing their Impacts.* Kuehr R. and E. Williams (eds.). Kluwer Academic Publishers: Dordrecht, The Netherlands.

Williams, E. D., and Y. Sasaki. 2003. "Energy Analysis of End-of-Life Options for Personal Computers: Resell, Upgrade, Recycle." *Proceedings of the 2003 IEEE International Symposium on Electronics and the Environment* Boston, May 2003.

Williams E. 2004. "Energy Intensity of Computer Manufacturing: Hybrid Assessment Combining Process and Economic Input-Output Methods." *Environmental Science & Technology* 38 (22).

Worrell, E., L. Price, N. Martin, C. Hendriks, and L. Ozawa Meida. 2001. "Carbon Dioxide Emissions from the Global Cement Industry." *Annual Review of Energy and the Environment* 26: 303–329.

Worrell, E., and C. Galitsky. 2004. *Energy Efficiency Improvement and Cost Saving Opportunities for Cement Making: An ENERGY STAR Guide for Energy and Plant Managers.* Berkeley, California: Lawrence Berkeley National Laboratory (LBNL-54036).

Zaniewski, J. P. 1989. *Effect of Pavement Surface Type on Fuel Consumption*. Skokie, Ill.: Portland Cement Association.

ZDNet.com. 2004. *IDC bumps up PC forecast*. http://news.zdnet.com/2100-9584_22-5071262.html/, accessed October 29, 2004.

7. Glossary

ASR	Alkali-silica reactivity
ASTM	American Society for Testing and Materials
CACPS	Clean Air and Climate Protection Software
Caltrans	California Department of Transportation
CHF_3	Trifluoromethane
CIEE	California Institute for Energy and Environment
CIWMB	California Integrated Waste Management Board
CKRC	Cement Kiln Recycling Coalition
CMU	Carnegie Mellon University
CO_2	Carbon dioxide
CO_2e	Carbon dioxide equivalent
CRT	Cathode ray tube
DOE	U.S. Department of Energy
DOT	U.S. Department of Transportation
EAF	Electric arc furnace
E_{EOL}	Primary energy consumed during PC end-of-life treatment
EIO-LCA	Economic Input-Output Life-Cycle Assessment
E_M	Primary energy consumed during PC manufacture
EPA	U.S. Environmental Protection Agency
E_U	Primary energy consumed during PC use per year
E_{UG}	Primary energy required to manufacture PC upgrade components
G_{EOL}	Greenhouse gases emitted during PC end-of-life treatment
G_M	Greenhouse gases emitted during PC manufacture
G_U	Greenhouse gases emitted during PC use per year
G_{UG}	Greenhouse gases emitted during manufacture of PC upgrade components
GHG	Greenhouse gas
GJ	Gigajoule

HDD	Hard disk drive
HFC	Hydrofluorocarbon
I-O	Input-output
IEEE	Institute of Electrical and Electronics Engineers
kgC	Kilograms of carbon
$kgCO_2$	Kilograms of carbon dioxide
$kgCO_2e$	Kilograms of carbon dioxide equivalent
kt	Kilotonnes
ktC	Kilotonnes of carbon
$ktCO_2$	Kilotonnes of carbon dioxide
$ktCO_2e$	Kilotonnes of carbon dioxide equivalent
kWh	Kilowatt-hour
L_1	Useful life of a PC
L_2	Useful life of an upgraded PC
LBNL	Lawrence Berkeley National Laboratory
LCA	Life-cycle analysis
LCD	Liquid crystal display
LCE	Life-cycle primary energy per year
LCE′	Life-cycle primary energy per year for upgraded PC
LCE_S	Life-cycle primary energy savings per year
LCI	Life-cycle inventory
LFGTE	Landfill gas-to-energy
m^3	Cubic meter
MBtu	Million British thermal units
MJ	Megajoule
MtC	Million tonnes of carbon
$MtCO_2$	Million tonnes of carbon dioxide
$MtCO_2e$	Million tonnes of carbon dioxide equivalent
NAICS	North American Industrial Classification System

