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A Lifecycle Emissions Model (LEM): Lifecycle Emissions from Transportation Fuels, Motor Vehicles, Transportation Modes, Electricity Use, Heating and Cooking Fuels, and Materials

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A LIFECYCLE EMISSIONS MODEL(LEM): LIFECYCLE EMISSIONS FROM TRANSPORTATION FUELS, MOTOR VEHICLES, TRANSPORTATION MODES, ELECTRICITY USE, HEATING AND COOKING FUELS, AND MATERIALS

-Documentation of methods and data-

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MAIN REPORT

December 2003

by

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ABBREVIATIONS

AEO = *Annual Energy Outlook* (Energy Information Administration)
AER = *Annual Energy Review* (Energy Information Administration)
AF = alternative fuel
AFV = alternative-fuel vehicle
ANL = Argonne National Laboratory
ARB = [California] Air Resources Board
bbl = barrel
BCF = billion cubic feet
BSFC = brake-specific fuel consumption
BTS = Bureau of Transportation Statistics (U. S. Department of Transportation)
BTU = British Thermal Unit
CEF = CO_2 -equivalency factor
CFC = chlorofluorocarbon
CFS = *Commodity Flow Survey* (U. S. Department of Transportation)
CH_2 = compressed hydrogen
CNG = compressed natural gas
CO = carbon monoxide
CO_2 = carbon dioxide
DDGS = distiller's dried grains and solubles
DOE = Department of Energy
DOT = Department of Transportation
EDI = economic damage index
EIA = Energy Information Administration (U. S. Department of Energy)
EPA = United States Environmental Protection Agency
EV = electric vehicle
FCRS = Farm Cost and Returns Survey
FHWA = Federal Highway Administration (U. S. Department of Transportation)
F-T = Fischer Tropsch
FTA = Federal Transit Administration (U. S. Department of Transportation)
FTP = Federal Test Procedure
g = gram
GHG = greenhouse-gas
GNP = Gross National Product
GRI = Gas Research Institute
GWP = global warming potential
HC = hydrocarbon
HDDV = heavy-duty diesel vehicle
HDV = heavy-duty vehicle
HFC = hydrofluorocarbon
HHV = higher heating value

ICEV = internal-combustion-engine vehicle
IPCC = Intergovernmental Panel on Climate Change
IEA = International Energy Agency
lb = pound
LDGV = light-duty gasoline vehicle
LDV = light-duty vehicle
LH_2 = liquid hydrogen
LPG = liquefied petroleum gases
LNG = liquefied natural gas
MCES = EIA's *Manufacturing Consumption of Energy*
MOBILEx = EPA's mobile-source emission-factor model, version x
NASS = National Agricultural Statistics Service
N_2O = nitrous oxide
NG = natural gas
NGLs = natural gas liquids
NMHC = non-methane hydrocarbons
NMOC = non-methane organic compound
NMOG = non-methane organic gas
NO_2 = nitrogen dioxide
NO_x = nitrogen oxides
O_3 = ozone
ORNL = Oak Ridge National Laboratory
OTA = Office of Technology Assessment (U. S. Congress; now defunct)
PART5 = EPA's mobile-source particulate emission-factor model
PM = particulate matter
PM_{10} = particulate matter of 10 micrometers or less aerodynamic diameter
$PM_{2.5}$ = particulate matter of 2.5 micrometers or less aerodynamic diameter
PSA = *Petroleum Supply Annual* (Energy Information Administration)
SIC = standard industrial classification
SO_2 = sulfur dioxide
SO_x = sulfur oxides
SRIC = short-rotation intensive-cultivation
TIUS = *Truck Inventory and Use Survey* (U. S. Bureau of the Census)
USDA = U. S. Department of Agriculture
U.S. DOE = U. S. Department of Energy
U. S. DOT = U. S. Department of Transportation
VMT = vehicle-miles of travel
VOC = volatile organic compound

INTRODUCTION

Background

This report documents changes to the methods and data in a recently revised version of the greenhouse-gas emissions model originally documented in *Emissions of Greenhouse Gases from the Use of Transportation Fuels and Electricity*, ANL/ESD/TM-22, Volumes 1 and 2, Center for Transportation Research, Argonne National Laboratory, Argonne (ANL), Illinois (DeLuchi, 1991, 1993). The revised Lifecycle Emissions Model (LEM) calculates energy use, air-pollutant emissions, and CO_2-equivalent emissions of carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), chlorofluorocarbons (CFC-12), nitrogen oxides (NO_X), carbon monoxide (CO), non-methane organic compounds (NMOCs) weighted by their ozone-forming potential, sulfur oxides (SO_X), hydrogen (H_2), and particulate matter (PM) from the lifecycle of fuels and materials for a wide range of transportation modes, vehicles, and fuels.

The LEM has been revised considerably since the publication of the original ANL report in 1993. These revisions are documented in this report and the accompanying appendices.

The **main report** presents most of the changes made to the LEM:

- changes to input and output;
- changes to data assumptions and model structure;
- emission sources added
- transportation modes, fuels, and vehicles added; and much more.

Separate appendices cover diesel-like fuel derived from soybean oil, analyses done for other countries, CO_2-equivalency factors, details of estimates of CH_4 and N_2O emission factors, the lifecycle of materials, emissions from agricultural soils, references to this main text, and other areas.

Note that this report presents only a sample of some of the *results* from the LEM. A complete set of results may be published in a separate report.

Because this report documents changes made to the model presented originally in DeLuchi (1991, 1993), it often refers to the relevant tables and sections of the original DeLuchi reports.

The need for this effort

Highway vehicles are a major source of urban air pollutants and so-called “greenhouse gases”. In most cities in North America and Europe, light-duty gasoline vehicles are major sources of volatile organic compounds (VOCs), nitrogen oxides (NO_X), and toxic air pollutants, and the single largest source of carbon monoxide (CO). Heavy-duty diesel vehicles can be significant source of NO_X, sulfur oxides (SO_X), and particulate matter (PM).

These air-pollutant emissions from highway vehicles lead to serious air quality problems. Urban areas throughout the world routinely violate national ambient air quality standards and international air-quality guidelines promulgated by the World Health Organization (WHO), especially for ambient ozone and PM. Clinical and epidemiological studies have associated ambient levels of PM, O_3, and other pollutants with human morbidity and mortality (U. S. EPA, 1996a, 1996b; McCubbin and Delucchi, 1999, 1996; Rabl and Spadaro, 2000). In response to these apparently serious health effects, the U. S. Environmental Protection Agency has promulgated new ambient air quality standards for O_3 and PM, and the WHO has determined that there is no “acceptable” or safe level of PM.

Motor vehicles also are a major source of carbon dioxide (CO_2), the most significant of the anthropogenic pollutants that can affect global climate. In the U. S., the highway-fuel lifecycle contributes about 30% of all CO_2 emitted from the use of fossil fuels (DeLuchi, 1991). In the OECD (Organization for Economic Cooperation and Development), the highway-fuel lifecycle contributes about one-quarter of all CO_2 emitted from the use of fossil fuels (DeLuchi, 1991; emissions in Europe are below the OECD-wide average, and emissions in the U. S. above). Worldwide, the highway fuel-lifecycle contributes less than 20% of total CO_2 emissions from the use of fossil fuels, primarily because outside the OECD relatively few people own and drive cars.

Many scientists now believe that an increase in the concentration of CO_2 and other “greenhouse” gases, such as methane and nitrous oxide, will increase the mean global temperature of the earth. In 1995, an international team of scientists, working as the Intergovernmental Panel on Climate Change (IPCC), concluded that “the balance of evidence suggests that there is a discernible human influence on global climate” (IPCC, 1996a, p. 5). In the long run, this global climate change might affect agriculture, coastal developments, urban infrastructure, human health, and other aspects of life on earth (IPCC, 1996b). (See Appendix D for a brief and somewhat dated overview of greenhouse gases and climate change.)

Interest in alternative transportation fuels and modes. These local, regional, and global environmental concerns are influencing international, national, and sub-national transportation policy. Over the past decade, policy makers worldwide have become increasingly interested in developing alternative fuels and vehicle technologies to reduce emissions of urban air pollutants and greenhouse gases from the transportation sector. For example, in the U. S., the “Climate Change Action Plan” proposed by President Clinton and Vice President Gore in 1993 calls on the “National Economic Council, the Office on Environmental Policy, and the Office of Science and Technology Policy to co-chair a process...to develop measures to significantly reduce greenhouse gas emissions from personal motor vehicles, including cars and light trucks” (Clinton and Gore, 1993, p. 30). The U. S. Energy Information Administration (*Alternatives to Traditional Transportation Fuels 1994, Volume 2: Emissions of Greenhouse Gases*, 1996) of the U. S. Department of Energy has published an analysis of emissions of greenhouse gases from alternative fuels, based mainly on an earlier version of the revised model described here. The IPCC (1996b) reviews the potential of alternative fuels and alternative transportation modes to reduce emissions of greenhouse gases from transportation.

There are similar initiatives in Europe, Japan, and elsewhere. (See Sperling and DeLuchi, 1993, for an evaluation of the air pollution and greenhouse gas impacts of alternative fuels in the OECD.) In 1991, the United Nations Development Program, the United Nations Environment Programme, and the World Bank established the Global Environment Facility (GEF), to help protect the global environment and promote sustainable economic growth. Transportation projects funded by the GEF have evaluated alternative transportation fuels and modes for their effectiveness in reducing the impact of transportation on air quality and global climate.

Given the growing consensus that emissions of greenhouse gases will affect global climate, the continuing problem of urban air pollution, and the expanding interest in transportation alternatives to gasoline-powered passenger cars, it is useful to keep the lifecycle energy-use and emissions model, which has been widely used and cited, up to date; hence, this major revision of the LEM. Because the model has been expanded to include PM and SO_2 emissions, ,in considerable detail, it is no longer referred to as a greenhouse-gas emissions model (even though PM and SO_2 do affect climate).

OVERVIEW OF THE REVISED LIFECYCLE EMISSIONS MODEL (LEM)

The task of developing and evaluating strategies to reduce emissions of urban air pollutants and greenhouse gases is complicated. There are many ways to produce and use energy, many sources of emissions in an energy lifecycle, and several kinds of pollutants (or greenhouse gases) emitted at each source. An evaluation of strategies to reduce emissions of greenhouse gases must be broad, detailed, and systematic. It must encompass the full “lifecycle” of a particular technology or policy, and include all of the relevant pollutants and their effects. Towards this end, I have developed a detailed, comprehensive model of lifecycle emissions of urban air pollutants and greenhouse gases from the use of variety of transportation modes.

The Lifecycle Emissions Model (LEM) estimates energy use, criteria pollutant emissions, and CO_2-equivalent greenhouse-gas emissions from a variety of transportation and energy lifecycles. It includes a wide range of modes of passenger and freight transport, electricity generation, heating, and more. For transport modes, it represents the lifecycle of fuels, vehicles, materials, and infrastructure. It calculates energy use and all regulated air pollutants plus so-called greenhouse gases. It includes input data for up to 30 countries, for the years 1970 to 2050, and is fully specified for the U. S. The remainder of this section highlights the capabilities of the LEM.

Transportation modes in the LEM

The LEM calculates lifecycle emissions for the following passenger transportation modes:

- light-duty passenger cars (internal-combustion engine vehicles [ICEVs]) operating on a range of fuel types [see below]; battery-powered electric vehicles [BPEVs]; and fuel-cell electric vehicles, with or without an auxiliary peak-power unit [FCVs];
- full-size buses (ICEVs and FCVs)
- mini-buses (albeit modeled crudely)
- mini-cars (ICEVs and BPEVs)
- motor scooters (ICEVs and BPEVs)
- bicycles
- heavy-rail transit (e.g., subways)
- light-rail transit (e.g., trolleys)

and the following freight transport modes:

- medium and heavy-duty trucks
- diesel trains
- tankers, cargo ships, and barges
- pipelines

Fuel and feedstock combinations for motor vehicles

For motor vehicles, the LEM calculates lifecycle emissions for a variety of combinations of end-use fuel (e.g., methanol), fuel feedstocks (e.g., coal), and vehicle types (e.g., fuel-cell vehicle). For light-duty vehicles, the fuel and feedstock combinations included in the LEM are:

Fuel --> ↓ Feedstock	*Gasoline*	*Diesel*	*Methanol*	*Ethanol*	*Methane (CNG, LNG)*	*Propane (LPG)*	*Hydrogen (CH2) (LH2)*	*Electric*
Petroleum	**ICEV, FCV**	**ICEV**				**ICEV**		**BPEV**
Coal	**ICEV**	**ICEV**	**ICEV, FCV**					**BPEV**
Natural gas		**ICEV**	**ICEV, FCV**		**ICEV**	**ICEV**	**ICEV, FCV**	**BPEV**
Wood or grass			**ICEV, FCV**	**ICEV, FCV**	**ICEV**			**BPEV**
Soybeans		**ICEV**						
Corn				**ICEV**				
Solar power							**ICEV, FCV**	**BPEV**
Nuclear power							**ICEV, FCV**	**BPEV**

The LEM has similar but fewer combinations for heavy-duty vehicles (HDVs), mini-cars, and motor scooters.

Fuel, material, vehicle, and infrastructure lifecycles in the LEM

The LEM estimates the use of energy, and emissions of greenhouse gases and urban air pollutants, for the complete lifecycle of fuels, materials, vehicles, and infrastructure for the transportation modes listed above. These lifecycles are constructed as follows:

Lifecycle of fuels and electricity:

- **end use**: the use of a finished fuel product, such as gasoline, electricity, or heating oil, by consumers.

- **dispensing of fuels**: pumping of liquid fuels, and compression or liquefaction of gaseous transportation fuels.

- **fuel distribution and storage**: the transport of a finished fuel product to end users and the operation of bulk-service facilities. For example, the shipment of gasoline by truck to a service station.

- **fuel production**: the transformation of a primary resource, such as crude oil or coal, to a finished fuel product or energy carrier, such as gasoline or electricity. A detailed model of emissions and energy use at petroleum refineries is included.

- **feedstock transport**: the transport of a primary resource to a fuel production facility. For example, the transport of crude oil from the wellhead to a petroleum refinery. A complete country-by-country accounting of imports of crude oil and petroleum products by country is included in the LEM.

- **feedstock production**: the production of a primary resource, such as crude oil, coal, or biomass. Based on primary survey data at energy-mining and recovery operations, or survey or estimated data for agricultural operations.

Lifecycle of materials:

- crude-ore **recovery** and finished-material **manufacture**: the recovery and transport of crude ores used to make finished materials and the manufacture of finished materials from raw materials (includes separate characterization of non-energy-related process-area emissions).

- the **transport** of finished materials to end users.

Lifecycle of vehicles:

- **materials use**: see the “lifecycle of materials”.

- **vehicle assembly**: assembly and transport of vehicles, trains, etc.

- **operation and maintenance**: energy use and emissions associated with motor-vehicle service stations and parts shops, transit stations, and so on;

- **secondary fuel cycle** for transport modes: building, servicing, and providing administrative support for transport and distribution modes such as large crude-carrying tankers or unit coal trains.

Lifecycle of infrastructure:

- **energy use and materials production**: the manufacture and transport of raw and finished materials used in the construction of highways, railways, etc., as well as energy use and emissions associated with the construction of the transportation infrastructure. (Presently these are represented crudely; future versions of the LEM will have a more detailed treatment of the infrastructure lifecycle.)

Sources of emissions in lifecycles

The LEM characterizes greenhouse gases and criteria pollutants from a variety of emission sources:

- Combustion of fuels that provide process energy (for example, the burning of bunker fuel in the boiler of a super-tanker, or the combustion of refinery gas in a petroleum refinery);
- Evaporation or leakage of energy feedstocks and finished fuels (for example, from the evaporation of hydrocarbons from gasoline storage terminals);
- Venting, leaking, or flaring of gas mixtures that contain greenhouse gases (for example, the venting of coal bed gas from coal mines);
- Chemical transformations that are not associated with burning process fuels (for example, the curing of cement, which produces CO_2, or the denitrification of nitrogenous fertilizers, which produces N_2O, or the scrubbing of sulfur oxides (SO_x) from the flue gas of coal-fired power plants, which can produce CO_2);• Changes in the carbon content of soils or biomass, or emissions of non-CO_2 greenhouse from soils, due to changes in land use.

Pollutant tracked in the LEM

The LEM estimates emissions of the following pollutants:

- carbon dioxide (CO_2);
- methane (CH_4);
- nitrous oxide (N_2O);
- carbon monoxide (CO);
- nitrogen oxides (NO_x);
- nonmethane organic compounds (NMOCs), weighted by their ozone-forming potential;
- sulfur dioxide (SO_2);
- total particulate matter (PM);
- particulate matter less than 10 microns diameter (PM_{10});
- hydrogen (H_2)
- chlorofluorocarbons (CFC-12);
- hydrofluorocarbons (HFC-134a);
- the CO_2-equivalent of all of the pollutants above

Ozone (O_3) is not included in this list because it is not emitted directly from any source in a fuel cycle, but rather is formed as a result of a complex series of chemical reactions involving CO, NO_x, and NMOCs.

The LEM estimates emissions of each pollutant individually, and also converts all of the pollutant into CO_2-equivalent greenhouse-gas emissions. To calculate total CO_2-equivalent emissions, the model uses CO_2-equivalency factors (CEFs) that convert mass emissions of all of the non-CO_2 gases into the mass amount of CO_2 with an equivalent effect on global climate. These CEFs are similar to but not necessarily the same as the "Global Warming Potentials" (GWPs) used by the Intergovernmental Panel on Climate Change (IPCC). The CEFs are discussed in Appendix D.

Material commodities in the LEM

Finally, the LEM includes the following materials:

- plain carbon steel
- high strength steel
- stainless steel
- recycled steel
- iron
- advanced composites
- other plastics
- fluids and lubricants
- rubber
- virgin aluminum
- recycled aluminum
- glass
- copper
- zinc die castings
- powdered metal components
- other materials (lead)
- sodium
- sulfur
- titanium
- sulfuric acid
- potassium hydroxide
- nickel and compounds
- lithium
- cement
- concrete
- limestone
- agricultural chemicals (mainly fertilizers)

Note that recycled steel and recycled aluminum are treated as separate materials from virgin steel and virgin aluminum. In this way, the full lifecycle of materials, including recycling, is explicitly represented.

Input: projections of energy use and emissions

As part of a major revision to the LEM, projections have been added of energy use and emissions, or changes in energy use and emissions, for the period 1970 to 2050. The user now specifies any target year between 1970 and 2050, and the model looks up or calculates energy-use intensities, emission factors, or other data for the specified year.

There are several different kinds of projections in the LEM:

- look-up tables (usually based on energy-use or emissions projections from the EIA);
- constant percentage changes per year;
- logistic functions with upper *or* lower limits; and
- logistic functions with upper *and* lower limits.

These projections are discussed in more detail in a separate section of the model documentation.

Major outputs of the LEM

The LEM produces the following tables of results (discussed in more detail in a separate section of this document).

- Emissions per mile from motor vehicles: CO2-equivalent emissions (in g/mi) by stage of fuel cycle and for vehicle manufacture, for the feedstock/fuel/vehicle combinations shown above.

- Emissions from electricity use: CO_2-equivalent emissions (in g/kWh-delivered) for different sources of electricity generation.

- Emissions from use of heating fuels: CO_2-equivalent emissions (in g/10^6-BTU-heat-delivered) for natural gas, LPG, electricity, and fuel oil.

- Summary of percent change in lifecycle g/mi emissions from alternative-fuel vehicles, relative to conventional gasoline LDVs or diesel HDVs.

- BTUs of process and end-use energy per mile of travel by stage of lifecycle, for different feedstock/fuel/vehicle combinations.

- Breakdown of energy use by type of energy (e.g., diesel fuel, natural gas, propane), stage of lifecycle, and feedstock/fuel combination.

- Vehicle characteristics: input data and results regarding vehicle weight and energy use.

- Emissions from EVs, by region: a macro runs the model for regional data for EV recharging and prints the g / mi results for up to six different regions.

- Emissions by IPCC sector: The g / mi results for vehicles are mapped into the IPCC sectors used in GHG accounting (e.g., “energy / road transport,” “energy / industry,” “land-use / forestry”).

- Emissions by geographic sector: The g / mi results for vehicles are mapped into a geographic framework that distinguishes in-country from outside-of-country emissions.

- Emissions by individual pollutant: one set of tables reports emissions of each individual pollutant (not weighted by CO_2-equivalency factors) for each stage of the upstream fuel cycle for each feedstock / fuel. Another table does the same for vehicle manufacture and assembly.

- CO_2-equivalent emissions by pollutant: a new table summarizes the contribution of each pollutant to upstream fuel cycle CO_2-equivalent emissions.

- Emissions from complete transportation scenarios: a new table of results shows g / passenger-mi emissions from a user-specified mix of travel by conventional motor vehicles, alternative-fuel vehicles (including electric vehicles), mini-cars, scooters, buses, trolleys, subways, bicycles, and walking.

- Print macros: the LEM has macros that run the model for up to 40 different target years and then prints a pre-selected group of results tables in publication-ready format.

- Emissions from other countries: the LEM can be programmed to calculate all results for the characteristics of any of up to 20 different countries. Separate data files exist within the LEM for each of the countries.

Overview of revisions to the LEM (since 1993 version)

The structure and input data of the LEM have been completely overhauled. For example, the inputs and model structure for vehicle emissions, vehicle fuel economy, feedstock recovery, transportation of feedstocks, fuel production, and distribution of fuels have been redone to be more detailed, flexible, consistent, and realistic. Many data on energy use, fuel characteristics, and emissions are estimated or projected from 1970 to 2050.

The output has been cleaned up and presented in considerably more detail. Estimates of $g/10^6$ BTU emissions are presented for each GHG (without the CEF weighting), for each stage of all of the fuel cycles. Fuel cycle GHG emissions for electric vehicles are calculated for the U.S. and each of six regions. A macro runs the LEM for any target year and prints all of the main results in publication-ready tables.

Many major new components have been added, as listed below:

• Projections of energy use, emissions, emission control, and other parameters through the year 2050.

• Updated energy use parameters and emission factors, on the basis of EPA's standard emission-factor handbook (AP-42), the IPCC's *Revised 1996 IPCC Guidelines for National GHG Emission Inventories,* and other sources.

• Models and default data to represent emissions and energy use for other countries (e.g., Canada) (Appendix B).

• Detailed original calculations of CO_2-equivalency factors for all gases, including aerosols (black carbon, organic matter, secondary organic aerosols, sulfate and nitrate), CO, NMOCs, NO_X, and SO_X (Appendix D).

• A complete representation of the nitrogen cycle, including representation of nitrogen deposition and associated environmental effects (Appendices C and D).

• Several modes added: mini-cars, motor scooters, mini buses, heavy-rail transit, light-rail transit, and bicycling.

• PM and SO_2 added as greenhouse gases and urban air pollutants.

• Hydrogen addes as an indirect greenhouse gas; hydrogen leakage represented in detail.

• NMOCs weighted by their ozone-forming potential (Appendix D).

• A mobile-source emission factor model, akin to a highly simplified version of the EPA's MOBILE model.

• Review and update of CH_4 and N_2O emission factors for cars and power plants (Appendix F).

• Update and revision of the representation and data for the modeling of the lifecycle of materials (Appendix H).

• More detailed treatment of motor-vehicle energy use, on the basis of weight, thermal efficiency, and aerodynamic drag (treatment for electric vehicles documented in Appendix G).

• New estimation of the relationship between vehicle weight, materials composition, and fuel economy.

• Fuel economy estimated as a function of number (weight) of passengers in cars, buses, mini-buses, mini-cars, and scooters.

• Fuel economy and, hence, GHG emissions estimated as a function of vehicle payload, including number of passengers in cars or buses.