NF_3	Nitrogen trifluoride
NOx	Nitrous oxide
PC	Personal computer
PCA	Portland Cement Association
PCB	Printed circuit board
PFCs	Perfluorocompounds
PIER	Public Interest Energy Research
PJ	Petajoule
RCA	Recycled Concrete Aggregate
RD&D	Research, Development, and Demonstration
SF_6	Sulfur hexafluoride
t	Tonne
TWh	Terawatt-hours
UEC	Unit energy consumption
U.S. DOE	United States Department of Energy
U.S. EPA	United States Environmental Protection Agency
WECC/CNV	Western Electricity Coordinating Council, California and Southern Nevada Subregion
X	Number of PCs in a stock
x_1	Fraction of PC stock that is upgraded
x_2	Fraction of PC stock that is not upgraded

Appendix A. Personal Computer Case Study Methodology

A.1 PC Manufacturing Stage Methodology

There are several key assumptions associated with the data in Table 3-6. Individual value added data were not available for tin, lead, nickel, silver, and gold, because these metals are lumped into the "other nonferrous metals" category for primary (NAICS 331419) and secondary (NAICS 331492) production by the U.S. Economic Census. For lead, silver, and gold, we assumed California's share of U.S. production to be 8.9% based on combined valued added data for primary (NAICS 331419) and secondary (NAICS 331492) "other nonferrous metals" production. California's share of U.S. production of tin and nickel is assumed to be zero, as it appears neither metal is produced in the state (USGS 2004). For silicon wafers, we assume that the U.S. share of global production is equal to the U.S. share of global semiconductor production (i.e., that silicon wafers for domestic chip manufacturing will be domestically sourced). We estimate California's share of domestic silicon wafer production at 2% based on the locations of U.S. silicon wafer manufacturing plants identified through web research. For specialized materials (used for semiconductor and PCB manufacture), we estimate the U.S. share of global production at 34%.[57]

The energy and GHG data for manufacturing bulk materials in Table 3-7 are calculated using control unit mass breakdown data provided by Williams (column 2) and published LCI data for each material (ETH-ESU 1996; Boustead 1999). The LCI data source employed for each bulk material is listed in the "additional data sources" column. No publicly available LCI data sources could be found for tin, silver, and gold, and thus only the primary energy estimates reported by Williams are used for these materials.[58]

Several different data sources have been employed to estimate the energy consumption and GHG emissions associated with specialty materials production in Table 3-7. The production of silicon wafers for one control unit is assumed to require 53 kWh of electrical energy (Williams 2003; Williams et al. 2002). The primary energy and GHG emissions associated with this electricity consumption are calculated using a weighted average of the conversion factors listed in Table A-1, based on the geographic allocation factors for silicon wafers listed in Table 3-6.[59]

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[57] This estimate is based on the assumption that specialized materials for U.S. semiconductor and PCB manufacturing will be domestically sourced. The 34% estimate is a weighted average of the U.S. shares of production for semiconductors and PCBs, based on the observation that 69% of global specialized chemicals and materials are used for semiconductor manufacturing (on an economic basis) and 31% are used for PCB manufacturing (Williams 2003).

[58] Williams (2003) does not report GHG emissions for the manufacture of tin, silver, or gold.

[59] For example, the weighted average conversion factor for primary energy is calculated as (0.8%) 9.2 + (37.5%)12.0 + (61.8%)10.9 = 11.3 MJ/kWh based on the electricity allocated to California, the United States, and the rest of the world for silicon wafer manufacture using the allocation factors Table 3-6.

Table A-1. Primary Energy and GHG Emission Conversion Factors for Domestic and International Electricity Production

Region	Primary Energy (MJ/kWh)	CO_2 Emissions (kg/kWh)	Source(s)
California	9.2	0.40	(McDougall et al. 2001; Marnay et al. 2002)
U.S. (non-CA)	12.0	0.75	(McDougall et al. 2001; U.S. DOE 2004)
International	10.9	0.64	(McDougall et al. 2001; IEA 2004)

The specialized materials associated with semiconductor and PCB fabrication considered in this analysis are listed in Table A-2 (adapted from Williams 2003). Estimates for the primary energy and GHG emissions associated with producing these specialized materials were obtained using the 1997 EIO-LCA database sector "photographic film and chemical manufacturing" (CMU-GDI 2004; Williams 2004) and an assumed average economic value of $70 of specialty materials required per desktop (Williams 2003).[60]

Table A-2. Specialized Materials and Chemicals Used in Semiconductor and PCB Manufacturing

Process	Item
Semiconductor fabrication	Silicon wafers
	Photoresists, developers, ancillaries
	Solvents, acids
	Metals for deposition
	Gases
	Photomasks
Semiconductor packaging	Lead frames
	Encapsulants (epoxy)
	Ceramic packages
	Micro-thin bonding wires
Wiring board fabrication	Photolithographic chemicals
	Solder masks
	Copper deposition chemicals
Wiring board assembly	Solder and solder fluxes
	Cleaning agents

Note: Adapted from Williams (2003).

The estimates in Table 3-7 for semiconductor manufacturing are based on two components: (1) the energy consumed in semiconductor fabrication, and (2) a PFC adjustment to account for process-related emissions of PFCs, which are potent global warming gases emitted

[60] Williams (2003) calculates the average economic value of specialized materials per desktop by dividing the 1999 global market value of the specialized materials listed in Table A-2 that are used in computers ($8.3 billion, not including silicon wafers) by the 1999 global PC production (114 million units). This number ($73) is converted to 1997 dollars to give $70 per desktop PC.

during semiconductor manufacture that would otherwise not be included. To calculate energy consumption, it is assumed that 281 kWh of electricity and 155 MJ of fossil fuels are required to manufacture the microchips in a generic control unit (Williams 2003). This electrical energy is allocated geographically using the allocation factors in Table 3-6 and converted to primary energy and GHG emissions for each region using the conversion factors in Table A-1. The 155 MJ of fossil fuel consumption is converted to GHG emissions by assuming equal proportions of heavy oil, gas, LPG, and kerosene (Williams et al. 2002) and multiplying by fuel-specific CO_2 conversion factors (IPCC 1997).

The U.S. EPA reports that 7.4 Tg CO_2e of HFC/PFC/SF_6 were emitted from the U.S. semiconductor industry in 2000 (US EPA 2002). Based on the global production data for semiconductors listed in Table 3-6, it is assumed that the U.S. share of global semiconductor production is 38%. Assuming that the U.S. data are representative of emissions of HFC/PFC/SF_6 in all global facilities, we estimate that 19.4 Tg CO_2e of HFC/PFC/SF_6 are emitted annually worldwide from semiconductor manufacturing (19.4 Tg CO_2e = 7.4 Tg CO_2e/38%). Assuming that 49% of all semiconductors are used in computers (Williams 2003), we estimate that 9.5 Tg CO_2e of HFC/PFC/SF_6 were emitted in 2000 to manufacture computer chips. This number is divided by the total number of PCs shipped in 2000 (135,000,000) (Gartner Dataquest 2001) to arrive at the "PFC adjustment" of 70.4 kg CO_2e of HFC/PFC/SF_6 emissions per control unit.

The totals for the semiconductor fabrication process show that it is the single largest contributor to the total manufacturing energy and GHG emissions associated with control unit manufacturing. Thus, any efforts to improve the energy efficiency and reduce the GHG emissions of control unit manufacturing should clearly focus on the semiconductor manufacturing process.

The energy consumption and GHG emissions of PCB manufacturing are calculated by assuming 23.75 kWh of electricity and 5 liters of oil are required to manufacture PCBs in the control unit (Williams 2003).[61] We also assume that PC assembly consumes 51 kWh of electricity and 35 MJ of direct fossil fuels per control unit (Williams 2004). These electricity and fuel consumption data are allocated geographically and converted to primary energy and GHG emissions in the same manner as for silicon wafers and semiconductor manufacturing.

A.2 PC Use Stage Methodology

We assume two usage patterns: one for California's estimated 8 million residential PCs and one for California's estimated 8 million commercial PCs. Our assumptions in the UEC calculation for each device and usage pattern are listed in Table A-3.

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[61] Williams reports 27 kWh and 5.6 liters of oil are required to manufacture the PCBs in a control unit and CRT monitor (Williams 2003). 88% of this energy (23.75 kWh of electricity and 5 liters of oil) is allocated to the control unit based on the fact that 88% of the total epoxy resin is used in the control unit, which is a proxy for the PCB mass contained in the control unit.