- Light-duty fuel cell vehicles using gasoline, methanol, ethanol, or hydrogen, with or without an auxiliary peak-power unit.
- A new model of refinery emissions, based on emissions from individual process areas.
- A more detailed calculation of emissions from the use of oxygenates.
- A much more detailed treatment emissions from corn/ethanol and wood bio-fuel cycles, including emissions from the combustion of residue.
- Perennial grasses as a feedstock for the production of ethanol.
- Soybeans to biodiesel fuel cycle (Appendix A)
- Natural gas to diesel fuel via the Fischer-Tropsch (F-T) process.
- F-T diesel and methanol made from associated natural gas that otherwise would be vented or flared.
- International transport of LNG imports.
- Natural gas to hydrogen via reforming.
- Coal to synthetic crude oil.
- Diesel fuel in LDVs, and gasoline in HDVs.
- Lifecycle emissions from the use of forklifts.
- Lifecycle emissions from the use of motor scooters.
- An option to specify HDVs as buses rather than trucks.
- A distinction between large-scale centralized liquefaction and small-scale liquefaction at service stations, for LNG and LH_2.
- A detailed analysis of energy used to manufacture agricultural chemicals (Appendix H).
- A model of changes in carbon sequestration in biomass and soil due to changes in land use (including changes associated with fossil-fuel production).
- More detailed representation of emissions of nitrogen species from soils, due to cultivation, and fertilizer use (Appendices C and D).
- Representation of feedstock production and fuel production in physical input/output terms.
- Detailed tracking of imports of crude oil, and venting and flaring emissions and refining emissions in individual exporting countries or regions.
- Tracking of imports and coal, and venting of coal bed methane in individual exporting countries or regions.
- Tracking of source of enrichment of uranium, with different energy intensities for different enriching countries.
- A detailed representation of natural gas transmission and distribution.
- Added explicit representation of international transport of coal.
- A more consistent and detailed representation of feedstock and fuel transport.
- Emissions from energy use by service stations and marketing facilities.
- Fuel cycle emissions from the use of NG, LPG, fuel oil, and electricity for space heating and water heating.
- Rudimentary treatment of the extent to which alternative-fuel production displaces existing production or stimulates new demand.

- A new treatment of “own use” of fuel.
- Explicit representation of geographic sources and shipping of materials and motor vehicles.

Overall, the present model is more powerful, and substantially easier to use, than the previous model. In general, the overall affect of the revisions is to make alternative fuels more attractive.

OUTPUT OF THE LEM

This section discusses the some of the outputs of the LEM in more detail.

Emissions per mile from the use of conventional and alternative transportation fuels for motor vehicles

The LEM estimates CO_2-equivalent emissions per mile for the motor-vehicle transportation fuel and feedstock combinations shown above. For baseline petroleum fuels (gasoline and diesel fuel), the results are reported as grams of individual gases or CO_2-equivalent emissions from each stage of the lifecycle of fuels. The lifecycle of fuels also include the manufacture and assembly of materials for vehicles, per mile of travel by the vehicle. For the alternative fuel vehicles, the results are reported in grams / mile as for gasoline and diesel vehicles, and also as a percentage change relative to the petroleum-fuel gram-per-mile baseline.

Emissions per energy unit from the use of electricity, and from end-use heating

The LEM calculates grams of individual gases and grams of CO_2-equivalent emission from the entire fuel cycle, per kWh of electricity delivered to end users. It analyzes coal, residual fuel oil, natural gas, methanol, nuclear, and hydro power plants, individually or in any combination. The analysis covers emissions from all stages of the fuel cycle, from feedstock recovery to scrubbing sulfur from flue gas to transmitting power via high-voltage lines, which can produce N_2O. The estimates of emissions of NO_x and SO_x account for the phase-in and effectiveness of emission controls. The gram / kWh emissions can be estimated for any power-plant efficiency, fuel mix, emission-control scenario, and time horizon.

The LEM also estimates lifecycle emissions from the use of NG, LPG, fuel oil, and electricity for space heating and water heating, in grams CO_2-equivalent emissions per 10^6 BTU of heat delivered.

Emissions by greenhouse gas

A new macro, called “Separate_gases”, calculates grams of each gas[1] emitted(), at every stage of every fuel cycle. A new table summarizes the resultant total upstream fuel cycle emissions by individual gas.

Another new table shows the contribution of each gas to total CO_2-equivalent emissions (this is comparable to Table 10 in DeLuchi [1991].) This table is filled in by the macro.

The “Separate_gases” macro works as follows: first, it sets all of the CEFs equal to 1.0, recalculates the model, and writes the results into a holding table. Then, it zeroes out the CEF for each gas, one at a time (leaving the other CEFs equal to 1.0), and takes the difference between the total with all CEFs set to 1.0 and the total with each gas zeroed out in turn. This difference is the weighted emissions contribution of each gas.

Results by emissions sector

Formerly, CO_2-equivalent g / mi emissions were presented *only* by “stage” of the fuel cycle:

- vehicle operation (fuel)
- fuel dispensing
- fuel storage and distribution
- fuel production
- feedstock transport
- feedstock and fertilizer production
- CH_4 and CO_2 gas leaks and flares
- emissions displaced by coproducts
- vehicle assembly and transport
- materials in vehicles
- lube oil production and use
- refrigerant (HFC-134a) use

Now, these results are mapped by stage into two different sectoral accounting frameworks.

First, a new set of tables maps the results calculated by “stage” of the fuel cycle (e.g., petroleum refining) into the emissions “sectors” used in the IPCC greenhouse-gas emissions-accounting frameworks. The IPCC sectors underlined in the table below comprise the fuel cycle stages used:

IPCC energy/road transport: fuels

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[1] CO_2, CH_4, N_2O, CO, NO_2, NMOCs, PM, SO_2, HFC-134a, and CFC-12.

Vehicle operation: fuel	*Note: This mapping includes credits for plant uptake of* CO_2*. Changes in soil and plant carbon are in "Land-use/forestry/agriculture".*
IPCC energy/industry: fuels	
Fuel dispensing	
Fuel storage and distribution	
Fuel production	
Feedstock transport	
Feedstock, fertilizer production	
CH_4 and CO_2 gas leaks, flares	*Note: related to fuel production and use.*
IPCC energy/industry: materials, vehicles	
Vehicle assembly and transport	
Materials in vehicles	
Lube oil production and use	
Refrigerant (HFC-134a)	
IPCC land-use/forestry/agriculture	
Land use changes, cultivation	*Note: this does not include any energy-related emissions (e.g., from fuel use by tractors).*
Not mapped to IPCC sectors:	
Emissions displaced by coproducts	

Second, a new macro ("Results_by_area") and another set of tables, maps the CO_2-equivalent emission results into six geographic sectors:

- the energy / road transport sector of the designated consuming country (the country selected for analysis; e.g., the U. S.);
- the energy / industry sector of the designated consuming country;
- the energy / industry sector of a selected major exporter (e.g., Canada) to the designated consuming country;
- the energy / industry sector of a second major exporter;
- international transport; and
- the rest of the world.

This mapping reveals how policies in one country affect emissions in other countries. International transport is a separate source because in the IPCC accounting it is not assigned to any country.

The macro works as follows. The LEM has a matrix of individual countries (plus "international transport"), as rows, and producer/consumer designations, as columns. The cells have zeros or ones, which determine whether the corresponding country is counted as a member of the corresponding producer/consumer category. These cell values are inserted as weights throughout the model, in the calculation of emissions associated with the production and transport of all major commodities traded internationally. For producing countries, the traded commodities are:

- crude oil
- petroleum products (PP)
- natural gas (NG)
- natural-gas liquids (NGLs)
- natural-gas-to-liquids (NGTLs)
- coal
- enriched uranium (separative work units, or SWUs)
- motor vehicles (MVs)
- aluminum
- steel and iron
- plastic
- other materials

The trading of each commodity is represented by a matrix, which shows, for each consuming country defined, the geographic distribution of the source of the commodity. (These trade matrices are discussed in the relevant commodity or process sections in the model documentation.) In essence, the zero/one "weights" mentioned in the preceding paragraph are applied within the trade matrix. The macro turns on and off the "weight" on a particular country and commodity in such a way that permits the calculation of the emissions attributable to the production or transport of that commodity from the particular country. The results are then aggregated to the six geographic sectors given above.

The following shows the producing countries and regions used in the model and the corresponding commodities produced:

Producing region or country	Commodity produced
U. S.	all
Canada	all except SWUs
Japan	SWUs, MVs, all materials
N. Europe	all except MVs, uranium
S. Europe	petroleum products, NG, NGTLs, all materials
Former Soviet Union	all except MVs
China	coal, SWUs
Korea	MVs, materials
Asian Exporters	all except SWUs, uranium, MVs
Venezuela	petroleum products, crude oil
North Africa (Algeria, Libya)	petroleum products, crude oil, NG, NGTLs
Nigeria	petroleum products, crude oil, NG (LNG)
Indonesia	coal, petroleum products, crude oil, NG, NGTLs
Persian Gulf	petroleum products, crude oil, NG, NGTLs
Malaysia	NG (LNG)
Caribbean Basin	petroleum products, crude oil, coal, NG (LNG)
Other	all
Mexico	crude oil, NG, NGTLs, MVs
France	SWUs, MVs
Germany	MVs, materials
Other Europe	MVs
Australia	coal, uranium, NG (LNG)
Colombia	coal
Poland, Czech Republic	coal
South Africa	coal, uranium
Other Middle East	crude oil
Other Africa	crude oil
Target developed (domestic)	all
Target LDC (domestic)	all
International transport	all except SWUs, uranium

(Note that in the case of coal: N. Europe = Germany and U. K.)

In all cases except some alternative fuels, the assignment of commodities to international transport is consistent with the assumptions regarding foreign production of the commodities. However, in the case of biomass feedstocks and fuels, LNG, and

LPG, it is possible to specify international transport without also having foreign production. This potential inconsistency exists because it is simple to model international transport, but more complicated to model foreign production. This is not considered significant for biomass, since there is not likely to be much international trade in biofuels.

EV emissions by region

Fuel cycle emissions for electric vehicles are calculated for the marginal mix of electricity in the entire U. S. (or Canada) and in each of six US regions— Northeast, East Central, South East, South Central, West Central, and West; as well as each of six Canadian regions— Quebec, Ontario, Manitoba, Alberta, Saskatchewan, and British Columbia). A macro command (“EVs_by_region”) runs the regional results.

Disaggregation of results

The revised LEM shows more disaggregated results than the 1993 version of the model. The fuel cycle stage formerly called “compression or liquefaction” now is called “fuel dispensing” and includes emissions from the use of energy to pump liquid fuels such as gasoline. (This pumping energy is new to the model; see the discussion in the main model documentation.) The stage formerly called “fuel distribution” now is called “fuel distribution and storage,” and includes emissions from the use of energy at bulk fuel-storage facilities.

The breakdown of energy use by fuel type, by stage and fuel cycle, formerly displayed as a single table (Table 4 of DeLuchi [1991]), has been split into three tables: one for feedstock processes (agricultural chemicals, feedstock recovery, and feedstock transport), one for fuel processes (fuel production, fuel distribution and storage, and fuel dispensing), and a separate breakout of fuel distribution and storage for individual petroleum fuels.

Formerly, the LEM displayed only net zero CO_2 emissions from biofuel vehicles. This net zero value was equal to total actual CO_2 emissions from biofuel combustion less the same amount assumed to be captured photosynthetically by the energy crops grown to make the biofuel. The revised LEM displays separately the total actual CO_2 emissions, the photosynthesis removal credit, and the net result (cf. Table B.2 of DeLuchi [1993]; Table 9 of DeLuchi [1991]). This is shown this way in the g/mi summary tables.

CO_2-equivalent emissions from changes in carbon sequestration in biomass and soils due to changes in land use are reported in a separate line in the g/mi summary tables (cf. Table 9 of DeLuchi [1991]).

The CO_2-equivalent GHG emissions displaced by the marketing of the co-products of fuel conversion processes [2] () now are shown as a separate line in the

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[2] e.g., the emissions associated with the corn feed that would have been used instead of the DDGS coproduct of the corn-to-ethanol process.

g / 10^6-BTU and g / mi tables (Tables 7, 9 and 10 of DeLuchi [1991]). The fuel conversion stage, which formerly showed "net" emissions equal to emissions from conversion less any emissions displaced by marketed coproducts, now shows the actual emissions from fuel production, with no credit for emissions displaced by marketed co-products.

Methanol from natural gas and methanol coal, and reformulated gasoline and conventional gasoline, have been separated into individually tracked fuel cycles.

BTU energy use per mile, and summary of percentage changes in g/mi emissions.

A new table shows BTUs of process and end-use energy used per mile of travel. Another new table summarizes all of the percentage changes in g / mi emissions, relative to the gasoline or diesel baseline.

One-step scenario analysis and table printout

A macro called "Print_results" has been added that runs the model for up to 40 different target years (any year from 1970 to 2050), and then prints the results, for each target year, in ready-to-publish tables. The user identifies which tables of results to be printed, then, for each results table (g / mi, g / 10^6-BTU, etc.), which target years are to be run. The macro runs the model for the first target year and table of results, sends the table to the printer, runs the next target year, sends the table to the printer, and so on, for each target year and table of results. The target year is printed in the title of each table.

The macro will run and print any of the following tables of results:

- fuel cycle CO_2-equivalent emissions from vehicles (g / mi);
- fuel cycle CO_2-equivalent emissions excluding end use (g / 10^6-BTU;
- Table 7 of DeLuchi [1991]); the energy intensity of fuel cycles (BTU-input / BTU-output;
- Table 3 of DeLuchi [1991]); the types of process fuel used in the fuel cycles (Table 4 of DeLuchi [1991]);
- fuel cycle CO_2-equivalent emissions from electricity generation (g / kWh); emissions of individual greenhouse gases (g / 10^6-BTU);
- input data and results regarding vehicle weight and energy use; summary of percentage changes in g / mi emissions; CO_2-equivalent g / mi emissions broken down by individual gas;
- CO_2-equivalent g / mi emissions mapped into IPCC sectors and geographic sectors, and results for fuels used for heating and cooking.

This macro calls other macros as necessary. For example, if the user wishes to print the g / mi results for EVs, by region, for different target years, the Print_results macro will call the macro EVs_by_region, for each target year. If the user wishes to print the g / 10^6-BTU results for individual greenhouse-gases, for different target years, the “Print_results” macro calls the “Separate_gases” macro. The “Separate_gases” and “Results_by_area” macros call the “EVs_by_region” macro automatically.

Analysis of emissions for other countries

The LEM can estimate emissions for countries other than the U. S. For most of the important parameters, such as fuel economy, vehicle emissions, efficiency of electricity generation, mix of fuels used to generate electricity, and leaks of natural gas, the user can enter data sets for up to 20 target countries (presently the U. S., Canada, Italy, India, China, and 16 “blanks”). The user then assigns weights, totaling 1.0, to the 20 countries, and the model applies the weights to the data sets for each of the 20 countries. In the U. S. base-case presented here, the U. S. data get a weight of 1.0. To run the model for another country, the user assigns a weight of 1.0 to the country of interest, and a weight of 0.0 to all other countries[3]. The row containing the country weights is a range called "Country_weights".

In several places, data for other countries are entered as part of an integrated representation of the international flow of key commodities. For example, the energy intensity of production of crude oil is entered for all major oil producing and exporting regions of the world. For any one of the 20 countries that can be selected for analysis, the production intensities of the producing countries are then weighted according to their contribution to the oil supply of the [consuming] country selected for analysis. Data pertinent to international flows that might involve the U. S. are discussed in the appropriate sections in the main text below.

The LEM has the following country-specific parameters, which are discussed in Appendix B:

Motor vehicles (conventional)	City fuel economy, highway fuel economy, and city fraction of total VMT, by vehicle type (light-duty vehicles, heavy-duty trucks, and buses)
Motor vehicles (conventional)	Emissions by pollutant (relative to emissions from US vehicles) and vehicle type (light-duty vehicles and heavy-duty vehicles)
Motor scooters	Fuel economy and emissions by pollutant, relative to US values
Mini cars (up to 500 kg)	fuel economy and emissions by pollutant, relative to US values
Motor vehicles (all)	Lifetime to scrappage
Rail transit (heavy rail and light rail)	Capacity factors, BTUs / capacity-mile for traction energy, BTUs / capacity-mile for station energy, and energy for construction relative to energy for traction

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[3] Although the LEM will work with a combination of fractional weights on several countries, such as 0.50 U. S. and 0.50 Canada, the meaning of such fractional weights is not clear. In the future I intend to define the weights so that combinations are meaningful.

Electricity generation	Generation efficiency by type of fuel (efficiency in a base year, and percent change in efficiency per year)
Electricity generation	Generation fuel mix for EV recharging, crop-ethanol production, biomass-ethanol production, operation of rail transit, water electrolysis (for hydrogen production), and generic power
Electricity generation	Efficiency of emission controls, by pollutant, relative to US values
Diesel fuel sulfur	Estimated in ppm for various years between 1970 and 2050, for highway, offroad, and heating fuels
Other fuel quality	Sulfur content of coal and various petroleum products, relative to that in the U. S.
Material flows	Imports of materials, transport distances, and transport modes, specified by material (iron, aluminum, plastic, and other materials), consuming country, and producing region
Oil flows	Imports of petroleum, transport distances, and transport modes, specified by type of petroleum (crude oil, light products, heavy products), consuming country, and producing region
Coal flows	Imports of coal, transport distances, and transport modes, specified by consuming country and producing region
Natural-gas flows	Imports of natural gas, transport distances, and transport modes, specified by consuming country and producing region
Natural gas	Leakage from domestic distribution systems
Motor-vehicle flows	Imports of vehicles, transport distances, and transport modes, specified by vehicle type (light-duty vehicles and heavy-duty vehicles), consuming country, and producing region
Uranium enrichment	Source of “separative work units” (SWUs) provided to consuming countries by SWU-producing countries, SWUs per MWh generated, and tons of enriched uranium per GWh generated
Agriculture	Crop production and fertilizer use: harvest yield in base year, change in harvest yield per year, rate of nitrogen use, and distribution of land types displaced, by crop type (corn, soy, grass, and wood).

Fuel production	Corn-ethanol production, energy use: Total BTUs / gallon, electricity use, and fuel type
Nitrogen deposition	Distribution of land types affected by deposition, by country.
Multi-modal emissions	Parameters for the estimation of emissions per passenger / mi and emissions per ton-mi for multi-modal transportation policies: vehicle occupancy by mode (passenger cars, motor-scooters, mini-cars, bicycles, minibuses, and buses), capacity fractions for rail heavy and light rail, passenger-miles of travel by mode (light-duty vehicles, buses, minibuses, minicars, and motor scooters by fuel type, including a wide range of alternative fuels and electric vehicles, heavy rail, light rail, bicycling, and walking), and tons and miles of travel by freight mode (large and medium diesel, CNG, and ethanol trucks, diesel trains, cargo ships, tankers, barges, and pipelines)

As a general rule, fuel qualities, CO_2-equivalency factors, land-use impacts, and energy intensity and emissions of new technologies have been assumed to be the same in all countries.

Analysis of emissions from complete transportation scenarios

The LEM estimates total average emissions per passenger-mile and per freight ton-mile from a complete transportation scenario. A complete transportation scenario includes passenger transport and freight transport by all modes, where the modal shares and other characteristics of the modes are specified by the user.

The passenger travel modes are:

- conventional motor vehicles,
- alternative-fuel vehicles (including electric vehicles)
- mini-cars (conventional and alternative-fuel)
- scooters
- buses (conventional and alternative-fuel)
- trolleys
- subways
- bicycles and walking

The freight modes include heavy-duty and medium–duty trucks (conventional and alternative-fuel), rail, cargo ship, tanker, barge, and pipeline. The user specifies the amount of passenger mile of travel or freight ton-mile of travel by each detailed mode. The user also specifies the occupancy and in some cases the efficiency of the mode.

In the case of the passenger-transport scenario analysis, the data for this calculation are: g / vehicle-mile emissions for each mode, calculated by the model; average persons / vehicle for each mode, input by the user; and each mode's share of total passenger-miles of travel, also input by the user. Formally:

$$GHGPMI = \sum_{M} \frac{GHGVMI_{M}}{OCC_{M}} \cdot MS_{M} \qquad \text{eq. 1}$$

where:

Subscript "M" = the various technology specific modes (e.g., electric scooters, 4-stroke gasoline scooters and others as in the list above);
GHGPMI = scenario-average CO_2-equivalent GHG emissions per passenger mile;
GHGVMI = CO_2-equivalent GHG emissions per vehicle mile from mode M (calculated by the LEM);
OCC_M = the average occupancy of mode M (person / vehicle; input by the user);
MS_M = the modal share of mode M (equal to person-miles of travel by mode M divided by total person-miles of travel by all modes; input by the user).

Format of output

The column headings in the main summary tables and in other tables throughout the model are formatted to automatically show the following:

- the fuel specification (methanol %, ethanol %, reformulated gasoline %, oxygenate %, propane %, butane %, low-sulfur diesel, LNG or CNG, or LH_2 or CH_2);

- the characteristics of the feedstock (oil, coal, natural gas, natural-gas liquids, refinery byproducts, corn, or wood);

- the mix of the process energy used for boiler fuel or power generation (coal, natural gas, fuel oil, biomass, nuclear power, or solar power); and

- in some cases, the year of the analysis.

The user specifies the fuel and feedstock characteristics in the appropriate input places in the LEM, and these input characteristics are automatically inserted in all of the relevant headings throughout the LEM. For example, if you specify that an ethanol-fuel vehicle uses 100% ethanol made from corn, and that the ethanol plant uses coal for process heat, the column headings in the tables of results will read: “Ethanol, E100 corn, C100 / NG0 / B0”. If you change the fuel to 85% ethanol / 15% gasoline, and change the process fuel to 50% natural gas / 50% corn stover, the headings will automatically change to “Ethanol, E85 corn, C0 / NG50 / B50”, where the “B50” part means “50% biomass”. The same automatic labeling happens for the other fuel / feedstock combinations.