Table A-3. Unit Energy Consumption (UEC) Assumptions for Residential and Commercial PC Control Units, CRT Monitors, and LCDs

Parameter	Unit	Control Unit		CRT Monitor		LCD	
		Residential	Commercial	Residential	Commercial	Residential	Commercial
Power in active mode	W	50	55	112	112	40	40
Power in low mode	W	25	25	13	13	6	6
Power in off mode	W	1.5	1.5	1	1	1	1
Power management use rate	%	25	25	75	75	75	75
Hours in active	hr/week	10	19	10	19	10	19
Hours in low	hr/week	5	61	5	61	5	61
Hours in off	hr/week	153	88	153	88	153	88
UEC	kWh/yr	**49**	**212**	**76**	**236**	**33**	**91**

The power consumption and residential and commercial usage patterns for the PC control unit are taken from published data by Kawamoto et al. (2001). The power consumption data for the CRT monitor and LCD are taken from published data on computer displays from the U.S. EPA (Socolof et al. 2001). A power management utilization rate of 25% is assumed for the control unit (Nordman et al. 2000) and a power management rate of 75% is assumed for the CRT monitor and LCD (Roberson et al. 2004).

We further assume that 80% of California's PCs currently use CRT monitors and that the remaining 20% employ LCDs (Roberson et al. 2004). Although it is difficult to estimate precisely the number of CRTs versus LCDs in use in California, the 80/20 split is based on the latest estimate of the ratio of CRTs to LCDs that was found in the published literature. We therefore assume 8 million control units, 6.4 million CRT monitors, and 1.6 million LCDs are in use in California homes and the same number of control units, CRTs, and LCDs are in use in California commercial buildings.

A.3 PC End-of-Life Phase Methodology

To estimate the primary energy and GHG emissions associated with landfilling California's non-recycled PC control units, we assume the following: an average PC control unit mass of 9 kg (Williams 2003), an average diesel fuel consumption rate for solid waste collection of 9.1 liters/t (McDougall et al. 2001), an average diesel fuel consumption rate for landfill equipment of 5.8 liters/t (Franklin Associates 1994), an average energetic value of 40 MJ/liter for diesel fuel, and a conversion factor of 3 kg CO_2e/liter for diesel fuel combustion (McDougall et al. 2001). Assuming that 92% of California's 3.6 million obsolete PC control units are landfilled each year, we estimate that roughly 18 TJ of primary energy and 1.4 $ktCO_2$ will be required to landfill California PC control units annually.

To estimate the primary energy and GHG emissions associated with demanufacturing California's CRT monitors, LCDs, and recycled PC control units each year, we assume the following: a CRT monitor mass of 24 kg and an LCD mass of 7 kg (Socolof et al. 2001), a CRT:LCD ratio of 80:20 in obsolete PCs, an average demanufacturing facility electricity consumption rate of 41 kWh/t (Fujitsu-Siemens 2001),[62] an average demanufacturing facility natural gas consumption rate of 103 kWh/t (Fujitsu-Siemens 2001), and a conversion factor of 0.21 kg CO_2e/kWh for natural gas combustion (McDougall et al. 2001). We convert demanufacturing electricity to primary energy and GHG emissions based on the California-specific conversion factors in Table 3-8. Based on these assumptions, we estimate that roughly 32 TJ of primary energy will be consumed and 2.9 ktCO_2 will be emitted for e-waste demanufacturing operations in California each year.

To estimate the primary energy and GHG emissions "credits" associated with recycling bulk materials from California's demanufactured CRT monitors, LCDs, and control units, we use mass breakdown data for control units from Williams (2003), which are summarized in Table 3-7, and mass breakdown data for CRTs and LCDs from Socolof et al. (2001). Table A-4 summarizes the maximum possible primary energy and GHG emissions "credits" associated with the bulk materials contained in each device.[63] The LCI data sources that were employed to estimate these "credits" from the mass breakdown data are also provided.

We estimate the annual primary energy and GHG emissions "credits" allocated to California for PC recycling based on the geographic allocation factors for bulk materials production listed in Table 3-6. Table A-5 summarizes the results of this geographic allocation, which assumes recycling volumes of 2.9 million CRT monitors, 700,000 LCDs, and 288,000 control units in California each year.[64] Although this allocation procedure is only a rough estimate, we feel it provides a reasonable geographic approximation of where bulk materials from e-waste are likely to be recycled (and hence are likely to substitute for virgin materials) given that: (a) many recyclable materials in California are exported for processing (CIWMB 1996), and (b) California's share of global production for the bulk materials contained in PCs is estimated to be quite small, as can be seen in Table 3-6.[65]

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[62] These demanufacturing energy data are based on a German facility, whose average energy consumption is likely to be greater than a California-based demanufacturing facility, because of differences in climate. However, no other data have yet appeared in the published literature that quantify the energy intensity of e-waste demanufacturing facilities. Thus, these data appear to be the best available estimates.