Comparison of the LEM with other recent modeling efforts

The structure and coverage of the LEM can be compared with that of several other recent modeling efforts:

Project	**GM -ANL U. S.**	**GM –LBST Europe**	**MIT 2020**	**EcoTraffic**	**LEM**
Region	North America	Europe	based on U. S. data	generic, but weighted towards European conditions	multi-country (primary data for U. S.; other data for up to 30 countries)
Time frame	near term (about 2010)	2010	2020	between 2010 and 2015	any year from 1970 to 2050
Transport modes	LDV (light-duty truck)	LDV (European mini-van)	LDV (mid-size family passenger car)	LDVs (generic small passenger car)	LDVs, HDVs, buses, light-rail transit, heavy-rail transit, minicars, scooters, offroad vehicles
Vehicle drivetrain type	ICEVs, HEVs, BPEVs, FCEVs	ICEVs, HEVs, FCEVs	ICEVs, HEVs, BPEVs, FCEVs	ICEVs, HEVs, FCEVs	ICEVs, BPEVs, FCEVs
Fuels	gasoline, diesel, naptha, FTD, CNG, methanol, ethanol, CH2, LH2, electricity	gasoline, diesel, naptha, FTD, CNG, LNG, methanol, ethanol, CH2, LH2	gasoline, diesel, FTD, methanol, CNG, CH2, electricity	gasoline, diesel, FTD, CNG, LNG, methanol, DME, ethanol, CH2, LH2	gasoline, diesel, LPG, FTD, CNG, LNG, methanol, ethanol, CH2, LH2, electricity
Feedstocks	crude oil, natural gas, coal, crops, ligno-cellulosic biomass, renewable and	crude oil, natural gas, coal, crops, ligno-cellulosic biomass, waste, renewable and	crude oil, natural gas, renewable and nuclear power	crude oil, natural gas, ligno-cellulosic biomass, waste	crude oil, natural gas, coal, crops, lignocellulosic biomass, renewable and

	nuclear power	nuclear power			nuclear power
Vehicle energy-use modeling, including drive cycle	GM simulator, U. S. combined city / highway driving	GM simulator, European Drive Cycle (urban and extra-urban driving)	MIT simulator, U. S. combined city / highway driving	Advisor (NREL simulator), New European Drive Cycle	simple model based on SIMPLEV-like simulator, U. S. combined city / highway driving
Fuel LCA	GREET model	LBST E^2 I/O model and data base	literature review	literature review	detailed model
Vehicle lifecycle	not included	not included	detailed literature review and analysis	not included	detailed literature review and analysis
GHGs [CEFs]	CO2, CH4, N2O [IPCC] (other pollutants included as non-GHGs)	CO2, CH4, N2O [IPCC]	CO2, CH4 [IPCC]	none (energy efficiency study only)	CO2, CH4, N2O, NOx, VOC, SOx, PM, CO [own CEFs, also IPCC CEFs]
Infra-structure	not included	not included	not included	not included	very simple representation
Price effects	not included	not included	not included	not included	a few simple quasi-elasticities
Reference	GM, ANL et al. (2001)	GM et al. (2002a, 2002b, 2002c)	Weiss et al. (2000)	Ahlvik and Brandberg (2001)	this report and appendices

Project	**ADL AFV LCA**	**CMU I/O LCA**	**Japan CO2 from AFVs**	**LEM**
Region	United States	United States	Japan	multi-country (primary data for U. S.; other data for up to 30 countries)
Time frame	1996 baseline, future scenarios	near term	near term?	any year from 1970 to 2050

Transport modes	subcompact cars	LDVs (midsize sedan)	LDVs (generic small passenger car)	LDVs, HDVs, buses, light-rail transit, heavy-rail transit, minicars, scooters, offroad vehicles
Vehicle drivetrain type	ICEVs, BPEVs, FCEVs	ICEVs	ICEVs, HEVs, BPEVs	ICEVs, BPEVs, FCEVs
Fuels	gasoline, diesel, LPG, CNG, LNG, methanol, ethanol, CH2, LH2, electricity	gasoline, diesel, biodiesel, CNG, methanol, ethanol	gasoline, diesel, electricity	gasoline, diesel, LPG, FTD, CNG, LNG, methanol, ethanol, CH2, LH2, electricity
Feedstocks	crude oil, natural gas, coal, corn, ligno-cellulosic biomass, renewable and nuclear power	crude oil, natural gas, crops, ligno-cellulosic biomass	crude oil, natural gas, coal, renewable and nuclear power	crude oil, natural gas, coal, crops, lignocellulosic biomass, renewable and nuclear power
Vehicle energy-use modeling, including drive cycle	Gasoline fuel economy assumed; AFV efficiency estimated relative to this	Gasoline fuel economy assumed; AFV efficiency estimated relative to this	none; fuel economy assumed	simple model based on SIMPLEV-like simulator, U. S. combined city / highway driving
Fuel LCA	Arthur D. Little emissions model, revised	own calculations based on other models (LEM, GREET..)	values from another study	detailed model
Vehicle lifecycle	not included	Economic Input-Output Life Cycle Analysis software (except end-of-life)	detailed part-by-part analysis	detailed literature review and analysis
GHGs [CEFs]	CO2, CH4, [partial GWP] (other	CO2, CH4, N2O? [IPCC] (other	CO2	CO2, CH4, N2O, NOx, VOC, SOx, PM,

	pollutants included as non-GHGs)	pollutants included as non-GHGs)		CO [own CEFs, also IPCC CEFs]
Infra-structure	not included	not included	not included	very simple representation
Price effects	not included	not included (fixed-price I/O model)	not included	a few simple quasi-elasticities
Reference	Hackney & de Neufville (2001)	MacLean et al. (2000)	Tahara et al. (2001)	this report and appendices

The terms in this table are defined as follows:

Region	The countries or regions covered by the analysis.
Time frame	The target year of the analysis.
Transport modes	The types of passenger transport modes included. LDVs = light-duty vehicles, HDVs = heavy-duty vehicles.
Vehicle drivetrain type	ICEVs = internal combustion-engine vehicles, HEVs = hybrid-electric vehicles (vehicles with an electric and an ICE drivetrain), BPEVs = battery-powered electric vehicles (BPEVs), FCEVs = fuel-cell powered electric vehicles.
Fuels	Fuels carried and used by motor vehicles. FTD = Fischer-Tropsch diesel, CNG = compressed natural gas, LNG = liquefied natural gas, CH2 = compressed hydrogen, LH2 = liquefied hydrogen, DME = dimethyl ether.
Feedstocks	The feedstocks from which the fuels are made.
Vehicle energy-use modeling	The models or assumptions used to estimate vehicular energy use (which is a key part of fuelcycle CO_2 emissions), and the drive cycle over which fuel usage is estimated (if applicable).
Fuel LCA	The models, assumptions, and data used to estimate emissions from the lifecycle of fuels.
Vehicle lifecycle	The lifecycle of materials and vehicles, apart from vehicle fuel. The lifecycle includes raw material production and transport,

	manufacture of finished materials, assembly of parts and vehicles, maintenance and repair, and disposal.
GHGs and CEFs	The pollutants (greenhouse gases, or GHGs) that are included in the analysis of CO_2-equivalent emissions, and the CO_2-equivalency factors (CEFs) used to convert non-CO_2 GHGs to equivalent amount of CO_2 (IPCC = factors approved by the Intergovernmental Panel on Climate Change [IPCC]; my CEFs are those derived in Appendix D).
Infrastructure	The lifecycle of energy and materials used to make and maintain infrastructure, such as roads, buildings, equipment, rail lines, and so on. (In most cases, emissions and energy use associated with the construction of infrastructure are smalled compared with emissions and energy use from the end use of transportation fuels.)
Price effects	This refers to the relationships between prices and equilibrium final consumption of a commodity (e.g., crude oil) and an "initial" change in supply of or demand for the commodity or its substitutes, due to the hypothetical introduction of a new technology or fuel.

The study by EcoTraffic (Ahlvik and Brandberg, 2001) provides a good comparison of their work with the GM WTW U. S. (GM et al., 2001), the MIT 2020 (Weiss et al. 2000), and several other studies.

Among the tools used in the studies in the table above, those used in the GM WTW studies are most similar to the LEM. In particular, the GREET model is similar to the fuel lifecycle parts of the LEM. (See Wang [1999] for documentation of the GREET model.) Even so, there are significant differences. Generally, the LEM is much broader in scope than the GM studies: it covers more countries, wider time frames, more transport modes, more pollutants, more aspects of the lifecyle (such as materials), and more relevant effects (such as price effects). One significant exception is that the GM studies, and indeed other studies listed in the table above, include a vehicle type (hydrid EVs) and some fuel pathways (such as fuels from waste) not included in the LEM.

My examination of the available documentation for the GREET model and the LBST E^2 I/O model (used in the GM WTW European study) indicates that, apart from the differences noted in the table above, the fuel lifecycle parts of the LEM are in some cases more detailed than are the GREET and E^2 models. For example, the LEM includes a more detailed carbon tracking (apportioning carbon between fuel, lubricating oil, biomass and non-biomass components) than do other models. More significantly, the LEM has a more comprehensive and detailed treatment of emissions associated with cultivation and land-use change. The LEM also uses complete, detailed input-output relationships, usually based on primary data (rather than secondary citation of literature), for every stage of the fuelcycle.

The comparison above covers only major, original, recent analyses of lifecycle emissions from a wide range of alternative transportation fuels. It does not include the following:

- older LCAs of alternative transportation fuels (see DeLuchi [1991] for a discussion of studies done before 1990, and Wang [1999] for a discussion of studies done in the 1990s);
- studies that are entirely derivative;
- studies of a single fuel or narrow range of transportation fuels (e.g., Marquevich et al., 2002; Sheehan et al., 1998; Kadam et al., 1999; SAIC, 2001; Spath and Mann, 2001; Pricewaterhouse Coopers LLP, 2003; Hill and Villaneuva, 1995);
- studies that focus mainly on the lifecycle of the automobile as opposed to automotive fuels (e.g., Sullivan et al., 1998; see Appendix H for more discussion pertinent to these analyses);
- LCAs not directly related to transportation (of which there area great many, for a wide range of non-transportation products and system, including power generation, building materials, and more).

It should be emphasized that many of these studies not covered here, and particularly some of those that focus on a single fuel or a narrow range of fuels, are of high quality. I have omitted them simply to keep my comparison manageable. It is also worth noting that many of the non-transportation LCAs and some of the transportation LCAs follow guidelines established by the International Organization for Standardization (ISO). The ISO's guidelines for LCA are laid out in ISO standards 14040 to 14049 (see the ISO web site, www.iso.ch/iso/en/iso9000-14000/iso14000/iso14000index.html). These guidelines reflect but generally do not advance the state of the art in lifecycle analysis.

PROJECTIONS OF ENERGY USE AND EMISSIONS

In the previous version of the LEM, the input data were for a single year -- in the base case, the year 2000. To analyze another year, the user had to estimate and input a separate set of data for the year of interest. This made it difficult to do multi-year analyses.

As part of a major revision to the LEM, projections of energy use and emissions, or changes in energy use and emissions, for the period 1970 to 2050 have been added. The user now specifies any year between 1970 and 2050, and the model looks up or calculates energy-use intensities, emission factors, or other data for the specified year. The actual projections are discussed below, in the pertinent subject areas.

There are several different kinds of projections in the LEM. One type of projection uses look-up tables based on energy-use or emissions projections from the EIA. Another type of projection uses constant percentage changes per year, logistic functions with upper *or* lower limits, and logistic functions with upper *and* lower limits.

Look-up tables

Look-up tables have data values such as energy use, emissions, etc. for each year from 1990 to 2020, and calculated values (based on a specified percentage change per year) for any target year between 2021 and 2050. The LEM simply looks up the data value corresponding to the user-specified target year.

Most of the data for the years 1990 to 1999 are from the Energy Information Administration's (EIA's) *Annual Energy Review (AER).* Presently, the LEM has one value for the period 1970-1989, equal to the actual 1990 value[4]. The data in the look-up tables for the years 1999 to 2020 are from the EIA's *Annual Energy Outlook (AEO)*, reference-case scenarios (available as spreadsheet files from the EIA's web site www.eia.doe.gov). The most recent *AEO*[5] projects to 2020, but presumably within a few years the EIA will extend the projection to 2025 and eventually to 2030. The data values in the look-up tables for the years beyond the EIA projections are currently equal to the values for the last projection year multiplied by a user-specified percentage change per year:

$$V_T = V_{LPY} \cdot \left(1 + \frac{PCY}{100}\right)^{T - T_{LPY}} \qquad \text{eq. 2}$$

where:

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[4] In future models, actual values for 1970-1989 will be added.

[5] In this report, a reference to the *AEO* without a date means the most recent *AEO* available at the time of writing.

V_T = the value of the projected energy or emission parameter in the target year T.

V_{LPY} = the value of the projected energy or emission parameter in the last projection year (LPY).

PCY = the percentage change per year (PCY) in the value of the energy or emissions parameter, beyond the last projection year.

T = the target year of the analysis

T_{LPY} = the last projection year

The LEM has EIA *AEO* projections for the U. S. for the following (EIA projections are available as Lotus WK1 files from www.eia.doe.gov/oiaf/aeo/supplements/index.html), or www.eia.doe.gov/oiaf/aeo/results.html:

-- electricity generation by fuel, consumption of fuel by electric generators, and emissions from electric generators (Table 72 in EIA *AEO* supplemental data)

-- refinery industry energy consumption (Table 24 in EIA *AEO* supplemental data)

-- coal production by region and type (Table 111 in EIA *AEO* supplemental data)

-- natural gas supply and disposition (Table 13 in EIA *AEO* reference case spreadsheet)

-- petroleum supply and disposition (Table 11 in EIA *AEO* reference case spreadsheet)

-- imported petroleum by source (Table 117 in EIA *AEO* supplemental data)

The percentage change per year (parameter PCY in Eq. 2) beyond the last projection year (parameter V_{LPY} in Eq. 2) were estimated on the basis of the trend evident in the last 10 years of the EIA *AEO* projections, and judgment. Generally, I assumed that the PCY is slightly less than the percentage change of the EIA projections in the last 10 years if the EIA percentage change is relatively large (at least about 0.8%/year), and approximately equal to the EIA percentage change if the EIA percentage change is relatively small (no more than about 0.5%/year). In the case of projecting petroleum supply and disposition, the Office of Transportation Technologies (2001) 50-year projections of U. S. oil use were also considered.

Constant percentage change per year

If reliable year-by-year projections are not available, the model calculates future values on the basis of a few user-specified parameters. In the simplest case, the user specifies a base-year value, a base year, the percentage change in the value per year, and the target year, and the model uses these values to calculate the value in the target year.

Logistic function with lower *or* upper limits

In cases in which there are natural or practical lower *or* upper limits to a parameter value, within the time frame of the analysis, the parameter value is assumed to follow a logistic path over time, which asymptotically approaches an upper or lower limit. The user must specify the upper or lower limit of the parameter value, a base value in a base year, and a shape or "steepness" parameter.

A single-sided logistic function is shown in Figure 1. The formula for a function with an upper limit V_U approached going forward in time is:

$$V_T = V_U - \left(V_U - V_{T_B}\right) \cdot e^{-k \cdot (T - T_B)} \qquad \text{eq. 3}$$

where:

V_T = the value of the projected energy or emissions parameter in the target year T.
V_U = the upper limit of the projected parameter value, approached asymptotically.
V_{TB} = the value of the projected energy or emissions parameter in the base year $T_{B.}$
k = the shape (steepness) factor (the greater the value of k, the steeper the function).
T = the target year of the analysis.
T_B = the base year.

The formula for a function with a lower limit V_L approached going forward in time is:

$$V_T = V_L + \left(V_{T_B} - V_L\right) \cdot e^{-k \cdot (T - T_B)} \qquad \text{eq. 4}$$

Reversing the sign of the exponent k in Equations3 and 4 flips the function around, so that the upper or lower limit is approached going backward in time.

Logistic function with lower *and* upper limits

In cases in which there are natural or practical lower *and* upper limits to a parameter value within the time frame of the analysis, the parameter value is assumed to follow a logistic path over time, between the upper and lower limits. The user must specify the upper and lower limits of the parameter value, a base value in a base year, and a value for k, the shape parameter.

The double-sided logistic function is shown in Figure 2. The formula for the function is:

$$V_T = V_L + \frac{V_U - V_L}{1 + e^{-k \cdot (T - T^*)}} \qquad \text{eq. 5}$$

where:

V_T, V_L ,V_U, k, and T_T are as defined for Equation 3.
T* = the year at which the parameter value is halfway between the upper and lower limits.

The mid-value time T* can be calculated by specifying a base-year parameter value (V_B) in the base year T_B:

$$V_B = V_L + \frac{V_U - V_L}{1 + e^{-k\cdot(T_B - T^*)}}$$

$$\frac{V_B - V_L}{V_U - V_L} = \frac{1}{1 + e^{-k\cdot(T_B - T^*)}}$$

$$e^{-k\cdot(T_B - T^*)} = \frac{V_U - V_L}{V_B - V_L} - 1$$

$$-k\cdot\left(T_B - T^*\right) = \ln\left(\frac{V_U - V_L}{V_B - V_L} - 1\right)$$

$$T^* = T_B + \frac{\ln\left(\frac{V_U - V_L}{V_B - V_L} - 1\right)}{k}$$

To obtain an expression in terms of user-specifiable parameters, we substitute the expression for T* into Equation 5:

$$V_T = V_L + \frac{V_U - V_L}{1 + e^{-k\cdot\left(T - T_B - \frac{\ln\left(\frac{V_U - V_L}{V_B - V_L} - 1\right)}{k}\right)}}$$

$$= V_L + \frac{V_U - V_L}{1 + e^{-k\cdot(T - T_B)}\cdot\left(\frac{V_U - V_L}{V_B - V_L} - 1\right)}$$

$$= V_L + \frac{V_U - V_L}{1 + e^{-k\cdot(T - T_B)}\cdot\left(\frac{V_U - V_B}{V_B - V_L}\right)} \qquad \text{eq. 6}$$

Equation 6 is the general form of the double-bounded logistic function used in this analysis, corresponding to Figure 2. In the case of a parameter decreasing rather than increasing with time (Figure 2 shows an increasing function), the sign on the exponent k is reversed. For a function bounded by 0 and 1, Equation 6 reduces to:

$$V_T = \frac{1}{1 + e^{-k \cdot (T - T_B)} \cdot \left(\frac{1 - V_B}{V_B} \right)} \qquad \text{eq. 7}$$

Note: Equations 8 to 24 have been deleted from this text and moved instead to a separate appendix.

FUELS

Sulfur content of diesel fuel

The sulfur content of diesel fuel has a significant impact on emissions of particulate matter. Over the past decade the sulfur content of highway diesel fuel has decreased dramatically, and current regulations call for further major decreases. In the U. S., the sulfur content of highway diesel decreased from about 3000 ppm (mass basis) to 500 ppm in the 1990s, and is slated to decrease to 15 ppm in 2006. Diesel for offroad uses contains about 3300 ppm sulfur (Beardsley and Lindhjem, 1998b).

In the previous model, all diesel fuel, for all end uses, was assumed to have the same sulfur content. Now, the LEM distinguishes diesel fuel for highway vehicles, diesel fuel for offroad use, and diesel fuel for commercial and residential heating. For each kind of end use, the user specifies the sulfur content (in ppm, mass basis) in various years over the entire projection period (1970-2050). (This is done for every target country in the analysis.) Because the sulfur content of highway diesel fuel has changed since 1990, and will continue to change through at least 2010, the model asks the user to specify the sulfur content every five years from 1990 to 2010.

For the U. S., the sulfur content (ppm) is specified as follows:

Highway diesel fuel

1970	1990	1995	2000	2005	2010	2020
3300	*2300*	*340*	*320*	*12*	*12*	*12*

Offroad

1970	2000	2010	2020
3300	*3300*	*320*	*12*

Heating

1970	2000	2010	2020
3300	*2300*	*340*	*12*

The sulfur content input to the model is the actual in-use sulfur content, not the maximum allowable sulfur content. In order to ensure that the sulfur content never exceeds the maximum allowable, refineries produce fuels with a sulfur content well below the maximum. For example, the current maximum allowable sulfur content of highway diesel is 500 ppm, but in-use highway diesel actually contains about 320 ppm sulfur (Beardsley and Lindhjem, 1998b). Estimates for off-road diesel fuel content are based in part on data in the Federal Register (2003).

For the purpose of calculating sulfur emissions from fuel combustion, the model looks up the sulfur content in the target year for the target country, for each of the three end uses (highway, offroad, heating).

For the purpose of calculating emissions from petroleum refineries, the model first estimates the emissions attributable to manufacturing a reference conventional diesel fuel (CD) with 5000 ppm S and a reference ultra-low-sulfur diesel (ULSD) with 5 ppm S, then weights the estimated ULSD and conventional emissions in accordance with the actual looked-up sulfur content (for the target year and target country) relative to the sulfur content of the reference ULSD. Formally, the weighting factors applied to the emissions estimated for the reference CD and the reference ULSD are estimated by assuming the following relationship:

$$SF_D = SFW \cdot SF_{ULSD} + (1 - SFW) \cdot SF_{CD} \qquad \text{eq. 25}$$

where:
SF_D = the sulfur content of the user-specified diesel fuel.
SFW = the weighting factor applied to the reference ULSD.
SF_{ULSD} = the sulfur content of the reference ultra-low-sulfur diesel (5 ppm).
SF_{CD} = the sulfur content of the reference conventional diesel (5000 ppm).

Solving for the weighting factor, SFW, results in:

$$SFW = \frac{SF_D - SF_{CD}}{SF_{ULSD} - SF_{CD}} \qquad \text{eq. 26}$$

This weighting factor is applied to emissions and energy use estimated for the reference ULSD, and 1-SFW is applied to emissions and energy use estimated for the reference CD.

Composition and sulfur content of gasoline

In this version of the LEM, reformulated gasoline can be characterized in more detail, and greater accuracy, than in the previous version. The user can specify any volumetric mixture of alkanes, aromatics, olefins, ETBE, MTBE, methanol, and ethanol. The LEM calculates the carbon content, density, and heating value of the specified gasoline. On the basis of new data from the Auto / Oil Air Quality Improvement Research Program (1995, 1997), the specification of conventional and reformulated gasoline has been changed in the model (Table 3). The base-case reformulated gasoline uses MTBE rather than ethanol as an oxygenate partly because it appears to offer greater emissions reductions (*New Fuels and Vehicles Report*," 1998).

Presently, the EPA is considering proposals to reduce the sulfur content of gasoline. Sulfur appears to reduce the effectiveness of the 3-way catalytic converter. Tests reported by Walsh (1998b) show that use of gasoline with only 40 ppm sulfur dramatically lowers emissions compared to gasoline with 150 or 330 ppm sulfur (% change vs 330 or 150 ppm):

	NMHC		NOx	
	330 ppm	*150ppm*	*330ppm*	*150ppm*
LDVs and LDTs class 1	-30	-21	-58	-40
LDTs class 2, 3	-20	-19	-40	-25

An Auto / Oil Air Quality Improvement Research Program (1997) study also found that reducing sulfur from 320 ppm to 30 ppm reduces emissions, although the percentage changes are much smaller than those reported by Walsh. A study by the Coordinating Research Council (1998) found that under some conditions, the deterioration in emissions caused by using high-sulfur fuel was not fully reversible upon switching back to low-sulfur fuel.

Conventional gasoline contains about 340 ppm sulfur, and Federal reformulated gasoline (RFG) about 240 ppm sulfur. The sulfur level of RFG has been assumed to decline logistically towards a lower limit of 30 ppm, with an upper limit of 340 ppm. The parameter values for Eq. 6 are:

V_U = the upper limit = 340 ppm
V_L = the lower limit = 30 ppm
V_{TB} = the base-year value = 236 ppm (the current value)
k = the shape or steepness factor = -0.9 (a steep decline)
T_B = the base year = 2000

With these values, the sulfur content of gasoline drops from 236 to 30 ppm in about 8 years, or by 2008.

Oxygenates in reformulated gasoline

The LEM now calculates in complete detail the greenhouse-gas impact of oxygenates added to gasoline. It considers three different kinds of oxygenates:

1) methanol or ethanol added directly to gasoline;
2) methanol or ethanol plus isobutylene made from field butanes in NGL plants, made into MTBE or ETBE additive; and
3) methanol or ethanol plus isobutylene made from crude oil in refineries, made into MTBE or ETBE additive.

There are three main parts to the calculation of GHG impact of oxygenates. First, the model calculates the mass (not volume) of crude oil displaced by methanol or ethanol added directly or embedded in MTBE or ETBE, and by isobutylene derived from butanes from NGL plants. The LEM then reduces the mass of crude oil that must be recovered, transported and refined to make a unit of gasoline. (Chemical properties for alcohols and ethers are from the *CRC Handbook of Chemistry and Physics* [1975].) This reduction in the amount of crude oil that must be recovered, transported, and refined, per unit of gasoline produced, reduces fuel cycle CO_2-equivalent emissions attributable to reformulated gasoline.

Stork and Singh (1995) modelled how much of the isobutylene in ETBE and MTBE will be derived from NGLs, and how much will be derived from crude oil, for several scenarios regarding the composition of reformulated gasoline. (They make separate estimates for summer gasoline and winter gasoline.) On the basis of their estimates, 7.5% of the isobutylene used to make ETBE, and 5% of the isobutylene used to make MTBE is assumed to come from crude oil.

Second, the model estimates complete fuel cycle CO_2-equivalent emissions from the production and transport of butanes (used to make isobutylene) from NGL plants, and from the production and transport of methanol or ethanol added directly or made into MTBE or ETBE. Methanol, either added directly or made into MTBE, comes from NG or coal in proportions specified by the user. (In the base case, I assume all methanol comes from NG.) Ethanol, either added directly or made into ETBE, comes from corn or cellulosic biomass in proportions specified by the user. In the base case, all of the ethanol is assumed to come from corn until the year 2004, after which the share of ethanol from cellulosic biomass increases by 4 absolute percentage points per year. Also, any crude-oil derived butane (used to make isobutylene in the third kind of oxygenate above) is assumed to have been produced anyway and used in conventional gasoline and, hence, would not change refinery use of energy or crude oil compared to the conventional gasoline baseline.

Finally, the model estimates and adds emissions from the manufacture of MTBE or ETBE -- that is, from the conversion of butanes and alcohols to MTBE or ETBE. (This last step might involve some minor double counting, because the energy required to convert butanes to isobutylene might be included already in the baseline estimates of energy use by refineries making conventional gasoline.) Presently the model assumes (Stork and Singh, 1995)

Process energy	*ETBE*	*MTBE*
Million BTU-NG / gallon (including NG for steam)	0.00718	0.00709
Million BTU-fuel gas / gallon (credit)	0.00815	0.00803
Million BTU-electricity / gallon	0.00039	0.00039

The model is based on the information in Singh and McNutt (1993) and Stork and Singh (1995), with three refinements: First, as mentioned above, the model estimates complete fuel cycle emissions from the production of oxygenates and oxygenate components. For example, it includes emissions from the use of electricity to produce the natural gas from which the butane used to make isobutylene is derived. Second, the model calculates chemical properties of mixtures and components from primary data on characteristics of organic compounds. Third, the model gives an emissions credit for the use of the fuel gas that is a byproduct of MTBE and ETBE production. I assume that this fuel-gas is composed partly of the hydrogen that must be removed from the butane to make isobutylene, and partly of the leftover butane or butane derivatives (more butane is consumed than is needed for the reaction stoichiometry). This suggests that the fuel-gas byproduct, which is considerable, is rather like refinery gas. Therefore, the fuel-gas byproduct is assumed to displace refinery gas.

Recently, Hesse et al. (1993) have described an integrated plant which produces ethanol, methanol, ETBE, and MTBE from input corn and butane. The butane is made into isobutane and then into isobutylene, and the corn is made into ethanol. The CO_2 off-gas from ethanol production and the hydrogen off-gas from isobutylene production are combined to make methanol. The methanol and ethanol can be combined with the isobutylene to make MTBE or ETBE. This process probably results in lower CO_2 emissions than the conventional process assumed above because the methanol is made from CO_2 from the corn section that otherwise would be vented.

Kadam et al. (1999) also model the manufacture of MTBE and ETBE.

Density of diesel fuel and conventional gasoline

Emissions of CO_2 from the use of a fuel are essentially proportional to the carbon / BTU content of the fuel, which is calculated from the fuel density, carbon content, and heating value. For any particular kind of fuel, such as gasoline, these three parameters are related, and hence in principle should not be specified or changed independently. For example, one should not change the assumption regarding fuel density without at least considering whether the carbon content and heating value should be changed simultaneously.

In the following, a recent estimate of the density of fuels is reviewed, and compare with the estimates in DeLuchi (1993). However, in light of the foregoing cautions regarding changing one fuel parameter without changing the others, this comparison is used only to see if there might be need for further investigation into the properties of fuels.

Browning (1998b) used the API gravity of regular unleaded gasoline, as reported by the National Institute for Petroleum and Energy Research (NIPER), to calculate the density of winter and summer fuel from 1987 to 1986. The year-round average over the period was 2800 g/gal, slightly higher than the 2791 g/gal assumed here. However, the density increased slightly over the period from 1987 to 1991, the average was about 2785; from 1993 to 1996, it was about 2814 g/gal.

Browning (1998b) also used the API gravity reported by NIPER to calculate the density of #2 diesel fuel. From 1987 to 1996, the average density was 3220 g/gal, somewhat higher than the value of 3192 assumed here. However, the density has been declining slightly since 1992, and the average from 1994 to 1996 was about 3202 g/gal.

It appears that the density of gasoline and diesel fuel now might be slightly higher than assumed for the LEM. If the density were to increase, without also changing the carbon content or heating value, the higher density would result in slightly higher fuel-cycle GHG emissions (as much as 1.0% higher). Hence, the estimates from DeLuchi (1993), but note that there is some possibility that in recent years there have been nontrivial changes in the properties of gasoline and diesel fuel.

Fischer-Tropsch (F-T) diesel from natural gas

Diesel fuel made from natural gas, via the F-T process, has been added to the model (see Knott [1997] for a review of F-T diesel projects).The F-T diesel made by Sasol (n.d.) has a density of 780 g/l, and a sulfur content of less than 1 ppm. Stork (1997) reports a density of 770 g/l, a carbon content of 84.82%, and a higher heating value (HHV) of 131,000 BTU/gallon. I use the Stork (1997) data.

Biodiesel derived from soybeans

Soy diesel has been added to the model. The characteristics of 100% soy diesel fuel are based on data from EPA (2002a) and other sources shown in Appendix A to this report. These sources indicate 128,200 BTU/gal (HHV), 883 g/l, and 77.8% carbon.

CO_2 from biomass-derived ETBE

A previous version of the model did not deduct from total CO_2 emissions any CO_2 emitted from the biomass-derived ethanol portion of ETBE additive. This has been corrected.

Mixtures of reformulated gasoline and conventional gasoline

The ability of the model has been expanded to calculate complete CO_2-equivalent fuelcycle emissions from mixtures of:

1) conventional gasoline and reformulated gasoline;
2) gasoline and methanol (from coal, natural gas, or wood); and
3) gasoline and ethanol (from corn or wood).

In the previous version of the model, one could specify either all reformulated gasoline or all conventional gasoline, but nothing in between. Now, the user can specify any volumetric mixture of reformulated and conventional gasoline. You input the characteristics of conventional gasoline, the characteristics of reformulated gasoline, vehicular g/mi emission factors for conventional gasoline, and g/mi emissions factors for reformulated gasoline. The model calculates the characteristic of the specified fuel mixture, the fuel economy of a vehicle using the mixture, and average g/mi emissions from a vehicle using the mixture. The average g/mi emissions are calculated simply as the input g/mi emissions for conventional gasoline multiplied by the fraction of miles driven on conventional gasoline, plus input g/mi emissions for reformulated gasoline multiplied by the fraction of miles driven on reformulated gasoline. The mileage fractions are calculated on the basis of the specified fuel mix and the thermal efficiency of each fuel.

Mixtures of alcohols and gasoline

The previous version of the model could not estimate emissions from a mixture of biomass-derived methanol and gasoline (from crude oil). Now the model can estimate emissions from any mixture of biomass-derived alcohol (methanol or ethanol) and gasoline. The model calculates g/mi emissions from vehicles using these mixtures in the same way that it calculates g/mi emissions from vehicles using mixtures of conventional gasoline and reformulated gasoline (explained above).

Mixtures of soy diesel and petroleum diesel

The model estimates complete fuel cycle emissions from any mixture of soydiesel and petroleum diesel. The user specifies the volume percentage of soy diesel in the fuel, and the model calculates the energy characteristics of the fuel mix, the fuel consumption of the vehicle, the emissions of the vehicle, and the upstream emissions associated with fuel production.

LPG intermediate results

In the intermediate calculation of grams-CO_2 equivalent fuelcycle emissions per million-BTU (Table 7 of DeLuchi [1991]), the LPG column has been separated into LPG from natural gas, and LPG from oil.

Source of LPG

In recent years the fraction of propane and butane being supplied from refineries rather than natural-gas-liquids plants has been increasing. For 1995, 43% of the LPG is estimated to be supplied to the market from refineries (EIA, *Petroleum Supply Annual [PSA] 1995*, 1996) (compare with assumptions in Appendix G of DeLuchi [1993]). This change in the source of LPG causes an increase in fuel cycle GHG emissions of less than 1%.

The 43% figure is proportional to the share of propane from refineries (42%) weighted by the fraction of propane in fuel LPG, plus the share of butane from refineries (62%) weighted by the fraction of butane in fuel LPG. The refinery-source share of butane or propane product supplied to the market is equal to refinery production divided by the quantity [field production+imports+refinery production-refinery inputs] (data from Tables 2, 15, 16, 17, and 20 of the *PSA*).

Heating value, carbon content, sulfur content, and ash content of coal

In Table C.1 of DeLuchi (1993), the carbon content of coal is 57.17 lbs-C / 10^6-BTU for generic coal and coal for methanol plants, and 57.35 lbs-C / 10^6-BTU for coal for power plants(from data presented in Table C.6 of DeLuchi (1993)). The C / BTU value was assumed to be independent of the rank of the coal. Recently, the EIA (Hong and Slatick, 1994) analyzed 5,426 coal samples, and concluded that in 1992 all U.S. coal averaged 56.65 lbs-C / 10^6-BTU, and coal for power averaged 56.68 lbs-C / 10^6-BTU. They also demonstrated that C / BTU content in fact varies slightly with the rank of the coal. Generally, as the rank decreases, from bituminous to sub-bituminous to lignite, the C / BTU content increases slightly. This, coupled with the shift in consumption from high-sulfur Eastern bituminous coal to low-sulfur Western sub-bituminous coal, has resulted in a steady increase in the average C / BTU content of coal consumed in the U. S. (Hong and Slatick, 1994; EIA, *Annual Energy Review [AER] 1995*, 1996). The limitations on sulfur emissions specified by the Clean Air Act Amendments of 1990 will continue the shift from high-sulfur Eastern bituminous coal to low-sulfur Western sub-bituminous coal (Hong and Slatick, 1994; EIA, *Annual Energy Outllook [AEO] 1996*, 1996). The EIA's *AEO 2001* (2001; supplemental table 89) projects that the production of high-sulfur coal will decline by 0.5% / year, and that the production of low-sulfur coal will increase by 2.0% / yr, from 1999 to 2020.

In light of these and other new data, the heating value, carbon content, sulfur content, and ash content of coal is projected to decrease steadily in the coming years. The new base-year values, and the projected rates of change, are shown and documented in Table 4. These changes in the specifications of coal increase coal-cycle emissions by about 0.5%.

Sulfur content, carbon content, and heating value of biomass

Switch grass has been added to the analysis as feedstock for the production of fuel ethanol. Perlack et al. (1992) assume that a mix of wheat grass and switch grass has a HHV of $15.00 \cdot 10^6$ BTU / ton,[6] and that switch grass contains 48.4% carbon

Mann and Spath (1997) report that hybrid poplar contains 0.09% sulfur and 50.88% carbon by dry weight. Perlack et al. (1992) assume 54.3% C for hybrid poplar. Lamlon and Savidge (2003) measured the carbon content of wood from 41 species of North American trees, and found that hardwoods had an average of 48.4% carbon, and softwoods 51.05%. I assume that hybrid poplar contains 52% C and 0.09% S, and that switchgrass contains 48.4% C and 0.09% S.

Agriculture and Agri-food Canada (1997) report that corn is 46% C (dry weight). The EIA (*Emissions of Greenhouse Gases in the United States 1997,* 1998) cites estimates of 47.1% C for corn, and 44.0% C for soybeans. Liang and MacKenzie [1992] measured 50% C in corn stover. I use the EIA estimates. In the absence of data I assume that corn and soybeans (and their residue) contain 0.05% S by weight.

Carbon content, specific gravity, and sulfur content of crude oil

On the basis of ultimate analyses of 1982 crude oil samples, the EIA (*Emissions of Greenhouse Gases in the United States 1987-1994,* 1995) has estimated the carbon content of crude oil as a function of the sulfur content and API specific gravity:

$$CF_{oil} = 0.7699 + 0.1019 \cdot SG - 0.76 \cdot SF_{oil}$$

$$SG = \frac{141.5}{API + 131.5} \qquad \text{eq. 27, 28}$$

where:

CF_{oil} = the calculated carbon weight fraction of crude oil.
SG = the specific gravity (g / ml).
API = the density of the oil in degrees API.
SF_{oil} = the sulfur fraction of the oil.

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[6] Tyson et al. (1992) , who wrote the summary of the analysis to which Perlack et al. (1992) contributed, stated that "throughout the energy analysis, lower heating values are assumed for all the fuels except for biomass" (p. 75).

The EIA (*AEO 1996*, 1996) projects that crude oil will become denser and more sulfurous as the lighter, higher-quality stocks are exhausted. On the basis of the projections in the *AEO 1996* (EIA, 1996), the API parameter decreases by 0.5%/year, and the parameter $S_{fraction}$ increases 1%/year, with respect to the values in 1994. With these assumptions, the model uses the equation above to calculate the carbon fraction ($C_{fraction}$) for any year desired. The equation projects slightly lower carbon contents than assumed in Table C.1 of DeLuchi (1993): about 0.850 versus 0.855. The model also now calculates the g/gal density of crude oil given the input API density value:

$$\text{density in g/gal} = SG \cdot 1000 \cdot 3.7854$$

where:

SG = the specific gravity calculated from the input API parameter value, per Eq. 28 .

The resulting g/gal densities generally are higher than assumed in Table C.1 of DeLuchi (1993): 3250 versus 3191.

Composition of refinery gas

The EIA (*Emissions of Greenhouse Gases in the United States 1987-1994,* 1995) reported and discussed four estimates of the composition of refinery gas. Three of their four estimates are from source "E" of Table C.5 of DeLuchi (1993). The fourth estimates a composition of 12.7% H_2, 28.1% CH_4, 17.1% C_2H_6, and 11.9% C_3H_8. The EIA (1995) concludes that refinery gas generally must comprise mainly "less valuable" feedstocks, such as CH_4 and CO. This conclusion, and the new (fourth) estimate cited above, are consistent with the original assumptions of Table C.3 of DeLuchi (1993). However, Kadam et al. (1999) believe that still gas is about 75% methane.

Now that the model estimates fuel cycle emissions of SO_2, it is important to know the sulfur content of refinery gas, the main fuel used at refineries[7]. Unfortunately,

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[7] SO_2 emissions from refinery-gas boilers, in grams per 10^6 BTU of refinery gas, are equal to:

$$SO2_{RG} = S_{RG} \cdot GBTU_{RG} \cdot 2 \cdot F_{RG}$$

where:

$SO2_{RG}$ = emission of SO_2 from the combustion of refinery gas (g-SO_2/10^6 BTU-refinery gas).

S_{RG} = sulfur content of refinery gas (mass fraction).

$GBTU_{RG}$ = higher heating value of refinery gas (g/10^6 BTU-refinery gas) the sulfur content of refinery gas may vary considerably from refinery to refinery, and there are no estimates of the national-average sulfur content, or of national-average SO_2 emissions from refinery gas boilers[8].

Several sources mentioned in DeLuchi (1993) indicate that refinery gas, at least as it is produced, contains 1-2% hydrogen sulfide (H_2S) by volume. If refinery gas containing 1-2% H_2S is burned in uncontrolled boilers, SO_2 emissions from refineries will be quite large. It is likely, however, that in places, state or local regulations limit the sulfur content of refinery gas. For example, in the South Coast Air Basin of California, the sulfur content of refinery gas cannot exceed 40 ppmv (0.004%) H_2S (rule 431.1 (c) (3); Fakhoury, 1997; see also http://arbis.arb.ca.gov/drdb/sc/curhtml/r431-1.pdf). In Northern California, the sulfur content probably is similar.

In other states, the regulations on sulfur content may not be as strict as they are in California. It appears that before the most recent tightening of the regulations in Southern California (in 1994 and 1996), refinery gas contained on the order of 100 to 200 ppmv (my calculations, based on data provided by Fakhoury, 1997). For the LEM, we assume a national-average H_2S content of 0.0150% in 1994, a minimum of 0.0034%, and a steepness parameter (exponent K in Eq. 4) of 0.04.

Table 5 shows the composition assumed in this report. The composition results in 15.4 kg-C/10^6-BTU-gas, substantially lower than the 17.5 figure that the EPA (1998c) states it gets from the EIA.

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2 = the ratio of the molecular mass of SO_2 to S.

F_{RG} = factor to account for the use of any stack emission controls (the ratio of controlled to uncontrolled stack emissions).

Because boilers that use "waste" fuels such as refinery gas are not subject to national New Source Performance Standards (DeLuchi et al., 1992), and any state and local emissions regulations probably govern the sulfur content of the gas rather than the SO_2 level of the emissions (e.g., in the South Coast Air Quality Management District), there probably are no stack controls on SO_2 emissions from boilers that burn refinery gas, and consequently the factor F is likely to be 1.0. Hence, SO_2 emissions from refinery-gas boilers probably are determined entirely by the sulfur content of the fuel.

[8] E. H. Pechan Associates, the contractor who prepares the national emission inventory for the EPA (*National Air Pollutant Emission Trends*, annual report), does not do an original calculation of SO_2 emissions from refinery boilers, but rather bases its estimates on estimates made by the states (Barnard, 1997). The state estimates, in turn, are based on the EPA's *AIRS Facility Subsystem Source Classification Codes and Emission Factor Listing for Criteria Air Pollutants* (1990). This document states that SO_2 emissions from petroleum refinery gas used in industrial boilers (source classification code1-02-007-01, p. 23) should be estimated on the basis of the sulfur content of the fuel, but does not specify the sulfur content.

Carbon content of petroleum coke

In DeLuchi (1993), petroleum coke was assumed to be 90% C by weight. The EIA (*Emissions of Greenhouse Gases in the United States 1997*, 1998) does state that coke is "about 90% carbon by weight" (p. 81), but its actual carbon emission factor (used by the EPA [1998c]) indicates that coke is about 92% carbon by weight. This assumption has now been changed to 92%.

Composition of natural gas (CNG and LNG)

The volumetric composition of pipeline natural gas has been changed slightly to the "industry average" shown by the Auto/Oil Study (1996) and the "typical" composition reported by the EIA (*Alternatives to Traditional Transportation Fuels,* 1994). The composition is shown in Table 5. The category "butanes plus" in DeLuchi (1993) has been broken out into the categories "butane" and "pentanes plus".

LNG. Powars et al. (1994) note that some members of the "LNG vehicle community" have argued that in order to maximize vehicle performance, LNG should be nearly pure methane -- i.e., that most of the higher alkanes should be removed from natural gas. Although it is true that pure methane has some advantages over pipeline NG, it is costly to remove the higher alkanes from NG, and recent improvements in closed-loop electronic fuel injection systems for NGVs have greatly reduced the advantages of pure methane. For example, as discussed elsewhere, emissions from state-of-the-art NGVs are not very sensitive to fuel composition. (Almost all of the arguments in favor of pure methane apply to CNG as well.) In general, it appears now that vehicle technology will adequately compensate for typical variations in gas quality. In this analysis, LNG is assumed to be liquefied pipeline natural gas.

Sulfur in NG. Previously, the EPA estimate of 7 ppm sulfur in pipeline NG was used. However, the EIA's *Natural Gas 1998, Issues and Trends* (1999) reports two studies in which natural gas contains less than 5 ppm of all sulfur compounds, and notes further that contracts usually limit sulfur content to 1.9 to 7.6 ppm, "in many cases 1.9 ppm" (p. 53). Given this, I assume that natural gas contains 4 ppm sulfur by weight.

Density and energy content of gases

In the model, the volumetric density and energy content of fuel gases are calculated on the basis of the molar heating value, molecular mass, and molar fraction of the individual compounds in the fuel gases, using Van der Waal's modification of the ideal gas law.

The molar density of an ideal gas (moles/liter) can be calculated using the ideal gas law:

$$C = \frac{n}{V} = \frac{P}{R \cdot T} \qquad \text{eq. 29}$$

where:

C = the molar concentration (moles / liter).
n = the number of moles.
V = the volume occupied by the gas, in liters (L).
P = the pressure (atm).
R = the gas constant (0.082057 L atm K^{-1} mol^{-1}).
T = the temperature in degrees Kelvin (K).

However, the ideal gas law does not account for the volume occupied by the molecules themselves, or for attractive or repulsive forces between gas molecules. For example, if gas molecules are mutually attractive, then the actual density of the gas will be greater than predicted by the ideal gas law. Conversely, the assumption that an ideal gas occupies no volume over predicts the density of a real gas, which occupies some non-zero volume. At high temperatures and pressures, these factors can cause a real gas to behave significantly differently from an ideal gas.

Van der Waal's equation of real gases accounts for the forces between gas molecules, and the space occupied by gas molecules:

$$n \cdot R \cdot T = \left(P + \frac{n^2 \cdot a}{V^2} \right) \cdot (V - n \cdot b) \qquad \text{eq. 30}$$

where:

a = gas-specific constant that accounts for forces between gas molecules (values given in the *CRC Handbook of Chemistry and Physics*, 1984).
b= gas-specific constant that accounts for the volume occupied by gas molecules[9] (values given in the *CRC Handbook of Chemistry and Physics*, 1984).

Van der Waal's equation can be expressed in terms of the molar concentration *D:*

$$D = \frac{(P + D^2 \cdot a) \cdot (1 - D \cdot b)}{R \cdot T}$$

or

$$D^3 + \left(-\frac{1}{b} \right) \cdot D^2 + \left(\frac{P \cdot b + R \cdot T}{a \cdot b} \right) \cdot D + \left(-\frac{P}{a \cdot b} \right) = 0 \qquad \text{eq. 31}$$

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[9] The constants *a* and *b* actually vary with temperature (*RC Handbook of Chemistry and Physics*, 1984).

The first expression can be solved by iterations; the second, the cubic polynomial, by a rather cumbersome series of expressions that involve complex roots. In the model, the first expression is solved by iterations. I assume the pressure (1 atm) and the temperature (60^{o} F, which is 288^{o} K) at which volumes of natural gas are reported to the EIA (as explained in the *AER*). The resulting molar concentration of the molecular-constituents of fuel gases is shown in Table 5[10].

Given these molar concentrations for the individual gas components at 288^{o}K and 1 atm, and assuming that the molar concentration of a combination of these gases is just the sum of the molar concentrations of the components weighted by their volume (molar) shares[11], it is straightforward to calculate the mass density and the volumetric higher heating value:

$$VHHV_g = \sum_i MHHV_i \cdot VF_{i,g} \cdot D_i \qquad \text{eq. 32}$$

$$Dm_g = \sum_i MW_i \cdot VF_{i,g} \cdot D_i \qquad \text{eq. 33}$$

subscript g= the gases analyzed in the model (raw natural gas, pipeline natural gas, coalbed gas, refinery gas, LPG, or hydrogen made from natural gas).
subscript i = the molecular compound constituents of the gases (Table 5).
$VHHV_g$ = the volumetric higher heating value of gas g (kJ/liter).
$MHHV_i$ = the molar higher heating value of compound i (kJ/mole) (Table 5).
$VF_{i,g}$ = the volume (molar) fraction of molecular compound i in gas g (Table 5).
D_i = the molar concentration of compound i (explained above).
Dm_g = the mass density of gas g (g/liter).
MW_i = the molecular mass of compound i (Table 5).

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[10] At 1 atm and 288^{o}K, most real gases behave rather like ideal gases, and render the Van der Waal's modification unnecessary. However, for the heavier alkanes, the density calculated using Van der Waal' equation and shown in Table 5 deviates from the "ideal-gas" density by 2-3%, which is beginning to significant. Furthermore, at temperatures and pressures significantly above the standard, the deviation becomes more significant.