[63] The primary energy and GHG emissions "credits" represent the theoretical maximum that can be obtained by recycling 100% of each bulk material. "Credits" do not take into account the energy consumed and waste generated during materials recycling processes—for which data are somewhat scarce—and thus represent the theoretical maximum savings that can be realized through recycling. Although this assumption results in an overestimation of potential primary energy and GHG savings, it helps to illustrate the maximum potential of PC recycling as an end-of-life strategy.

[64] Based on the previously-stated assumptions that 3.6 million obsolete PCs are discarded each year in California, 80% will contain CRT monitors, 20% will contain LCDs, all displays will be recycled, and only 8% of the 3.6 million control units will be recycled.

[65] The promotion and development of in-state recycling options for e-waste materials would help

Table A-4. Estimated Primary Energy and GHG Emissions "Credits" Associated with PC Recycling

Bulk material	Control Unit			CRT Monitor			LCD			LCI data sources
	Mass (g)	Recycling Credit		Mass (g)	Recycling Credit		Mass (g)	Recycling Credit		
		Primary energy (MJ)	GHG (kg CO_2e)		Primary energy (MJ)	GHG (kg CO_2e)		Primary energy (MJ)	GHG (kg CO_2e)	
Glass	--	--	--	9,480	115	7.25	585	7	0.45	BUWAL 1998
Steel	6,050	226	13.25	6,610	247	14.46	3,055	114	6.68	ETH-ESU 1996
Copper	670	65	3.66	--	--	--	--	--	--	ETH-ESU 1996
Aluminum	440	92	4.64	441	92	4.65	65	14	0.69	ETH-ESU 1996
Plastics	650	28	1.97	3,750	159	11.36	2,600	110	7.89	Boustead 1999
Epoxy	1,040	102	7.13	--	--	--	--	--	--	Boustead 1999
Tin	47	11	--	--	--	--	--	--	--	
Lead	27	1	0.03	--	--	--	--	--	--	ETH-ESU 1996
Nickel	18	3	0.27	--	--	--	--	--	--	ETH-ESU 1996
Silver	1.4	2	--	--	--	--	--	--	--	
Gold	0.36	30	--	--	--	--	--	--	--	
Total	**8,944**	**560**	**30.95**	**22,040**	**613**	**37.72**	**6,500**	**245**	**15.71**	

The recycling credits for glass are allocated entirely to the United States, excepting California. This allocation is based on: (a) the observation that leaded glass export is regulated under the Basel Convention, to which the United States is not a party (BAN 2004), and thus leaded glass export from the United States is technically prohibited, and (b) the locations of the only three CRT glass recyclers in the United States are outside of California (Toto 2003).

The estimates in Table A-5 suggest that while PC recycling has significant potential for reducing the energy consumption and GHG emissions associated with bulk materials production, the majority of savings are likely occur outside of California.

Table A-5. Estimated Geographic Allocation of Annual Primary Energy and GHG Emissions "Credits" from Recycling California PCs

Region	Annual Recycling "Credits"		
	Primary Energy (PJ/yr)	GHG Emissions ($ktCO_2/yr$)	GHG Emissions (ktC/yr)
California	0.01	0.35	0.10
U.S. (non-CA)	0.65	41.35	11.27
International	1.45	87.61	23.89
Total	2.11	129.29	35.26

ensure that virgin materials are substituted within California, rather than outside of the state, which would increase California's recycling "credits."

To estimate the potential savings in primary energy and GHG emissions associated with upgrading PCs in California, we use the following approach.[66]

We define the average life-cycle primary energy per year (LCE) associated with a PC as follows:

(1) $$LCE = (E_M + L_1E_U + E_{EOL})/L_1$$

where E_M (MJ) is primary energy required for PC manufacture, E_U (MJ/yr) is the annual primary energy consumed during PC use, E_{EOL} (MJ) is the primary energy required at PC end-of-life, and L_1 (yr) is the useful life of the PC.