[11] This weighting procedure does not account for attractive or repulsive forces between different compounds (the constant *a* in Van der Waal's equation accounting only for forces between molecules of a particular compound). However, to the extent that fuel gases comprise mainly one compound, this inter-compound effect probably is not significant.

Hydrogen from natural gas and from water electrolysis

I assume that hydrogen fuel made from natural gas contains trace amounts of CH_4, CO, CO_2, and N_2 (Table 5). The CO_2-equivalent effect of combustion or evaporative emissions of these trace compounds is duly counted in the model.

I assume that hydrogen fuel made from electrolysis of water is 100% H_2 and that hydrogen shipped in pipelines, and liquefied hydrogen, is 100% H_2.

In the model, the user can specify any mix of natural gas or water feedstock for hydrogen. The model weights the end-use compositions (as shown in Table 5 for hydrogen from NG; 100% H_2 for hydrogen from electrolysis) and calculates emissions accordingly.

In the case of water electrolysis, the user can specify any mix of electricity, including electricity derived from fossil fuels. However, one should keep in mind that it is cheaper and more efficient to reform fossil fuels (such as natural gas) directly into hydrogen than to use them to generate power to split water. Also, in the electrolytic hydrogen fuelcycle, the “feedstock recovery” and “feedstock transmission” stages now pertain to the water feedstock, not to the feedstock used to generate the electricity. All of the emissions related to the electricity cycle are included in the “fuel production” stage.

MOTOR VEHICLES: ENERGY USE, FUEL STORAGE, WEIGHT, AND MATERIALS

Fuel economy, drive cycle, and vehicle weight

In the previous version of the model, one entered the following:

- the fuel economy of baseline gasoline vehicle.
- the fuel economy of the baseline diesel vehicle.
- the thermal efficiency of the AF ICEVs relative to that of the baseline gasoline ICEV.
- the thermal efficiency of the AF ICEVs relative to that of the baseline diesel ICEV.
- the efficiency of the EV power train relative to the efficiency of the ICEV power train.
- weight parameters.
- the effect of weight on fuel economy.

Given the input data, and an equation that calculated the weight of the baseline vehicle on the basis of a statistical relationship between weight and EPA city/highway mpg, the previous version of the model calculated the weight and energy use of the all of the vehicles. Note that in order for the weight/fuel-economy equation to have given the correct result, the input mpg had to have been the combined city/highway mpg.

For example, to compare EVs with ICEVs in city driving -- one had to enter the city mpg of the baseline gasoline vehicle, but then overwrite the weight-calculation equation with the weight calculated from the city/highway mpg. This work-around was cumbersome.

The LEM has been rewritten to correctly calculate weight and fuel-use for all vehicles for any user-specified mix of city and highway driving. The user now supplies the following input data for the baseline gasoline and diesel vehicles:

- the fuel economy of baseline vehicle using conventional gasoline, in city driving.
- the fuel economy of baseline vehicle using conventional gasoline, in highway driving.
- the fuel economy of the baseline diesel vehicle, in city driving.
- the fuel economy of baseline diesel vehicle, in highway driving.
- the city fraction of total miles driven by light-duty ICEVs.
- the city fraction of total miles driven by heavy-duty ICEVs.
- the weight of the baseline diesel vehicle.

With these inputs, the model calculates the fuel economy of the baseline gasoline and diesel vehicles over the specified driving cycles. The weight of the baseline gasoline vehicle is calculated on the basis of the original statistical relationship between weight and the 45/55 fuel economy. (The 45/55 fuel economy is calculated from the input data above.) The base-case parameter values used in this analysis are shown in Table 6.

To calculate the energy use of the alternative-fuel vehicles (AFVs) , the model first calculates the drivetrain efficiency and the weight the AFVs *relative* to the drivetrain efficiency and weight of the baseline gasoline or diesel vehicle. The relative drivetrain efficiency, expressed as the mi/BTU efficiency of the alternative-fuel engine or EV drivetrain divided by the mi/BTU efficiency of the baseline gasoline or diesel engine, is projected with Eq. 3. Table 6 shows the input values of V_U, V_{T_B}, and k. (These relative efficiency parameters are discussed in the next section.) Note that in this application of Eq. 3, the parameter T is the vehicle model year in the selected target year, V_T is the relative efficiency of the model year in the selected target year, T_B is a base model year, and V_{T_B} is the relative efficiency of the base model year.

In the case of EVs, the relative drivetrain efficiency is calculated as the ratio of the efficiency of the EV powertrain to the efficiency of the LDGV powertrain. The powertrain efficiencies for the EV and the LDGV are shown in a separate part of Table 6.

The relative weight, expressed as the difference between the weight of the AFV and the weight of the baseline gasoline or diesel vehicle. The relative weight is calculated as the change in the weight of the powertrain and body plus the change in the weight of the fuel storage system. The change in the weight of the fuel storage system, in turn, is calculated from input data on the range of the vehicle and the characteristics of the fuel storage system (discussed more below). The total change in weight is multiplied by a weight compounding factor (shown in Table 6) for the extra structure associated with any extra weight.

Finally, given the calculated or input fuel economy and weight of the baseline gasoline and diesel vehicles, the calculated relative drivetrain efficiency and vehicle weight, and the user-specified relationship between changes in weight and changes in energy use, the model calculates the actual energy use of the AFVs over the specified drive cycle (see below). The model assumes that alternative-fuel ICEVs follow the same drive cycle as baseline gasoline or diesel vehicles. However, the user now can specify a separate drive-cycle for EVs.

Efficiency of AF ICEVs relative to that of baseline gasoline or diesel vehicles

As mentioned above, the model calculates the energy use of all vehicles other than the baseline gasoline and diesel vehicle. The calculation is based on the drivetrain efficiency and the weight the AFVs *relative* to the drivetrain efficiency and weight of the baseline gasoline or diesel vehicle. Of these two parameters, the relative drivetrain efficiency is the most important, and is discussed briefly next.

The parameters for AF ICEVs are shown in Table 6. These parameters are estimated on the basis of the data cited in Appendix B of DeLuchi (1993) and in studies published since, including: Van Blarigan (1998), NREL (1996), Milkins and Edsell (1996), and, for soydiesel, Appendix A to this report.

NREL (1996) reports the following mi/BTU efficiency ratios for alternative-fuel transit buses (relative to diesel) tested as part of an extensive evaluation program:

Houston	Portland	Miami	Tacoma	Peoria	Peoria	St. Paul	Miami	NY	St. Louis
LNG	LNG	CNG	CNG	E95	E93	E95	M100	M100	BD-20
0.87	0.70	0.97	0.77	1.02	0.96	0.94	0.99	0.87	1.01

Norton et al. (1996) and NREL (1997) report similar values -- about 0.91 to 1.0 -- for four HD trucks operated on E95. Milkins and Edsell (1996) report ratios of 0.79 to 0.86 for Cummins L10 CNG buses operated over three different drive cycles. According to Cummins technical specification sheets, the Cummins 5.9L LPG engine has about the same fuel consumption (g/bhp-hr) as the Cummins 5.9L NG engine. NREL (2002) reports a ratio of 0.73 for medium duty CNG trucks (with a catalytic converter) relative to diesel controls, but states that with newer CNG technology, the ratio is 0.85 to 0.90.

Note that the efficiency ratios from NREL (1996) and Milken and Edsell (1996) include the effect of the extra weight of alternative-fuel storage systems. If the effect of weight is removed, so that the efficiency ratio reflects only differences in thermal efficiency (and perhaps drive cycle), then the ratios for methanol, ethanol, and LNG will increase by 1 to 2%, and the ratios for CNG by about 5%. Note too that the low values reported by NREL (2002) are due part to the CNG vehicles having to forego the efficiency advantages of lean operation on account of the catalytic converter.

Schaedel et al. (1996) state that a turbo-charged lean-burn NG engine can approach the efficiency of HDDVs at full load. They note that lean-burn NG engines are sensitive to small variations in the fuel composition, but suggest that this can be handled by advanced fuel and air sensors. Nimocks (1995) state that direct-injection NG technology can achieve the same efficiency as diesel engines. Given that advanced, closed-loop, electronically controlled lean-burn CNG HD engines have very low emissions, as well as high efficiency. It seems reasonable to assume that most heavy-duty CNG engines will be of this type, and hence will have a relatively small efficiency penalty compared to diesel.

Van Blarigan (1998) reports on the development of hydrogen engines that use ultra-lean burn and high compression ratios to achieve very high thermal efficiency and essentially zero NO_X emissions. (See also Lipman and Delucchi, 1996).

Efficiency of LD diesel vehicles versus LD gasoline vehicles

Diesel vehicles can have a much higher fuel economy than gasoline vehicles, in part because diesel fuel contains 11% more BTUs per gallon than gasoline, and because compression-ignition diesel engines are more thermally efficient than are spark-ignition gasoline engines. They are more thermally efficient mainly because they operate at a much higher compression ratio, and use a leaner air/fuel ratio.

The relative thermal efficiency of diesel engines depends on the type of engine technology. Direct-injection engines are more efficient than prechamber diesels; and turbocharged engines are more efficient than non-turbocharged engines.

Energy and Environmental Analysis (1991) compares the EPA composite city/highway fuel economy of 1987 diesel passenger cars with that of their gasoline counterparts, and finds an increase of 20-36% in mpg. However, they note that all of the diesels in this comparison are of the prechamber type, and that conversion to direct injection results in an additional 15% fuel economy benefit. They note further that with diesels the difference between the real world fuel economy and the EPA test-cycle fuel economy is less than with gasoline.

Estimates by Schipper (1999) are Consistent with those of EEA (1991): Schipper (1999) reports that the on-road fuel economy of diesel vehicles in Europe in 1995 was 19-36% higher than the on-road fuel economy of gasoline vehicles.

Redsell et al. (1988) state that light-duty diesel vehicles “offer a generally accepted fuel savings of 25%” (p. 1). They also cite a study in which a diesel vehicle had 28% lower fuel consumption than a comparable gasoline vehicle (p. 10). In their own work, they compared the fuel consumption of a 1600 cc diesel Vauxhall Cavalier with that of a 1300 cc gasoline Vaxhual Cavalier of a comparable performance. The test route was a mixture of urban, suburban, and freeway driving. The diesel vehicle had 22% lower fuel consumption in urban driving, 17% less in suburban driving, and 4% less in highway driving.

An unpublished presentation by M. Walsh compares the fuel economy of a gasoline VW Golf and a gasoline VW Passat with the fuel economy of the diesel-fueled, turbo-charged, direct-injection (TDI) versions, and finds the following changes (diesel TDI vs. gasoline):

	European City Cycle	90 km/hr	120 km/hr
VW Golf	+73%	+47%	+39%
VW Passat	+59%	+44%	+33%

The advantage of the diesel decreases as the average power required over the drive cycle increases, because the throttling losses of the gasoline engine decrease as the power demand increases and the throttle opens up.

These percentages refer to the increase in the mi/gallon fuel economy, whereas our model requires as an input in the change in the mi/BTU fuel economy. A gallon of diesel fuel has 11% more BTUs than a gallon of gasoline; hence, the data above imply that mi/BTU fuel economy of diesel vehicles is something on the order of 10-55% greater than that of gasoline vehicles, depending on the technology and drive cycle[12]. We will assume an increase of 35% in city driving, and 20% in highway driving.

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[12] There are data on aggregate fuel consumption by broad classes of diesel and gasoline vehicles, but the data are not disaggregated enough to permit comparison of "similar" gasoline and diesel vehicles. For example, the EIA's Residential Transportation Energy Consumption Survey (EIA, *Household Vehicles Energy Consumption 1994,* 1997) reports gallons of fuel, number of vehicles, and miles per vehicle for household diesel and gasoline sedans and pickups, in 1994. With these data, one can calculate the fuel economy for the different vehicle types:

	Sedans				Pickup trucks			
	10^9 gal	10^6 veh	10^3 mi/v	mpg	10^9 gal	10^6 veh	10^3 mi/v	mpg
Gasoline	50.4	98.6	11.2	21.9	18.4	27.3	11.0	16.3
Diesel	0.3	0.7	10.5	24.5	0.9	1.0	13.4	14.9

The diesel sedans have 12% greater fuel economy than the gasoline sedans, but the diesel pick ups actually have a slightly lower fuel economy than the gasoline pickups. However, it is likely that the diesel pickup trucks are considerably larger, on average, than the gasoline pickup trucks.

Data from the TIUS also are problematic. As discussed in the text, Browning (1998b) extracted data, from the 1992 TIUS, on actual fuel economy by model year and gross-vehicle-weight (GVW) category, and then estimated fuel economy as a power function of model year, for each GVW class. His function results in the ratios of diesel mpg to gasoline mpg, by weight class and model year:

min wt.	MY 1985	MY 1995
8,501	1.27	1.28
10,001	1.23	1.24
14,001	1.16	1.10
16,001	1.21	1.23
19,501	1.15	1.08
26,001	1.11	1.02
33,001	0.94	0.92

Electric vehicles

Relative drivetrain efficiency. As mentioned above, the relative drivetrain efficiency of the EV is calculated) as the ratio of the efficiency of the EV powertrain to the efficiency of the LDGV powertrain. The powertrain efficiencies for the EV and the LDGV, are shown in a separate part of Table 6. The efficiencies are estimated on the basis of a detailed second-by-second drive-cycle energy-use analysis of EVs and ICEVs. The drivecycle energy-use model is documented in Delucchi (2000a), and discussed briefly below. This model was developed in part because DeLuchi (1991) found that the relative drivetrain efficiency was the most important and uncertain variable in the EV analysis, and needed to be characterized much better. Table 7 shows the relative drivetrain efficiency estimated by the second-by-second energy-use model. The assumptions shown in Table 6 produce results consistent with the results of Table 7[13].

The efficiency ratio is equal to the mi/BTU efficiency of the EV drivetrain (measured at the battery terminals) divided by the mi/BTU efficiency of the ICEV system. The mi/BTU efficiencies are calculated by a detailed vehicle energy-use model, documented in Delucchi (2000a). This model calculates the energy consumption of the vehicles from the efficiency or energy consumption of individual components (the battery, the engine, the transmission, the motor controller, and the vehicle auxiliaries), the characteristics of the drive cycle, the characteristics of the vehicle, and the energy requirements for heating the battery. The efficiency of the battery, electric motor, and controller are calculated from plots of efficiency as a function of torque and rpm.

In the model, the drivecycle followed by the EVs and ICEVs consists of up to 160 linked segments, defined by the user. For each segment, the user specifies the vehicle speed at the beginning, the speed at the end, the wind speed, the grade of the road, and the duration in seconds. Given these data for each segment of the drivecycle, and calculated or user-input vehicle parameters (total weight, coefficient of drag, frontal area, coefficient of rolling resistance, engine thermal efficiency, and transmission efficiency), the model uses the physics equations of work and empirical approximations to calculate the actual energy use and power requirements of the vehicle for each segment of the drivecycle.

The model properly calculates the extra energy made available by regenerative braking. The model calculates the amount of energy applied to the brakes, then cycles that available energy back through the powertrain to the energy-storage device (e.g., a

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It is not immediately clear why the fuel economy advantage of diesel should decline with increasing vehicle weight.

[13] The results of Table 7 can be compared with the analysis in Table B.1 of DeLuchi (1993). In Table B.1, the relative efficiency in city driving ranged from 5 to 6, and was 5.7 on average. The values of Table 7 are quite a bit higher, and result in considerably lower greenhouse-gas emissions.

battery) and through the energy-storage device to its outgoing terminals. The model restricts regenerative power to be less than or equal to a user-specified maximum, and restricts regenerative energy to be less than or equal to the available capacity of the energy-storage device.

The model uses an empirical formula to calculate the amount of frictional work within an engine. Friction work is equal to kJ of friction work per liter of displacement per revolution of the engine, multiplied by the displacement in liters (an input variable) and the number of engine revolutions. The parameter [kJ of friction work per liter of displacement per revolution of the engine] is itself a function of the power output of the engine. The model calculates the exact number of engine revolutions over each segment, given a user-defined shift schedule, user-input gear ratios, and starting and ending speeds. The model properly accounts for any number of gear shifts within a segment, at any point within the segment.

Other EV parameters. Several other assumptions and calculation methods regarding EVs have been changed:

i) the lifetime of the vehicle has been reduced to be only 1.1 times longer than the ICEV life, rather than 1.42 times (as in Table P.2 of DeLuchi [1993]), on the basis of a reconsideration of the likely longevity of EVs;

ii) the lifetime of the battery now is calculated as:

$$L = CL \cdot MU \cdot DoD \qquad \text{eq. 34}$$

where:

L = the battery life in miles.
CL = the cycle life (see below).
MU = the urban driving range (see below).
DoD = the average depth of discharge per cycle (assume 75%).

iii) the specific energy of the battery, a key determinant of the weight and hence efficiency of the EV, has been added as an input variable. With this, the weight of the battery is calculated as:

$$WB = \frac{EC \cdot R \cdot 1000 \cdot 2.205}{DoD \cdot SE} \qquad \text{eq. 35}$$

where:

WB = the weight of the battery (lbs).
EC = the energy consumption of the EV, from the battery terminals (kWh / mi; calculated as a function of the drivetrain efficiency and the weight of the vehicle).

R = the driving range of the vehicle (projected to increase as the specific energy and performance of the EV improve [Table 8]).
1000 = Wh / kWh.
2.205 = lbs / kg.
DoD = the depth of discharge at the desired driving range (1.00).
SE = the specific energy of the battery (Wh / kg; projected with Eq. 6 [Table 8]).

iv) the cycle life of the battery, the efficiency of the battery, the efficiency of recharging, the relative weight of the EV powertrain, and the urban driving range of the EV now are projected for every *model year*, with Eq. 6. Table 8 shows the values of V_U, V_L, V_{TB}, and k in Eq. 6, for these parameters. The values assumed here based on a review of recent literature, summarized in Table 9. Note that the driving range is projected to increase as the specific energy and performance of the EV improve. Also, the values are projected for the vehicle *model year*, which is not necessarily the same as the target year of the analysis. The relationship between model year and target year is discussed elsewhere.

Definition of heavy-duty diesel vehicles

Heavy-duty diesel vehicles are now defined more precisely, as having an average vehicle weight (AVW) of more than 26,000 lbs. In making my definition, I considered the following facts:

- The Federal emission standards for "heavy-duty" diesel vehicles apply to all trucks of greater than 8,500 lbs gross vehicle weight (GVW) (EPA, *Emission Standards Reference Guide for Heavy-Duty and Non-Road Engines,* 1997).
- Similarly, in the EPA's MOBILE5A NO_X, VOC, and CO emission-factor model, and in the official emissions inventory, emissions are reported for two diesel truck weight classes (EPA, *Compilation of Air Pollutant Emission Factors, Vol. II: Mobile Sources,* 1991; EPA, *National Air Pollutant Emission Trends,1900-1996,* 1997): "light," which is 6,001 - 8,500 lbs GVW, and "heavy," more than 8,500 lbs GVW.
- However, in the EPA's PART5 PM and SO_X emission-factor model, emissions are reported for the following diesel truck weight classes (EPA, *Draft User's Guide to PART5: A Program for Calculating Particulate Emissions from Motor Vehicles*, 1995):

  light: 6,001 - 8,500 lbs GVW
  light class 2: 8,501 - 10,000 lbs GVW
  light heavy: 10,001 - 19,500 lbs GVW
  medium heavy: 19,501 - 33,000 lbs GVW
  heavy-heavy: 33,000+ lbs GVW

- The 1992 *TIUS* (Bureau of the Census, 1995) shows billion truck miles of travel in four average-weight classes:

light: less or equal to 10,000 lbs AVW (681.3 billion VMT)
medium: 10,001 - 19,500 lbs AVW (14.0 billion VMT)
light heavy: 19,501 - 26,001 lbs AVW (8.1 billion VMT)
heavy-heavy: 26,000+ lbs AVW (82.8 billion VMT)

Thus, the bulk of trucks that are not "light" are in fact heavy-heavy trucks of more than 26,000 lbs. These trucks account for most of the fuel use and emissions by non-light trucks. (Note too that an AVW of 26,000 lbs might be close to a GVW of 33,000 lbs.)

Fuel economy and brake-specific fuel consumption of heavy vehicles

The emissions model requires several energy-use parameters for heavy-duty vehicles (buses or trucks):

• miles / gallon fuel economy, used to calculate CO_2 emissions from fuel use, and upstream fuel requirements per mile;

• BTU-work / BTU-fuel thermal efficiency, which depends on the type of engine and fuel, and is used in the calculation of the bhp-hr / mi;

• bhp-hr-work / mile; this is used to convert input g / bhp-hr emission factors to grams per mile.

Fuel economy. In the model, the fuel economy (mpg) of the HDV is an input parameter. This input value should be appropriate for the assumed average weight and calculated model year of the vehicle. As a guide, the model provides the EPA / MOBILE6 estimate of the average in-use economy for the given weight and model year, according to projections developed for EPA by Browning (1998b). Browning (1998b) extracted data, from the 1992 TIUS, on actual fuel economy by model year and gross-vehicle-weight (GVW) category, and then estimated fuel economy as a power function of model year, for each GVW class:

$$MPG_{w,MY} = C_w \cdot (MY - 1900)^{D_w} \qquad \text{eq. 36}$$

where:

$MPG_{W,MY}$ = the fuel economy of vehicle of GVW class w and model year MY (miles / gallon).
C_W = coefficient for GVW class w (shown below).
MY = the vehicle model year (estimated elsewhere).
D_W = exponent for GVW class w (shown below).

GVW		*Gasoline parameters*		*Diesel parameters*	
class	min wt.	C_W	D_W	C_W	D_W
2B	8,501	0.1253	0.9624	0.1072	1.0506
3	10,001	0.1157	0.9632	0.0989	1.0450
4	14,001	0.0409	1.1902	0.5020	0.6598
5	16,001	0.4416	0.6348	0.2474	0.8078
6	19,501	0.0338	1.2015	0.5336	0.6117
7	26,001	0.1277	0.8909	4.0206	0.1374
8A	33,001	0.0647	1.0285	0.1548	0.8194
8B	60,001	n.a.	n.a.	0.0119	1.3742

The Browning (1998b)/MOBILE6 power function and parameter values are used to estimate the mpg of HD trucks by weight and model year. This equation gives reasonable results back to MY 1970, and up to the MY 2010. If MY < 1970, MY-1970 is used, and if MY > 2010, MY = 2010 is used. Note, again, that this calculated mpg is presented as a guide for the user in his or her choice of input fuel economy for trucks.

Energy conversion efficiency. The energy conversion efficiency, in BTUs of brake-work for every BTU of fuel consumed (HHV), can be calculated from a recent EPA analysis of brake-specific fuel consumption (BSFC) of heavy-duty vehicles. As part of his update for EPA of the factors to be used in MOBILE6 to convert g/bhp-hr emissions to g/mi emissions, Browning (1998a) obtained data from six engine manufacturers on the BSFC of heavy duty engines from model year 1987 to 1996. With these BSFC data, and data on engine sales by weight class, and other data and assumptions, he estimated the sales-weighted BSFC by weight class and model year for heavy-duty engines. Finally, in order to be able to estimate the BSFC of model years outside the range for which he received data, he used regression analysis to estimate the BSFC as a logarithmic function of the model year, in each class:

$$BSFC_{w,MY} = A_w \cdot \ln(MY - 1900) + B_w \qquad \text{eq. 37}$$

where:

subscript w = the heavy-duty engine GVW classes (see below).
subscript MY = the heavy-duty vehicle model year.
$BSFC_{W,MY}$ = the brake-specific fuel consumption in GVW class w in model year MY (lb/bhp-hr).
A_W = coefficient for GVW class w (shown below).
MY = the engine model year.
B_W = constant for GVW class w (shown below).

GVW		*Gasoline parameters*		*Diesel parameters*	
class	min wt.	A_W	B_W	A_W	B_W
2B	8,501	-0.7211	3.8473	-0.4806	2.6959
3	10,001	-0.5656	3.1535	-0.5183	2.8529
4	14,001	-0.5583	3.1319	-0.1780	1.2897
5	16,001	-0.5435	3.0630	-0.0349	0.6162
6	19,501	-0.7339	3.9284	-0.1706	1.1985
7	26,001	-0.8224	4.3266	-0.0863	0.7854
8A	33,001	-0.7681	4.0725	-0.1141	0.9107
8B	60,001	n.a.	n.a.	-0.2003	1.2858
Transit	bus	-0.8652	4.4842	-0.5058	2.7092
Intercity	bus	-0.4951	2.8221	-0.3648	2.0764
School	bus	-0.4648	2.6918	-0.5311	2.8123

The Browning (1998a)/MOBILE6 logarithmic equation and parameter values are used to estimate the BSFC of HD engines by weight and model year. This equation gives reasonable results back to MY 1970, and up to the MY 2005 and, if values years are outside the range, the end-values are used.

BSFC (lb/bhp-hr-work) is converted to thermal efficiency (bhp-work/bhp-fuel, HHV) as follows:

$$EFF_{W,MY} = \frac{3412 \cdot 0.745712}{BSFC_{W,MY} \cdot FD} \qquad \text{eq. 38}$$

where:

3412 = BTU/kWH.
0.745712 = kWh/bhp-hr.
other terms as defined above.

<u>Bhp-hr/mi.</u> The work per mile can be calculated from the fuel economy, the brake-specific fuel consumption, and the fuel density:

$$BHPMI_{W,MY} = \frac{\text{FD}}{BSFC_{W,MY} \cdot MPG_{W,MY}} \qquad \text{eq. 39}$$

where:

$\text{BHPMI}_{\text{W,MY}}$ = the energy use of GVW class w in model year MY (bhp-hr/mi).

FD = the density of diesel or gasoline fuel (BTU / lb; discussed elsewhere in this report).

$BSFC_{W,MY}$ = the brake-specific fuel consumption in GVW class w and model year MY (lb / bhp-hr; Eq. 37).

$MPG_{W,MY}$ = the fuel economy of vehicle of GVW class w and model year MY (miles / gallon; input by the user based on Eq. 36).

We emphasize that these formulae are based on the vehicle model year, which in general will not be the same as what we call the "target year" of the analysis.

Formula to calculate energy efficiency of AFVs

In Appendix A and Appendix B of DeLuchi (1993), the following equation was used to calculate energy efficiency:

$$\frac{1}{M_i} = \frac{\left(1 + EFF_i\right) \cdot \dfrac{MPG_p}{D_p}}{1 + Wf \cdot \dfrac{W_i}{W_p}}$$

where:

M_i = 10^6-BTU / mi efficiency of AFV i.

$1+EFF_i$ = the powertrain efficiency of AFV i relative to that of baseline petroleum vehicle p $\left(\frac{mi/\ BTU_{powertrain-i}}{mi/\ BTU_{powertrain-p}}\right)$.

Wf = % decrease in fuel economy (in mi / BTU) per 1% increase in vehicle weight (Table 6).

W_i = the extra weight of AFV i compared to petroleum-fuel vehicle p.

W_p = the total driving weight of petroleum-fuel vehicle.

MPG_p = the miles-per-gallon fuel economy of petroleum-fuel vehicle p.

D_p = the 10^6-BTU / gallon heating value of petroleum fuel p.

This equation is wrong. The correct equation is:

$$\frac{1}{M_i} = \left(1 + EFF_i\right) \cdot \frac{MPG_p}{D_p} \cdot \left(1 - Wf \cdot \frac{W_i}{W_p}\right) \qquad \text{eq. 40}$$

This has been corrected in the model.

Range and fuel storage of heavy-duty vehicles

As mentioned above, the driving range of a vehicle, combined with the lb-storage / lb-fuel characteristic of the fuel-storage system, determines the weight of the fuel storage system. The weight of fuel storage in turn affects the efficiency and hence greenhouse-gas emissions of the vehicle. The weight of the fuel-storage system also directly determines greenhouse-gas emissions from the manufacture of materials for the storage system.

In Table 2 of DeLuchi (1991), alternative-fuel HDVs were assumed to have a shorter driving range than the baseline diesel-fuel HDV, and methanol and ethanol HDVs were assumed to weigh the same as their diesel counterparts. In the present model, the driving range of all of the alternative-fuel HDVs has been increased to make it closer to that of the diesel baseline, on the assumption that most operators of HDVs want to minimize "down time" spent refueling. The lb-storage-system / lb-fuel-weight characteristic of some of the storage systems has also been increased. (probably underestimated in the previous model) Together, these two changes increase the weight of fuel-storage systems on alternative-fuel HDVs, and hence reduce efficiency and increase GHG emissions. The new assumptions are shown in Table 10. Note that the extra weight of the methanol and ethanol HDV now is calculated, and not just assumed to be zero. The resulting calculated weights are consistent with those reported for transit buses by the National Renewable Energy Laboratory (NREL, 1996).

Soy diesel vehicles: range, fuel storage, and energy use

Soydiesel has been added as a fuel for heavy-duty vehicles. The tanks and engines for soydiesel are the same as those for diesel fuel (see assumptions in Table 10). However, on the basis of a few studies discussed in Appendix A to this report, soydiesel is assumed to be less efficient than diesel fuel, in the base year of 1995. Table 6 shows assumptions regarding the relative efficiency of soy diesel.

F-T diesel vehicles: range, fuel storage, and energy use

F-T diesel is similar enough to conventional low-sulfur petroleum diesel that it is reasonable to assume that range, fuel storage, and energy use are the same.

Vehicle weight

1) A minor mistake in the calculation of vehicular curb weight versus loaded weight has been corrected (Table 2 of DeLuchi [1991]).
2) The curb weight is still calculated on the basis of a relationship between combined city / highway mpg and vehicle weight, but the model now uses projections of weight vs. mpg by Greene and Duleep (1998) rather than a historical statistical relationship between weight and mpg. On the basis of the projections of Greene and Duleep (1998), the following values are used for light-duty vehicles in the U. S.:

mpg, city cycle	0.0	15.0	24.0	28.5	50.0	58.0	71.5	101.0	500.0
weight empty (lbs)	6,000	4,500	3,600	3,350	2,981	2,641	1,975	1,781	1,700

The empty weight is without passengers or payload, but with a full fuel tank. The LEM calculates the vehicle weight for any input city cycle fuel economy by interpolating between the pertinent points in the table above. The LEM now also calculates a change in the materials composition as a function of fuel economy, as. discussed elsewhere.

3) A new parameter, the relative weight of the AFV powertrain and body, has been added, so that the user may model the effect on efficiency and hence emissions of assuming a lighter or heavier EV body or powertrain. In the base-case, however, all ICEV powertrains and bodies are assumed to weigh the same (Table 10).
4) The model now contains a “weight-compounding” factor, which adds or subtracts weight from the vehicle chassis and suspension as needed according to the difference in weight between the AFV and the baseline ICEV. This parameter, shown in Table 6, is expressed as pounds of additional chassis and suspension weight per pound of extra weight in the powertrain, fuel-storage system, or body. It makes the treatment of weight changes more realistic.

Fuel storage in light-duty vehicles

After reviewing two new studies, and reconsidering the original data and analysis presented in DeLuchi (1992), several of the estimates of pounds of fuel storage system per pound of fuel (expressed hereinafter simply as lb-tank/lb-fuel) have been revised. Table 10 shows the new estimates and notes the minor revisions. Major revisions to the estimates for pressure vessels for hydrogen and natural gas are discussed below.

Hydrogen. The two hydrogen storage options now are compressed gaseous hydrogen and liquefied hydrogen, rather than metal-hydride storage and liquefied hydrogen. Pressure vessels are lighter and more compact than hydrides, for a given amount of hydrogen (Lipman and Delucchi, 1996), and probably less expensive as well (Berry and Aceves, 1998)[14].

Berry and Aceves (1998) report that a 1996 DOE study of onboard hydrogen storage systems for LDVs estimates that a 5,000-psi carbon-fiber wrapped vessel with a metallized polymer liner weighs 7 to 10 lbs per lb of fuel. Chalk et al. (1998a) report that present 5000-psi systems weigh 14 lbs per lb of fuel, but that this could be lowered with new high pressure tanks. They also note that the DOE goal is 6.5 lbs per lb. These

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[14] The best option may be a combination of low-temperature and high-pressure storage. Berry and Aceves (1998) believe that a hybrid high-pressure (5000 psi)/cryogenic system will cost less (per kg of H_2 stored) than either a low-pressure cyrogenic tank or an ambient-temperature high-pressure tank. They use data from Richards et al. (1996) to estimate that the system would weigh 12.8 lbs per lb of H_2.

are much lower than DeLuchi's (1992) estimate of 21 lbs/lb for carbon-wrapped aluminum-lined vessels estimated. The difference presumably is due to improvements in the strength-to-weight-ratio of carbon fiber, and the use of a metallized polymer liner rather than an aluminum liner. I assume the upper-end value from Berry and Aceves (1998), with an allowance for the extra weight of auxiliary equipment such as regulators, pumps, mounting brackets, and heavy-gauge fuel lines (Table 10). For the purpose of estimating emissions related to the materials lifecycle, I assume that the carbon-fiber-wrapped metallized-polymer-liner pressure vessel is 75% plastic/composites, 10% aluminum, 10% high-strength steel, and 5% stainless steel.

Natural gas. Richards et al. (1996) analyzed the weight and cost of steel, aluminum, and plastic CNG (3000 psi) storage cylinders. The lightest low-cost option was high-strength steel (about 0.2 lbs/SCF [cylinder only] and $0.30/SCF [OEM selling cost]), and the cheapest low-weight option was carbon-fiber-wrapped plastic (about 0.07 lbs/SCF [cylinder only] and $1.06/SCF [OEM selling cost]). (Liss et al. [1998] show similar figures.) The weight figures correspond to 4.3 lbs/lb for steel, and for 1.5 lbs/lb for carbon/plastic. With auxiliary equipment, these figures probably would be about 4.5 and 1.7. Because the high-strength steel vessels are considerably less expensive, and also safer, than the fiber-wrapped plastic-lined vessels, these are assumed to be used. The assumptions are shown in Table 10.

For the purpose of estimating emissions related to the materials lifecycle, the high-strength-steel pressure vessel and auxiliaries are assumed to be 90% high strength steel, 5% stainless steel, and 5% aluminum.

See also the U. S. DOE (1992).

Choice of LNG or CNG and LH_2 or CH_2

The process of modeling liquefied natural gas (LNG) and liquefied hydrogen (LH_2) has been reduced to a single toggle. Before, in order to switch from CNG (compressed natural gas) to LNG, or CH_2 (compressed hydrogen) to LH_2, one had to change several parameters values, and copy data from one column to afnother, throughout the model. Now, one specifies a set of input data (once), and switches between CNG and LNG and CH_2 and LH_2 with a single toggle.

Lifetime of vehicles

The vehicle lifetime (LVMT), which is used in the calculation of g/mi emissions due to materials manufacture and vehicle assembly, and formerly was input directly, now is estimated on the basis of the year-by-year VMT and survival probability of each model-year vehicle. (The year-by-year VMT schedule also is needed to calculate the model year of the vehicle given an assumption regarding a target year of analysis, and accumulated VMT in the target year. See the discussion of the estimation of emission factors in a target year.) Formally:

$$LVMT_{MY} = \sum_{A} SP_{MY,A} \cdot VMT_{MY,A}$$ eq. 41

where:

$LVMT_{MY}$ = lifetime VMT of model-year MY.
$SP_{MY,A}$ = the survival probability of model year MY at age A (Table 13).
$VMT_{MY,A}$ = the annual VMT of model year MY at age A (Table 13).

LVMT is calculated separately for LDVs, LDTs, and HDTs. The resultant lifetime vehicle miles are substantially higher than the assumptions in Table P.2 of DeLuchi (1993).

MOTOR VEHICLES: FUEL-CELL VEHICLES

In the LEM, lifecycle emissions depend on vehicular energy consumption, which in turn depends on vehicle weight. In other words the life cycle emissions is a function of driving range and the unit weight (e.g., lbs/kW) of major components such as batteries and fuel cells. In this section, we document our assumptions regarding the unit weight and efficiency of fuel-cell systems.

The U. S. Department of Energy (DOE) has sponsored detailed studies of the performance of fuel cell vehicles. The Allison Gas Turbine Division of General Motors (1994), under contract to DOE, performed conceptual design studies for an optimized fuel-cell vehicle with a proton-exchange membrane (PEM) and methanol fuel processor. The complete optimized electrochemical engine -- fuel-cell stack, fuel processor, and thermal and water management and system control -- weighed about 9 lbs/kW. The weight was “approximately equally divided” between the three parts (stack, processor, auxiliaries). The reformer was about 77% efficient.

Chalk et al. (1998a) report the following DOE technical targets for fuel-cells stacks and reformer systems:

	1997	2000	2004
fuel cell stack (lbs/kW)	7.4	6.3	4.4
fuel cell stack efficiency @ 25% of peak power (LHV, %)	50	55	60
fuel processor (lbs/kW)	5.5	3.7	2.9
fuel processor efficiency (%)	70	75	80

We adopt the 1997 values of Chalk et al. (1998a) for 1996, and assume minimum values of 3.0 lbs/kW (stack) and 2.0 lbs/kW (reformer), and maximum values of 8.5 lbs/kW (stack) and 7.0 lbs/kW (reformer). We also assume that system auxiliaries are

2.5 lb / kW in 1996, with a minimum of 1.0 lbs / kW and a maximum of 3.5 lbs / kW. (In the LEM, auxiliaries are included with the stack.)

MOTOR VEHICLES: EMISSIONS

Emission-factor model

In the real world, emissions from motor vehicles are a function of the driving pattern, the ambient conditions, and the characteristics of the engine, fuel, and emission-control system. Engines, fuels, and emission-control systems sometimes can change from one year to the next -- for example, as a result of a change in emissions regulations. Moreover, over the life of any particular vehicle, the emissions usually increase as the engine and emission-control system deteriorate or fail with accumulated mileage.

Detailed emissions model, such as the EPA's MOBILE model, estimate emissions in a target year as a function of zero-mile emissions rates by model year, emission-deterioration rates, average speed, temperature, and other factors. In the MOBILE emissions models and in reality, the model year and the accumulated mileage in the target year are important variables. However, as far as I know, no fuelcycle emissions model properly estimates motor-vehicle emissions as a function of model year and accumulated mileage in a target year. Rather, all of the existing models, including the previous version of this one, simply have estimated or assumed fleet-average emissions in some target year.

In the revised model documented here, emissions in a target year are estimated as a function of the zero-mile emission rate, the emissions-deterioration rate, the accumulated mileage in the target year, and the annual mileage accumulation. For LDGVs, the zero-mile emissions and the emissions deterioration rate are estimated as a function of the model year. Formally:

$$EM_T = ZM_{MY} + DR_{MY} \cdot \frac{Mi_T}{10000}$$

$$MY = T - AGE$$

$$AGE = f(AMS, Mi_T)$$

For LDGVs :

$$ZM_{MY}, DR_{MY} = f(V_U, V_B, V_L, T_B, T)$$

For HDDVs :

$$ZM_{MY} = ZM_{BMY} \cdot \left(1 + \frac{\Delta ZM}{100}\right)^{MY - BMY}$$

$$DR_{MY} = constant \qquad \text{eq. 42}$$

where:

EM_T = emissions in target year T (g / mi).
ZM_{MY} = zero-mile emissions for model year MY (Table 12; see discussion below).
DR_{MY} = the deterioration rate in emissions for model year MY (Table 12; see discussion below).
Mi_T = total mileage on the vehicle in the target year T (miles) (specified by user; in the cases presented here, assumed to be half of the life of the vehicles)[15].
MY = model year of the vehicle (calculated on the basis of the vehicle mileage in the target year).
T = target year of the analysis (specified by the user).
AGE = the age of the vehicle (years).
AMS = the annual mileage accumulation schedule (Table 13)[16] .

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[15] Note that first the user specifies the mileage on the vehicle in the target year, and then the model looks up the age of the vehicle and calculates the model year, rather than the other way around. In this way, one does not have to worry that the model year is after the target year.

ZM_{BMY} = zero-mile emissions from a base-model-year vehicle (Table 12; see discussion below.

?ZM = the annual percentage change in the zero-mile emission rate (Table 12).

BMY = base model year for setting emission factors (1993).

This model is used to estimate emissions from baseline gasoline and diesel vehicles. The parameter values are shown Table 12, and discussed in the following sections. The final calculated g/mi emissions are shown in Table 2.

Emission-factor parameters for CO, NMOC, and NO_X emissions from light-duty gasoline vehicles

In the previous version of the model, the baseline CO, NMOC, and NO_X g/mi emission factors were estimated by adjusting the output of MOBILE4.1 to account for the effects of the then-new 1990 Clean Air Act Amendments (Tables B.2 and B.3). As noted above, the model now estimates emissions as a function of a zero-mile emission rate and an emissions-deterioration rate for each model year.

Since the publication of DeLuchi (1993), it has become clear that EPA's MOBILE emission factor model, presently in version 5C, underestimates emissions from light-duty vehicles. The main problems are (Delucchi and McCubbin, 1996):

- It does not account for or properly represent the significant increase in emissions during high speeds, hard accelerations, and steep climbs, mainly because the official emissions test, the FTP, does not run vehicles at high engine loads. Because these emissions result from loads not "in" the official test regime, they usually are called "off-cycle" emissions.
- It probably underestimates the total number of starts that occurred with a cool or cold catalyst.
- It does not represent well the effect of air conditioning on emissions (the use of air conditioning greatly increases NO_X emissions).

MOBILE6 (available from the US EPA) should have corrected these and other problems in MOBILE5, including some that tend to cause a minor over-prediction of

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[16] There is a difference between annual miles of travel as model-year X ages, which is what we wish to know, and annual miles of travel by vehicle age in year X, which is what usually is reported. The latter confounds the change in VMT as a particular vehicle ages with changes in annual VMT in each new model year. If, as appears to be the case, annual VMT increases with each new model year, then in a survey in year X, annual VMT declines with vehicle age more rapidly than annual VMT will decline as model-year X ages in the future, because in the future people will be drive more. (The difference between model-year X and calendar-year X is of no consequence here.) The analysis of Table 13 separates these effects.

emissions. However, in the meantime, we must look to other sources for an estimate of emissions from motor-vehicles as they really are driven.

Zero-mile emissions (ZM_{MY}). Assumptions for the upper-limit zero-mile emission rate, the lower-limit zero-mile emission rate, the base-year zero-mile emission rate, and the steepness parameter (see Eq. 6) are shown in Table 12. The *upper- limit* zero-mile emissions are on the basis of emissions from completely uncontrolled, high-emitting vehicles. For reference, the EPA (1985/1991) reports the following zero-mile g/mi emission rates for pre-1968 vehicles at low altitude:

EPA zero-mile estimates	CO	HC exhaust	HC evap.	NO_X
pre-1968	78	7.3	> 6.0	3.4

Davis (2000) reports similar estimates for pre-1968 vehicles: 80 g/mi CO, 11 g/mi total HC, and 4.0 g/mi NO_X.

I assume that the *lower limit* on zero-mile CO and NMOG exhaust emissions is just above estimated emissions from the combustion of the engine lubricating oil alone (discussed elsewhere in this report). In the case of NO_X, the lower limits are estimated on the basis of U. S. Federal Tier 2 and California "ULEV" (ultra-low-emission-vehicle) emission standard of 0.07 g/mi (Walsh, 2002; Davis, 2000; the California "SULEV" – super-low-emission-vehicle – standard is lower, 0.02 g/mi). In the case of evaporative emissions, judgment was used to estimate the lower limit.

Base-year zero-mile emission rates were determined on the basis of estimates in Ross et al. (1998). Ross et al. (1998) performed a detailed analysis of emissions that result from malfunctioning emission-control equipment, air conditioning, and high-power driving not represented in the standard emissions test (the Federal Test Procedure, or FTP). They add these malfunction and so-called "off-cycle" emissions to "on-cycle" emissions from properly functioning cars, as estimated by MOBILE5, to obtain an estimate of real-world emissions of CO, NMOCs, and NO_X from model-year 1993, 2000, and 2010 passenger cars using conventional gasoline. Their estimates of the lifetime average tailpipe emissions are as follows:

lifetime average (g/mi)	CO	HC exhaust	NO_X
MY 1993	14.2	1.2	1.5
MY 2000	10.6	0.8	1.0
MY 2010	4.4	0.4	0.6

Zero-mile emissions were estimated by subtracting from the above lifetime averages their estimates of emissions due to "degradation" and malfunction:

zero-mile (g/mi)	CO	HC exhaust	NO_X

MY 1993	6.1 (12.1)	0.39 (0.99)	0.60 (1.05)
MY 2000	3.8 (8.8)	0.24 (0.64)	0.36 (0.71)
MY 2010	2.3 (4.3)	0.13 (0.33)	0.23 (0.39)

The result if one subtracts only degradation emissions from the average are shown in parentheses -- i.e., if one decides that malfunction emissions should be part of zero-mile emissions. I believe that malfunction emissions should not be part of zero-mile emissions, but rather should be incorporated into the emissions deterioration rate.

Ross et al. (1998) use MOBILE5A to estimate vehicular evaporative emissions of 0.37 g / mi for model years 2000 and 2010. However, MOBILE5A includes emissions from refueling in its estimates of vehicular evaporative emissions, and because refueling emissions are counted separately as “fuel dispensing” emissions, the refueling-emission portion must be deducted from the Ross et al. (1998) estimate of total vehicular emissions. I assume that the refueling emissions included in the Ross et al. estimate, and to be deducted here, are about 0.1 g / mi.

Emissions deterioration rate (DR). Ross et al. (1998) also estimate lifetime average g / mi emissions from “degradation” and malfunction for model-year 1993, 2000, and 2010 vehicles. Assuming that the lifetime average that they show is the same as the rate at the midpoint of the vehicle life, the g / mi / 10,000 mi deterioration factors (degradation plus malfunction) were estimated from the Ross et al. (1998):

deterioration (g/mi/10k mi)	CO	HC exhaust	NO_X
MY 1993	1.2 (0.3)	0.12 (0.03)	0.14 (0.07)
MY 2000	0.9 (0.3)	0.07 (0.02)	0.09 (0.04)
MY 2010	0.4 (0.1)	0.04 (0.01)	0.05 (0.02)

The deterioration factors shown in parentheses count only degradation; i.e., counting malfunction emissions as part of zero-mile emissions rather than as part of deterioration. (The parenthetical deterioration-rate estimates here correspond to the parenthetical zero-mile estimates above.)

In a 1989 update to its documentation of its mobile-source emission-factor model, the EPA (1985 / 1991, Table 1.1.1a) shows two sets of deterioration rates, in g / mi / 10000-miles, for exhaust HC, CO, and NO_X. The first set applies up to 50,000 miles of life, and the second set applies after. Each set shows the g / mi / 10,000-miles deterioration rate for each model year from pre-1968 to post-1992. The following shows the average of the two rates for selected model years in the EPA estimates:

deterioration (g/mi/10k mi)	CO	HC	NO_X

MY pre 1968	2.3	0.18	0.0
MY 1985	0.91	0.074	0.035
MY post 1992	0.84	0.066	0.034

The EPA rates for deterioration are reasonably consistent with the Ross et al. (1998) estimates. The Ross et al. (1998) estimates were used for the base-year estimates, using the EPA estimates as a guide in setting an upper limit. Judgment was used to set the lower limits. I assume that deterioration factors continue to decline with models after 2010, in part because on-board diagnostic equipment will help keep the emission control system operating properly. Also, it appears that the deterioration rates for recent model year vehicles may be lower than the EPA has estimates. In a recent study, 1988 and later model-year vehicles had substantially lower emissions at high mileage than did 1985 to 1987 model-year vehicles (Walsh, 1996).