We define the average life-cycle primary energy per year associated with an upgraded PC (LCE′) as follows:

(2) $$LCE' = (E_M + L_2E_U + E_{EOL} + E_{UG})/L_2$$

where E_{UG} (MJ) is the primary energy required to manufacture the PC upgrade components (such as a new hard drive) and L_2 (yr) is the total extended useful life of the PC. The total annual primary energy savings (LCE_S) that can be realized by upgrading the PCs in a given stock is then defined as:

(3) $$LCE_S = X\,(LCE - x_1LCE' - x_2LCE)$$

where X is the total number of PCs in a given stock, x_1 is the fraction of the PC stock that is upgraded, and x_2 is the fraction of the PC stock that is not upgraded. Equation (3) therefore provides an estimate of the long-term savings that could be realized by upgrading a certain percentage of a PC stock on an ongoing basis.[67]

We consider the case of upgrading 100% of California's 16 million PCs to determine an upper bound estimate on annual primary energy savings. We estimate that upgrading 100% of California's PCs would lead to primary energy savings of nearly 300 TJ per year and GHG emissions savings of nearly 19 ktCO_2 (5 ktC) per year. Table A-6 summarizes the data we employ in Equations (1-3) to calculate these savings estimates. We consider only the energy consumption allocated to California at each life-cycle stage. A similar approach to Equations (1-3) is employed to estimate the upper bound on annual GHG savings. [68]

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[66] The calculation methodology is loosely derived from an analysis of PC resale, upgrade, and reuse by Williams and Sasaki (2003).

[67] The savings quantified by Equation (3) are steady-state; that is, the savings that could be achieved once the fractions of upgraded PCs and non-upgraded PCs in a given PC stock have stabilized over time.

[68] We substitute G_M, G_U, G_{EOL}, and G_{UG} for E_M, E_U, E_{EOL}, and E_{UG} in Equations 1–3 to estimate annual GHG emissions savings in the same manner as primary energy savings are estimated.

Table A-6. Assumptions for Estimating the Annual Savings in Primary Energy Consumption and GHG Emissions Associated with PC Upgrading

Parameter	Unit	Value
Total CA PC Stock (X)	#	16,000,000
E_M	MJ	321
E_U	MJ/yr	2463
E_{EOL}	MJ	12
E_{UG}	MJ	59
Upgrade fraction (x_1)		1
Non upgrade fraction (x_2)		0
L_1	yr	4
L_2	yr	6
Annual Energy Savings	**TJ/yr**	**288**
G_M	kg CO_2	24.8
G_U	kg CO_2/yr	107.1
G_{EOL}	kg CO_2	1.1
G_{UG}	kg CO_2	5.9
Annual GHG Savings	**ktCO_2/yr**	**18.7**

We assume an average useful life of 4 years for non-upgraded PCs; for upgraded PCs, we assume a total useful life of 6 years. To estimate E_M and G_M in Table A-6, we divide California's annual energy and GHG emissions associated with PC manufacture in Table 3-8 by 169 million—the estimated total global production volume of PCs. This provides estimates for the primary energy and GHG emissions necessary in California for the manufacture of a single control unit.[69] To estimate E_U and G_U, we divide California's annual primary energy and GHG emissions associated with PC use in Table 3-10 by 16 million (California's estimated PC stock). This provides an estimate of the annual primary energy and GHG emissions associated with an average California PC during use. To estimate E_{EOL} and G_{EOL}, we calculate the average primary energy and GHG emissions associated with processing a single end-of-life PC in California by dividing the annual energy and GHG emissions in Table 3-12 by 3.6 million, the estimated number of end-of-life PCs in California each year.

To estimate E_{UG} and G_{UG}, it is first assumed that the typical PC upgrade requires a new central processor, an expansion of memory, and a new hard disk drive. Williams and Sasaki list the 2003 costs of a new Pentium IV 2 GHz processor, 128 MB RAM addition, and 20GB

[69] Recall that CRT monitors and LCDs are assumed to be manufactured entirely overseas; thus the manufacturing that is "offset" in California due to upgrading is the manufacturing associated with control unit manufacture.

hard drive at \$190, \$30, and \$80, respectively (Williams and Sasaki 2003). We assume the hard drive is manufactured entirely overseas (McKendrick 1998) and that only the manufacture of the processor and memory are relevant to California. Next, we use the 1997 EIO-LCA database (CMU-GDI 2004) to estimate the primary energy and GHG emissions of a \$187 purchase from the "semiconductors and related devices" manufacturing sector.[70] The EIO-LCA database reports that 835 MJ of primary energy and 85 kg of CO_2e emissions are required to manufacture the processor and memory; of this, we allocate 59 MJ of primary energy and 5.9 kg CO_2e to California based on the geographic allocation factors for semiconductor and PCB manufacture listed in Table 3-6.

[70] \$187 is the combined cost of the processor and memory converted to 1997 dollars.