Because the CEFs for these pollutants are relatively small, the changes in the emission factors have only a minor effect on CO_2-equivalent GHG emissions.

Emission-factor parameters for CO, NMOC, and NO_X emissions from heavy-duty diesel vehicles

Zero-mile emissions in base year (ZM) and change in zero-mile rate (?ZM). There are no definitive estimates of the extent to which MOBILE5 mis-estimates emissions from HDDVs in the real world. Therefore, my estimates (Table 12) are based on the analysis and data in Appendix B of DeLuchi (1993), and consideration of the NO_X standards for HDDVs, which are as follows (EPA, *Emission Standards Reference Guide for Heavy-Duty and Non-Road Engines,* 1997; Davis, 2000, Walsh, 2002; www.dieselnet.com)[17]:

Model year	g/bhp-hr	g/mi (assuming 2.6 bhp-hr/mi)
1983 and earlier	no standard	no standard
1984-1989	10.7	27.8
1990	6.0	15.6
1991-1997	5.0	13.0
1998 –2003	4.0	10.4
2004 – 2006	2.0	5.2
2007-2010+ (phased in)	0.2	0.5

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[17] There also are standards of 1.3 g/bhp-hr for HC and 15.5 g/bhp-hr for CO for model years 1984 and on (Davis, 2000). These however are easily met.

Standards for urban buses are similar.

Diesel vehicles with catalyzed particulate traps, low-NOx engine calibration, and ULSD fuel (less than 15 ppm S) apparently can meet the 2007 standard for PM but cannot yet come close to the 2007 NOx standard. For example, a diesel bus with a continuously regenerating particulate trap, low-NOx engine calibration, and 11 ppm S diesel fuel emitted about 15 mg/mi PM, but over 20 g/mi NOx, over all driving cycles (Ayala et al., 2002). This low PM emissions are impressive, but the NOx emission are more than order of magnitude above the 2007 standard. Indeed, the NOx emissions exceeded even the 1990 standard.

It thus is not yet clear how a heavy-duty diesel vehicle will meet the NOx and PM standards at the same time. Nonetheless, I assume that they will. (The NOx standard is to be phased in through 2010.)

Emissions deterioration rate (DR). The EPA (1985/1991, Table 1.7.1) assumes no deterioration in emissions of HC or NOx from 1979 MY and later vehicles, and relatively little deterioration in CO emissions. However, it is likely that some old, worn-out, out-of-tune, and occasionally malfunctioning HD engines burn fuel less efficiently, and therefore emit more unburned, or incompletely burned fuel (and PM), as well as more CO. (NO_X emissions do not necessarily increase in such circumstances, and in fact might even decrease.) Therefore, small but nonzero deterioration rates for CO and NMOC were assumed.

CH_4 emissions from gasoline LDVs and diesel HDVs

Methane emissions were estimated on the basis of new emissions data (Appendix F; EPA, 1999c) and a reconsideration of some of the data in Table M.1 of DeLuchi (1993). The EPA (1999c), for example, estimates that California low-emission vehicles emit 0.04 g/mi; vehicles with advanced 3-way catalysts, 0.05 g/mi; vehicles with early 3-way catalysts, 0.06 g/mi; and uncontrolled vehicles, 0.22 g/mi. The EPA's MOBILE model indicates that uncontrolled vehicles can emit more than 0.30 g/mi (Appendix F).

An upper-limit zero-mile rate (V_U in Eq. 6) and a base -year zero-mile rate (V_B in Eq. 6) for LDGVs were estimated on the basis of estimates from MOBILE5 (Appendix F) and EPA (199c). The lower-limit zero-mile emission rate (V_L in Eq. 6) for LDGVs is assumed to be just above the rate due to combustion of lubricating oil alone (estimated elsewhere in this report). The assumptions are shown in Table 12. Note that zero-mile methane emissions decrease less rapidly over time than NMOC emissions because methane *per se* is not regulated, and is not as effectively oxidized as are NMOCs by catalytic converters.

For any given model year, methane emissions were assumed to rise slowly with the age of the catalyst.

The emission factor for HDDVs, shown in Table 12, is based on the data in Appendix F, and is similar to the factor in DeLuchi (1993).

N_2O emissions from gasoline LDVs and diesel HDVs

Appendix F presents a comprehensive analysis of the available data on N_2O emissions from LDGVs and HDDVs. (See also Feijen-Jeurissen et al. [2001] for a good discussion of N_2O emissions from motor vehicles.) These data were used to estimate the parameters in the logistic function (Eq. 6) that represents the change in zero-mile emissions with model year, and the change in emissions with vehicle age, by model year. The main points drawn from Appendix F and used here in the estimation of the emissions functions for LDGVs are:

- emissions from uncontrolled vehicles, without catalytic converters, can be quite low -- less than 0.010 g/mi;
- lifetime average emissions from vehicles with ca. 1980-1990 emissions control appear to be over 0.100 g/mi, and even over 0.200 g/mi in certain cases;
- emissions from future “low-emission” vehicles probably will be less than emissions from 1990s vehicles, even though N_2O emissions are not regulated, because some of the techniques used to reduce regulated emissions also reduce N_2O emissions;
- N_2O emissions from LDVs equipped with a 3-way catalyst are a function of the age of the catalyst.

Consequently, instead of assuming a constant emission factor of 60 mg/mi for the life of gasoline LDVs (as in the Table B.2 of DeLuchi [1993]), Emissions are calculated as a function of zero-mile emission and deterioration rate. The assumptions are shown in Table 12. The data summarized above indicate that both the zero-mile rate and the deterioration rate rise with model year through about 2005, then decline with model year thereafter. Consequently, two logistic functions were used: one for model years 2005 and earlier that rises steeply from 1970; and a second for post-2005 model years that drops from 2005 on (Table 12). These assumptions result in emission rates higher than 60 mg/mi for most target years.

The emission factor for HDDVs is based on the data in Appendix F, and is similar to the factor in DeLuchi (1993).

PM emissions from gasoline LDVs and diesel HDVs

The EPA has a separate model, called PART5, that estimates emissions of SO_X and PM from motor vehicles (see EPA, *Draft User's Guide to PART5: A Program for Calculating Particulate Emissions from Motor Vehicles*, 1995). Evidence presented in

Delucchi and McCubbin (1996) and McCubbin and Delucchi (1999) suggests that the model underestimates in-use PM emissions, most likely because the emission factors were developed from a few tests on relatively low-mileage, properly tuned vehicles driven over a standard drive cycle. Hence, the predictions of PART5 do not account for very high emissions from old, poorly tuned, or malfunctioning vehicles, or (in the case of LDvs) for high emissions from hard accelerations that are not part of the standard emissions tests procedure.

Zero-mile emissions. Given this, the PART5 estimates might reasonably well represent zero-mile emissions from well running vehicles of model year 1990 and earlier. With this consideration, an upper-limit zero-mile rate (V_U in Eq. 6) and a base-year zero-mile rate (V_B in Eq. 6) for LDGVs were estimated on the basis of estimates in PART5 and McCubbin and Delucchi (1998). In addition, the lower-limit zero-mile emission rate (V_L in Eq. 6) for LDGVs is just above the rate due to combustion of lubricating oil alone. Lubricating-oil emissions of PM are 0.002 g/mi, which is consistent with data in Durbin et al. (1999), Cadle et al. (1998) and Mulawa et al. (1997) showing that emissions of *total* PM from new, late-model, properly functioning LDGVs are in the range of 0.002 to 0.003 g/mi. Assumptions are shown in Table 12.

In the case of HDDVs, zero-mile PM emissions were estimated on the basis of the estimates of PART5, the additional analysis in Delucchi and McCubbin (1996) and McCubbin and Delucchi (1999) (see also the review in Yanowitz et al. [2002]) and the EPA's PM emissions standards, which are as follows (EPA, *Emission Standards Reference Guide for Heavy-Duty and Non-Road Engines,* 1997; Davis, 2000; Walsh, 2002; www.dieselnet.com):

MY 1987 and earlier:	no standard
MY 1988-1990:	0.6 g/bhp-hr
MY 1991-1993:	0.25 g/bhp-hr
MY 1994-2006:	0.10 g/bhp-hr
MY 2007+:	0.01 g/bhp-hr

Diesel vehicles with catalyzed particulate traps, low-NOx engine calibration, and ULSD fuel (less than 15 ppm S) apparently can meet the 2007 standards for PM (Ayala et al., 2002).

All assumptions are shown in Table 12.

Emissions deterioration. Unlike PART5, I do assume that PM emissions increase as vehicles age[18]. Emissions-deterioration rate parameters were used that result in life-

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[18] It is not just that PART5 assumes a zero deterioration rate, it does not even have a deterioration function for [carbon] PM emissions. There is but one emission factor for each model year and emission-control category (EPA, *Draft User's Guide to PART5: A Program for Calculating Particulate Emissions from Motor Vehicles,* 1995).

time average emissions that are consistent with the emissions analysis presented in Delucchi and McCubbin (1996), McCubbin and Delucchi (1999), and other sources. The deterioration rate decreases slightly with each model year.

I assume that PM emissions are 77% carbon by weight (Cadle et al., 1998; Williams et al., 1989b), and subtract this carbon from the total fuel carbon available when calculating CO_2 emissions from fuel combustion. Generally, the amount of carbon emitted in PM is quite small compared with the amount emitted as CO or CO_2, but very badly smoking vehicles can convert appreciable amounts of fuel carbon to particulate carbon.

Sample results

With the input assumptions given in Table 12, the LEM estimates the following g/mi emission rates for gasoline LDVs, by model year:

MY:	1966	1975	1985	1995	2005	2020	2045
Fuel evaporation	3.09	1.98	1.16	0.67	0.40	0.22	0.16
NMOC exhaust	3.77	2.59	1.62	0.97	0.57	0.30	0.20
CH_4 exhaust	0.21	0.15	0.10	0.07	0.05	0.03	0.02
CO exhaust	36.02	26.74	18.60	12.10	7.35	3.49	1.57
N_2O exhaust	0.003	0.060	0.124	0.133	0.138	0.064	0.040
NO_2 exhaust	3.14	2.36	1.70	1.20	0.82	0.45	0.20
PM exhaust	0.115	0.082	0.057	0.040	0.028	0.019	0.015

Diesel LDVs and gasoline HDVs

In the model, diesel LDVs and gasoline HDVs are treated like alternative fuels, in the sense that one enters emission rates relative to gasoline LDVs or diesel HDVs, and not absolute g/mi emission factors (see discussion of alternative fuels elsewhere in this major section). Estimates of relative emissions of CO, NMOC, NO_x, and PM are taken from the EPA's MOBILE and PART5 databases (EPA, *AP-42 Vol. 2*, 1991; EPA, *Draft User's Guide to PART5: A Program for Calculating Particulate Emissions from Motor Vehicles*, 1995). In the case of PM, we note that Durbin et al. (1999) found that PM emissions from LDDVs are one to two orders of magnitude higher than PM emissions from LDGVs, a finding consistent with the assumptions in PART5.

Estimates of relative N_2O and CH_4 emissions for diesel LDVs and gasoline HDVs are based on the data and analysis presented in Appendix F to this report. The data indicate that, for post-1990 model-year vehicles, CH_4 emissions from diesel vehicles are on the order of 50% of CH_4 emissions from gasoline vehicles. In the case of N_2O, the limited data indicate that it is most reasonable to assume that diesel engines emit roughly the same amount of N_2O as do gasoline engines of a similar size and emission control. This might indicate a factor of 10-30 mg / mi for diesel LDAs, and 40-60 mg / mi for diesel HDVs. These factors probably are on the order of 25% of those for gasoline vehicles with 3-way catalysts.

Emissions related to the use of lubricating oil

The gradual oxidation of lubricating oil produces CO_2, CO, CH_4, NMOCs, SO_x, and PM. The production lifecycle of the lubricating oil also produces emissions. The CO_2 emissions can be estimated on the basis of the carbon content and the consumption rate of oil. SO_2 emissions can be estimated on the basis of the sulfur content and the oil consumption rate. Production lifecycle emissions can be estimated on the basis of the consumption rate and the production lifecycle emission rate. Emissions of CO, CH_4, NMOCs, and PM can be estimated directly, on the basis of actual measurements, and other considerations. (It is necessary to distinguish the portion of organic emissions from lubricating oil because these emissions do not get any biofuel-carbon credit.)

The use of lubricating oil. The estimation of the use of lubricating oil by motor vehicles is based on the total consumption of lubricants in the U. S. The basic premises of the calculation are:

i) that the use of virtually all lubricants is related in one way or another to the use of fuel for engines; and

ii) that this relationship is best expressed in terms of the heating value of the fuels presently used. In 1997, the ratio of the weight of lubricants supplied in the U. S. to the HHV of gasoline, distillate, and jet fuel supplied was 322 g / 10^6 BTU (based on data in EIA, *PSA 1997*, 1998). (The ratio of retail sales of automotive lubricants to retail sales of automotive fuels is similar[19].) Multiplying this by 0.0042 10^6-BTU-fuel / mi results in 1.35 g-lube-oil / mi, which is close to the value in DeLuchi (1993).

This new method of estimating the use of lubricant per mile has two advantages over the old method of DeLuchi (1993). First, it gives a more accurate accounting of the use of lubricating oil, because it is based on total lubricating oil use in the U. S. Second,

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[19] In 1992, retail stores sold $114 .7 billion worth of automotive fuels, and $3.5 billion worth of automotive lubricants (Bureau of the Census, *1992 Census of Retail Trade, Merchandise Line Sales,* 1995). The average price of automotive fuel sold by service stations (which sold the bulk of all gasoline sold retail) was $1.12 (based on dollar sales reported in *1992 Census of Retail Trade, Merchandise Line Sales,*, and total gallons reported in Bureau of the Census, *1992 Census of Retail Trade, Miscellaneous Subjects,* 1995). Dividing the total sales by $1.12 / gallon results in 102.4 billion gallons of automotive fuel, or 12.8 billion million BTU, sold at retail in 1992. Assuming that the lubricants sold at $0.75 / quart excluding sales taxes, there were 1.17 billion gallons, or about 4,000 billion grams, of automotive lubricants sold in 1992. The resulting ratio is about 310 g-lube oil / 10^6-BTU-fuel.

because it assumes that the consumption of lubricating oil is proportional to fuel consumption, it results in g / mi consumption of lubricating oil being related automatically to the fuel economy of the vehicle.

Not all of the lubricant supplied annually oxidizes in use, or immediately after use. Some of it is permanently sequestered in the environment, and some is recycled back to consumers and so re-appears in the EIA's supply statistics.

The foregoing is for petroleum-fuel vehicles. For alternative fuel vehicles, the consumption rate is equal to the rate for petroleum fuel vehicles multiplied by the relative oil consumption rate (Table 12).

To avoid double counting, the initial lubricant fill is not counted as a "material" in the analysis of emissions from vehicle materials and assembly.

CO_2 emissions from the use of lubricating oil. CO_2 emissions are from the use of lube oil are calculated on the basis of the difference between total carbon available for oxidation, and carbon emitted in compounds other than CO_2:

$$CO2_{\mathrm{lub}\,e} = \left(CF_{\mathrm{lub}\,e} \cdot LOC \cdot SFO \cdot FC - NCM_{\mathrm{lub}\,e}\right) \cdot \frac{MW_{CO2}}{MW_{C}} \qquad \text{eq. 43}$$

where:

$CO2_{\mathrm{lube}}$ = net CO_2 emissions from the use of lubricating oil related to motor-vehicle use (g / mi).
CF_{lube} = the weight fraction of carbon in lubricating oil (assumed to be the same as in residual fuel oil).
LOC = the rate of consumption of lubricating oil, in grams of oil per BTU of fuel consumed (discussed above).
SFO = of total lubricating oil supplied annually, the fraction that oxidizes in use or immediately after; i.e., the fraction *not* eventually permanently sequestered in the environment, or recycled back to the petroleum refineries (I assume 0.90, as discussed below).
FC = the motor-vehicle fuel consumption rate (BTU / mi; calculated by the model as described elsewhere in this report).
NCM_{lube} = emissions of carbon in NMOCs, CH_4, CO, and PM in vehicle exhaust from the combustion of lubricating oil (discussed below).
MW_{CO2} = the molecular mass of CO_2 (Table 5).
MW_C = the molecular (atomic) weight of C (12.01 g / mole).

Of the annual supply of lubricants,I assume that 80% oxidizes during or after use, and 20% is recycled to consumers or permanently sequestered in the environment.

The fraction of oil that oxidizes during or shortly after use. Lubricating oil has several fates: some is combusted in vehicle engines, some is combusted in other non-transportation applications as used oil, some is landfilled, some is disposed of in storm sewers, and some is recycled. The EPA (2002) has estimated the percentage of oil that goes to each fate, and the percentage of carbon oxidized in each fate:

Fate	% of total oil use	% carbon oxidized
combusted during use	20	99
combusted as used oil	64	99
dumped on ground or in sewers	6	100
landfilled	2	10
re-refined	8	97

With these data, the EPA (2002) estimates that, on average, 97% of the carbon in lubricating oil eventually oxidizes. However, for our purposes the percentage of carbon oxidized in re-refined oil should be zero, because any subsequent of oxidation of re-refined oil in the motor-vehicle sector is assigned to the *next* use of the oil, not to the original use that we are modeling[20]. Thus, if in the fate accounting above we assume that 0% of re-refined oil oxidizes in the first use, then about 90% of the carbon in lubricating oil is oxidized per use by the transportation sector.

Non-CO_2 carbon emissions from the combustion of lubricating oil. DeLuchi (1993) assumes that 6% of measured tailpipe emissions of CO, CH_4, and NMOCs come from the combustion of lubricating oil. For two reasons, this figure appears too high. First, as documented above, the consumption of lubricating oil is only about 320 g-lube / 10^6-BTU-fuel, which corresponds to about 0.014 g-lube / g-fuel. If the formation of organic emissions per gram of lube oil consumed is the same as the formation per gram of fuel consumed, then lube oil contributes only 1.4% of total organic emissions. Second, comparison of emissions from properly functioning hydrogen vehicles with emissions from similar gasoline vehicles indicates that organic emissions from hydrogen vehicles -- which come entirely from the lubricating oil -- are on the order of 2% of organic emissions from gasoline vehicles (Sperling and DeLuchi, 1993). The difference between the 2% and the 1.4% implies that engine oil forms more emissions per gram than does the fuel . Hence, oil is less completely burned than the fuel, which seems likely.

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[20] Put another way, we are estimating emissions associated with *each* use of lubricating oil by the transportation sector, and CO_2 from oil that is recycled from first use to second use and then oxidizes in the second use gets assigned to the second use, not the first use.

However, lube oil might contribute a higher share of particulate emissions. Williams et al. (1989a) used ^{13}C-labeled lubricating oil to measure PM emissions from oil from a 1978 and a 1981 light-duty gasoline vehicle, and found that PM from lubricating oil was 15% of total emitted PM. It is not clear if they measured PM from lubricating oil, carbon in PM from lubricating oil, or carbon emitted in any form from lubricating oil. In any case, lubricating oil contributed an even greater fraction to PM emissions from diesel vehicles (Williams et al., 1989b).

With these data and considerations, the following emissions are assumed due to combustion of lubricating oil (g/mi):

	LDGV w/cc	LDGV wo/cc	HDDV
NMOC emissions	0.013	0.110	0.060
CH4 emissions	0.005	0.020	0.010
CO emissions	0.102	1.280	0.300
PM emissions	0.002	0.006	0.06

SO$_2$ emissions from fuel and lube oil. Emissions of SO$_2$ are calculated on the basis of the sulfur content of the fuel and lube oil, assuming that all sulfur is burned completely to SO$_2$. In reality, some of the sulfur is emitted as sulfate or H$_2$S, but the amounts of these are small compared to the amount of SO$_2$.

Note the previous version of the model did not include SO$_2$ emissions from lube oil. Now, SO$_2$ emissions from the baseline gasoline or diesel vehicle are calculated as:

$$SO2_{\mathrm{lub}\,e} = SF_{\mathrm{lub}\,e} \cdot \frac{MW_{SO2}}{MW_{S}} \cdot LOC \cdot SFO \cdot FC \qquad \text{eq. 44}$$

where:

SO2$_{\mathrm{lube}}$ = SO$_2$ emissions from the use of lubricating oil related to motor-vehicle use (g/mi).

SF$_{\mathrm{lube}}$ = the weight fraction of sulfur in lubricating oil (assumed to be the same as in residual fuel oil).

MW$_{\mathrm{SO2}}$ = the molecular mass of SO$_2$ (64.06 g/mole).

MW$_{\mathrm{S}}$ = the molar mass of S (32.06 g/mole).

LOC, SFO, and FC are as defined above for Eq. 43.

The lube-oil SO$_2$ emission rate for AFVs is assumed to be equal to the rate for the gasoline or diesel vehicle multiplied by the rate of oil consumption for the AFV relative to the rate for the gasoline or diesel vehicle. This relative rate is shown in Table 12.

These previously ignored SO_2 emissions from lubricating oil are not trivial. At about 0.024 g / mi for the baseline gasoline vehicle, and 0.14 g / mi for the baseline diesel vehicle, they are over one third of the sulfur emissions from fuel.

Emissions from the lube-oil production lifecycle. For simplicity, t the lube-oil production lifecycle, up to the point of end use, is the same as the residual-fuel lifecycle. Thus, "upstream" CO_2-equivalent emissions per mile is:

$$GMI_{\text{lub}\,e} = GLF \cdot EC_{\text{lub}\,e} \cdot GBTU_{\text{lub}\,e} \cdot FC \qquad \text{eq. 45}$$

where:

GMI_{lube} = CO_2-equivalent GHG emissions per mile, from the production lifecycle of lubricating oil.

GLF = grams of lubricating oil consumed for every 10^6 BTU of engine fuel (discussed above).

FC = the motor-vehicle fuel consumption rate (10^6-BTU / mi; calculated by the model.

EC_{lube} = the energy content of lubricating oil (0.00004246 10^6-BTU / g; equal to $6.065 \cdot 10^6$ BTU / bbl [EIA, *AER 1996*, 1997] divided by 142,842 g / bbl [EIA, *International Energy Annual 1996*, 1998]).

$GBTU_{lube}$ = CO_2-equivalent GHG emissions per 10^6-BTU of lubricating oil, from the production lifecycle of lubricating oil (assume value for residual fuel).

The emissions for alternative-fuel vehicles are equal to the emissions for petroleum vehicles, calculated as above, multiplied by the relative oil consumption factor.

Recall that the calculation of lubricating oil consumed per BTU of engine fuel (the parameter GLF) is based on the total amount of lubricant supplied, with no distinction between recycled and first-run lube oil. The lifecycle of recycled lube oil is different from the lifecycle of first-run lube oil and one in principle should distinguish the two streams. However, because the emissions are so small, the distinction is not worth the effort. On average, all lube oil supplied was assumed to have a lifecycle similar to that of residual fuel oil.

Input of heavy-duty vehicle emission factors: g/bhp-hr vs. g/mi

In the previous version of the model, emission factors for heavy-duty diesel vehicles (HDDVs) were input directly in grams / mile. However, the emission standards for HDVs actually are in grams / brake-horsepower-hour (g / bhp-hr), not grams / mile. If all HDVs meet a given g / bhp-hr standard, then the more efficient ones -- the ones that use fewer bhp-hrs per mile -- will emit fewer grams per mile. Therefore, in the new model, the input emission factors for HDDVs are in g / bhp-hr, and the gram / mile emissions then are calculated from the input g / bhp-hr data, and estimates of fuel density and BSFC:

$$GMI_{w,MY} = GBHP_{w,MY} \cdot BHPMI_{w,MY} \qquad \text{eq. 46}$$

where:

subscript w = the heavy-duty engine gross-vehicle-weight (GVW) classes (see below).
subscript MY = the heavy-duty vehicle model year.
GMI = emissions in grams per mile.
GBHP = emissions in g / bhp-hr (Table 12 and pertinent sections below).
BHPMI = the energy consumption (work) of the heavy-duty vehicle (bhp-hr / mi; Eq. 39).

This is the method used in the EPA's MOBILE model. The parameter values, discussed elsewhere in this report, are based on Browning's (1998a, 1998b) recent updates for MOBILE6.

Note that this formulation assumes that improvements in the bhp-hr-work / bhp-hr-fuel thermal efficiency will not reduce g / mi emissions. This assumes that the emission standards, and hence presumably the emission-control design bases of the manufacturers, are per unit of brake work output from the engine, not per unit of fuel input to the engine. If the standards were given and the emission controls designed per unit of fuel input to the engine, then improvements in thermal efficiency as well as improvements in bhp / mi energy use always would reduce gram / mi emissions. Alternatively, when the standards are given per mile of travel, improvements in thermal efficiency and bhp / mi energy use will reduce g / mi emissions only to the extent that manufacturers allow g / mi emissions to decline further below the standard as a "margin of safety".

Emission factors for AFVs: relative to gasoline LDVs and diesel HDVs

Previously, one entered emission factors for AFVs directly in grams / mile. Now, one enters for the AFVs a set of emission factors *relative* to actual g / mi emissions for the baseline gasoline ICEV. Thus, if before one entered 9.0 g / mi CO for the gasoline

ICEV, and 4.5 g / mi CH_4 for the NGV, one now enters 9.0 g / mi CO for the gasoline ICEV, and relative emissions of 0.50 for the NGV. To the extent that the *relative* emissions of AFVs are constant over time and technology, this simplifies the process of modeling the effect of a completely different set of emissions standards, or of emissions over time. One needs to change only the baseline g / mi emissions factors for the gasoline LDV or diesel HDV.

The relative emission factors, shown in Table 12, are based on estimates cited in Appendix B of DeLuchi (1993), and other literature published since then (Bevilacqua, 1997; Fanick et al, 1996; Auto / Oil Air Quality Improvement Research Program, 1996; NREL 1996; Lynd, 1996b; Kelly et al., 1996c; Whalen et al., 1996; U. S. DOE, 1995a, 1996[21]; Wang et al., 1993; Baudino et al., 1993; Appendix A to this report).

Criteria pollutants, for methanol, ethanol, CNG, and LPG HDVs. I have re-estimated the relative emission factors for alternative-fuel HDVs. For criteria pollutants, the main new sources of data are NREL (1996, 1997, 1998, 2002), EPA (2002a), Milkins and Edsel (1996), Ortech (1998), Storkman (1998), and Wang et al. (1993).

NREL (1996) tested 20 transit buses using CNG (model years 1991 to 1994), 10 100% methanol transit buses (model years 1992 and 1993), 10 ethanol (E93 and E95) transit buses (model years 1991 and 1992), 4 transit buses using 20% biodiesel(model year 1988), and 32 diesel buses (model years 1988 to 1993) on a portable heavy-duty chassis dynamometer, over the Central Business District driving cycle. The emissions results were:

	PM	NOx	HC	CO
All diesels without PM trap (g / mi)	1.48	27.37	2.39	11.86
Diesel with PM trap (% change vs. counterparts without trap)	-51%	4%	44%	254%
20% biodiesel (% change vs counterparts)	5%	4%	-15%	0%
Methanol (% change vs counterparts)	-85%	-57%	1686%	57%
Ethanol (% change vs counterparts)	-42%	-73%	289%	391%
CNG L10-240G (% change vs counterparts)	-99%	26%	529%	17%
CNG L10-260G (% change vs. counterparts)	-99%	-54%	546%	-94%

The EPA (2002a) has summarized publicly available data on the emissions impact of biodiesel, and found that 100% biodiesel has the following effects on emissions relative to diesel fuel (see also Appendix A to this report):

	PM	NOx	HC	CO
biodiesel relative to diesel	-48%	+10%	-67%	-49%

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[21] For a summary of USDOE research programs on advanced automotive technologies, see USDOE (1998).

More recently, NREL (2002) tested 13 CNG and 3 comparable diesel medium-duty delivery vehicles operated by United Parcel Service. The diesel vehicles were built in 1995, and had a Cummins/B5.9 engine, without a catalytic converter. The CNG vehicles were built in 1996 and had a Cummins B5.9 natural gas engine, with a catalytic converter. Tested on portable chassis dynamometer built by West Virginia University for medium-duty vehicles, the CNG vehicles had the following emissions relative to the diesel controls:

	PM	NOx	HC	CO
CNG relative to diesel (no trap)	-95%	-49%	+4%	-75%

NREL (1998) compares emissions from two E95 snowplows, operated in Minnesota, with emissions from a diesel control: PM -30% to -64%; NO_X -12% to +12%; HC +174% to +427%; CO +23% to +452%. NREL (1997), Norton et al. (1996), and ANL (1997) report similar emissions, relative to diesel, from 4 heavy-duty ethanol trucks with DDC 6V-92TA engines. My estimates from graphed results are: PM -65%, NO_X -19%, HC +280%, CO +290%. Thus, three separate studies -- on snow plows, HD trucks, and buses -- show that ethanol HDVs have moderately lower PM, considerably higher HC and CO, and perhaps lower NO_X than diesel vehicles.

Wang et al. (1993) tested 8 HD CNG vehicles (model years 1987-1992), 4 HD methanol vehicles (model years 1987-1992), and 14 HD diesel vehicles (model years 1985-1992) on a portable heavy-duty chassis dynamometer (the same one used later in the NREL study), over the Central Business District cycle. The average emission results were:

	PM	NOx	HC	CO
All diesel vehicles (g/mi)	1.2	31.2	2.6	18.7
Methanol (% change vs. diesel counterpart)	-78%	-48%	278%	-2%
CNG (% change vs. diesel counterparts)	-97%	-36%	265%	-94%

These are similar to the NREL (1996) results.

Milkins and Edsell (1996) report that 5 Cummins L10 CNG buses tested over three different drive cycles had emissions reductions of 96%-98% for NMOG, 96%-99% for CO, 30%-60% for NO_X, and 96%-97% for PM.

Storkman (1998) reports the latest emission-test results (heavy-duty engine transient test cycle) for the Cummins B5.9 series engine (g/bhp-hr, and % change relative to diesel)

Fuel (hp)	catalyst?	PM	NOx	NMHC	CO	HCHO
diesel (250)	yes	0.089	3.74	0.10 (THC)	0.96	

NG (195)	yes	0.016 / -82%	1.00 / -73%	0.25 / 250%	0.15 / -84%	0.022
LPG (195)	yes	0.013 / -85%	2.29 / -39%	0.76 / 760%	0.07 / -93%	

The Cummins NG and LPG B5.9 engines have advanced electronic engine management, closed-loop air / fuel ratio control, lean-burn technology (27:1 air / fuel ratio, instead of the stoichiometric 17:1), and optimized and integrated subsystems (Cummins, 1998). For reference, the emissions standards for the 1994-1997 model years are: 1.3 g / bhp-hr HC, 15.5 g / bhp-hr CO, 5.0 g / bhp-hr NO_X, and 0.1 g / bhp-hr PM. Note that the diesel engine reported by Storkman (1998) has unusually low PM emissions -- on the order of 0.3 g / mi (presumably because the engine must certify to an 0.1 g / bhp–hr standard. As a result, the percentage reduction in PM emissions with CNG and LPG is less than in the other tests shown above.

Ortech (1998) tested the effect of LPG fuel composition on emissions from a Cummins B5.9-195 (5.9L, 195 hp) LPG engine, over the EPA Heavy-Duty Transient Test Cycle. The propane content of the fuel varied from 76% to 95%, the propylene content from 3% to 21%, and the butane content from 2% to 20%. Emissions of NMHC, CH_4 and PM (g / bhp-hr) were only moderately sensitive to the fuel composition:

CH_4	NMHC	CO	NO_X	PM
0.029 - 0.046	0.59 - 0.82	0.32 - 0.82	2.9 - 3.6	0.006 - 0.008

Since the variation is moderate, and not evidently systematic with respect to fuel type, emissions from LPG vehicles were not related to the propane or butane content. Note that CH_4 emissions are similar to those from diesel engines.

F-T diesel from natural gas. Diesel fuel made from natural gas, via the F-T process, has virtually no sulfur, and relatively little aromatic content, and as result is relatively clean burning. Sasol (n.d.) reports 38% lower HC, 46% lower CO, 29% lower PM, and 8% lower NO_X than conventional diesel, and 20% lower HC, 35% lower CO, 24% lower PM, and 5% lower NO_X than “reformulated” diesel. SO_X emissions presumably are nearly eliminated. Given the quality of the fuel, these emissions reductions seem plausible. I assume that methane emissions are reduced by 10%, but that N_2O emissions are unchanged, compared to low-sulfur petroleum diesel.

CH_4 and N_2O, HDVs and LDVs. On the basis of analyses of new data, and a re-analysis of data from DeLuchi (1993), some of the relative CH_4 and N_2O emission factors have been revised. For example, I now assume that advanced NGVs emit 15 times as much CH_4 as comparable advanced gasoline vehicles. In addition, vehicles using 100% ethanol are assumed to emit 1.5 times as much CH_4 as do comparable advanced gasoline vehicles. The change in the ethanol-vehicle relative CH_4 emission factor, from 0.5 to 1.5, causes an increase of only 0.5% in fuelcycle CO_2-equivalent emissions. See Appendix F to this report for details.

There are no data on N_2O emissions from alternative-fuel HDVs, and few data on CH_4 emissions from alternative-fuel HDVs. In the absence of data, the ratio of emissions from advanced alternative-fuel HDVs to emissions from advanced gasoline HDVs is assumed to be the same as the ratio for advanced LDVs.

NO_X emissions, LDVs. In Appendix B of DeLuchi (1993), I assumed that all light-duty ICEVs will emit roughly the same amount of NO_X . In spite of the different NO_X emission characteristics of fuels . All ICEVs will be designed to just meet the relatively stringent new NO_X standards. This presumed that auto manufacturers will capitalize low-NO_X emissions potential into savings on emission-control equipment. In reality, though, manufacturers might not find it worthwhile to capitalize all of the potential emissions reductions into savings in emission-control equipment, and instead might prefer to meet the emissions standard with a greater margin of safety. This will result in some small variation in NO_X emissions across fuel types. Accordingly, I have assumed that alternative-fuel light-duty vehicles, which in emission tests generally emit slightly less NO_X than do gasoline vehicles, will have slightly lower NO_X emissions on the road.

PM emissions, LDVs. There are not many comparisons of PM emissions from alternative-fuel LDVs. Recently, Fanick et al. (1996) measured the size distribution of particulate emissions from a 1994 Ford Taurus operating “off-cycle” (fuel-rich) on five fuels: RFG, M85, E85, LPG, and CNG. The vehicle was programmed to run rich of stoichiometric, in effect simulating hard accelerations. They reported the following changes in g/mi particulate emissions, relative to RFG:

	M85	E85	CNG	LPG
all PM	-35	-47	-66	-79
PM < 3.0 µm	-31	-36	-64	-77
PM < 0.2 µm	-57	-63	-80	-71

“Off-cycle” emissions, LDVs. Virtually all emissions tests of light-duty alternative-fuel vehicles have been performed using the FTP, which as noted above does not represent high-speed, high-power driving. Recently, however, the Auto/Oil Program (1996) tested methanol, ethanol, CNG, and gasoline vehicles over a new high-speed, high-power drive cycle, the REP05, as well as over the FTP. The emissions from AFVs relative to the emissions from gasoline vehicles over the REP05 were different from the relative emissions over the FTP. Emissions from ethanol and methanol relative to emissions from gasoline were lower in the REP05 than in the FTP, but emissions of CO, NMHC, and NO_X from CNG relative to emissions from gasoline were higher in the REP05.

Bevilacqua (1997) found that dedicated NGVs emit significantly more CO, NMOG, and NO_X in the new high-acceleration phase of the supplemental FTP than in the FTP itself.

These findings are provocative, and warrant further investigation. For now, I have folded them into the emissions data base that serves as the basis of my assumptions in Table 12.

Gas loss from gaseous fuel vehicles

There are several kinds of gas loss from vehicles using CNG, LNG, CH_2, LH_2, or LPG:

- ordinary or “fugitive” leakage from the fuel system of the vehicle
- loss of fuel due to tank failure, for example as a result of an accident
- losses related to refueling the vehicle
- venting of evaporated liquefied gaseous fuel from a cryogenic tank (called “boil-off” loss)
- purging of unused hydrogen from fuel-cell vehicle stakcs

Many analysts, myself included, have assumed that there is no appreciable vehicular loss of a gaseous fuel, mainly because the vehicular fuel system is supposed to be sealed. For example, Milkin and Edsell (1996) write that “in the case of a CNG-fuelled vehicle, the fuel system is sealed, as is the fuel supply compressor-dispenser system,” and that as a result, the only leakage from vehicles is “a very small gas release when the CNG coupling is disconnected after refuelling” (p. 601). However, other analysts have a contrary opinion. Victor (1992) speculates that leaks due to “automobile accidents, poor maintenance, and tank purges...may be large, in addition to wellhead and pipeline leaks” (p. 129), but he does not estimate what the leakage rate might be[22].

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[22] Of course, we care about fuel leaks in garages or accidents primarily because they are dangerous, not because they pollute. In this regard, it is interesting to note the different behaviour of alternative fuels. Swain et al. (1998) used a model of fuel leakage and gas-cloud motion to determine the volume of combustible gas formed after 2 hours of leakage from the fuel line of a hydrogen vehicle, a propane vehicle, a CNG vehicle, and a gasoline vehicle in a closed single-car garage. They simulated leaks from a puncture (1000 L/h

In the following paragraphs, the scant evidence regarding gas loss from vehicles is reviewed. In this section, ordinary leakage emissions, loss due to tank failure, boil-off loss, and purging loss from fuel cells are considered. Emissions from refueling and refueling stations are estimated elsewhere in this report.

Ordinary fuel-system losses. Kelly et al. (1996c) measured diurnal and hot-soak evaporative emissions from CNG and gasoline vehicles, and found that CNG vehicles did indeed have "evaporative" emissions, albeit less than did gasoline vehicles: 0.38 - 0.57 g-THC / test for CNGVs vs. 0.59 to 1.42 g-THC / test for gasoline vehicles. They concluded:

> "There is some evaporative emissions leakage associated with the CNG fuel systems, but the mass is no more than would typically be expected from evaporative emissions in a corresponding gasoline vehicle. Any such leakage primarily consists of methane[23], a non-reactive and non-toxic compound which arises from many sources and is naturally released into the atmosphere (p. 11)."

These results indicate that diurnal and "hot-soak" evaporative emissions from CNGVs are about half those from gasoline vehicles. If this ratio applies to running-loss and resting-loss evaporative emissions as well, then, given that total evaporative emissions from a low-mileage gasoline vehicle are about 0.2 g / mi (Ross et al., 1998), a CNGV emits on the order of 0.1 g / mi. This is about 0.1% of the 80-g / mi fuel consumption rate of a CNGV comparable to a 30 mpg gasoline vehicle. Since CNGVs do not have control systems for evaporative emissions, the emission rate probably does not increase appreciably with mileage.

With these considerations, I assume a loss rate of 0.1% for CNG, as well as for LPG and LNG, for model-year 2000. I assume that this rate decreases by 1.5% / year, in relative terms. I assume that the rate for LNG is 25% higher than the rate for CNG, and that the rate for LPG is 100% higher, in relative terms. The rate for CH_2 relative to the assumed rate for CNG is calculated as follows:

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hydrogen, 300 L / h methane, and 118 L / h propane) and leaks from a crack (1000 L / h hydrogen, 680 L / h methane, and 602 L / h propane), in a garage with 2.92 air changes per hour. (The gasoline leakage was 1 drop / s.) Methane and hydrogen dispersed and did not produce appreciable volumes of combustible gas, but propane and gasoline pooled in combustible volumes. Specifically:

- for hydrogen and methane, the combustible cloud (at least 4.1% hydrogen or 5.3% methane by volume) did not extend more than 10 cm beyond the point of leakage;
- for propane, the combustible cloud (at least 2.1% propane by volume) covered half the floor after 2 hrs of 118 L / h leakage, and filled over 25% of the garage volume after 2 hrs of 602 L / h leakage;
- for gasoline, the combustible cloud (1.3% gasoline by volume) covered 80% of the floor after 2 hrs of 1 drop / s.

[23] Kelly et al. (1996c) did not actually measure individual hydrocarbons; they measured only TCH, and simply *assumed* that that the leaked gas was fuel, and hence mainly methane. It is conceivable components and materials other than those in the fuel system -- for example, coolant, lubricants, and plastics -- emit non-trivial amounts of hydrocarbons.

$$FL_{CH2} = FL_{CNG} \cdot RFL \cdot \left(\frac{PSI_{CH2}}{PSI_{CNG}} \right)^{0.5} \qquad \text{eq. 47}$$

where:

FL_{CH2} = the fuel-leakage rate for compressed hydrogen (CH2)
FL_{CNG} = the fuel-leakage rate for compressed natural gas (CNG) (discussed above)
RFL = the leakage rate for hydrogen relative to that for NG, for a given pressure and system design (discussed below)
PSI_{CH2} = the storage pressure of hydrogen (Table 35)
PSI_{CNG} = the storage pressure of natural gas (Table 35)

The key parameter in this analysis is the leakage rate for hydrogen relative to that for NG, at a given pressure for a given system design. Because hydrogen molecules are much smaller and lighter than methane molecules, one might expect that the leakage rate of hydrogen would be much greater than the leakage rate of natural gas. However, because hydrogen leaks are dangerous and even costly, it may be worthwhile to build and operate hydrogen systems so that the leakage rate is equal to or less than that for natural gas. I assume that the ratio of the hydrogen leakage rate to the natural-gas leakage rate at a given pressure is 1.50 – less than what would be expected purely on the basis of the relative mobility of the two gases, but probably higher than what could be achieved with best practice.

This method assumes that the leakage rate varies with the square root of the storage pressure.

For LH_2, a rate of 0.2% is assumed, not including venting of boil-off gases, which is estimated separately, below.

Tank failure. A simple calculation reveals that gas loss due to tank failures, as a result of fires, accidents, vandalism, improper maintenance and operation, and so on, is utterly insignificant. Worldwide, since 1976, there have been only 16 known ruptures of CNG cylinders (mainly steel) , and on the order of 20+ leaks (all from fiber-wrapped plastic cylinders) (Richards et al., 1996). If we assume then that 10 to 50 full tanks of CNG have ruptured or leaked since 1976, and that the roughly 1 million NGVs in the world (Richards et al., 1996) travel 10,000 miles per year over the 20 years, we can calculate an emission rate of less than 10 micrograms of gas per mile of travel by NGVs[24]. It is reasonable to assume similar rates for compressed hydrogen storage, and

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[24] A similar calculation can be done for petroleum-fuel vehicles. There are on the order of 300,000 vehicle fires a year (U. S. Fire Administration, 1992), and on the order of 500,000 serious accidents per year (National Highway Traffic Safety Administration, 1994). Assuming then that at the very most there are 500,000 tank ruptures per year, the maximum emission rate is less than 0.01 g / mi of VOCs. This is considerably more than the rate for NGVs, but still small enough to ignore.

cryogenic fuel storage. We therefore ignore tank failure, for any reason, as a source of pollution.

Boil-off loss. The liquefied light gases, methane and hydrogen, must be kept at low temperatures: below 112° K for LNG, and below 20° K for LH_2. Although, the cryogenic storage vessels are well insulated (double-walled, with a vacuum between the walls), they naturally are not perfectly insulated, and thus, gradually the liquefied gas evaporates, or "boils off". If the vehicle is driven regularly, this boil-off gas is consumed as fuel. However, if the vehicle sits, the boil-off continues, and the vapor pressure in the tank builds until the maximum allowable pressure is reached, at which point the accumulated gas is vented to the atmosphere. An LH_2 tank can sit for a matter of days before it vents.

Ewald (1998) reports the evaporation rate for LH_2 tanks was 1%/day as of 1995. (This, presumably, is the rate *after* the tank starts venting, not an average rate calculated on the basis of some assumed frequency of use.) Advanced tanks might have an even lower loss rate: for example, Wetzel (1998) describes a recent design in which the elimination of the solenoid cryovalve and associated parts reduces residual heat leak into the inner vehicle tank by 20%. If once a year an LH_2 vehicle sits unused long enough to vent fuel, and then vents for, say, 5 days, the total loss would be, at most, 5% of one tank in a year (assuming that the 1%/day rate applies to a full tank). If a vehicle consumes at least 50 full tanks a year, the year-round average loss is less than 0.1%. I assume 0.1% for model year 1995, dropping by 3.5%/year in relative terms.

The length of time that an LNG tank can sit before it vents also depends on the heat loss and vent pressure of the tank, which in turn depend on the design of the tank. O'Brien and Siahpush (1998) tested a 70-gallon and a 17-gallon tank at Idaho National Engineering Laboratory, and found that the large tank vented after 7 days of sitting, but the small tank vented after only 2.5 days, on account of its higher surface-to-volume ratio. Upon venting, the large tank lost about 3% of its fuel per day, and the small tank 6.5%. However, O'Brien and Siahpush (1998) imply that it would not be difficult to increase the time-to-vent of the small tank. Powars et al. (1994) state that the time to vent is always at least 5 days, commonly 7 days, and sometimes 10 to 14 days. I assume a year-round average loss of 0.1%[25].

LPG is liquefied by virtue of being compressed, rather than cooled, which means that the storage tanks are designed to withstand any ambient vapor pressure, and hence do not vent.

Application in the model. With these assumptions, the total vehicular fuel loss rate in g/mi is calculated as:

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[25] I note that it is not implausible to assume that LNG tanks will end up venting as much as will LH_2 tanks, even though LH_2 is much colder, because it is might not be worthwhile to invest in better insulation for LNG tanks.

$$FLGM = FLR \times GMBTU / MIBTU \qquad \text{eq. 48}$$

where:

FLGM = the fuel loss rate, in grams/mile.
FLR = the fuel loss rate, in % of throughput (ordinary loss plus boil-off loss); the loss due to failure is ignored, and the loss from refueling is handled elsewhere.
MIBTU = the mile/BTU fuel economy of the vehicle (calculated as explained elsewhere).
GMBTU = the energy content of the fuel in grams/BTU (calculated on the basis of the HHV of the individual components of the gas, and a modification of the ideal gas law; see the discussion earlier in the text).

Again, these leakage and boil-off emissions are in addition to emissions from refueling and fuel distribution and storage, which are estimated separately.

The emissions have the composition of the fuel itself, which in the case of CNG is mainly, but not entirely methane (Table 5). Emissions from hydrogen vehicles are included, even though hydrogen itself is of no concern environmentally, because hydrogen made from natural gas may contain small amounts of CO, CO_2, and CH_4, which can affect climate.

There is a final methodological issue here. Should this vehicular fuel loss be counted as fuel that has to be made up by producing more fuel upstream, just as gas leakage from pipelines has to be made up in order to deliver a fixed amount of fuel to end users? The answer is “probably not”. If we understand our fuel economy measure, mi/10^6-BTU, to refer to BTUs of fuel delivered to the vehicle, then the loss of any fuel from the vehicle in principle is accounted for already in the mi/10^6-BTU. Also the mi/gal figure for gasoline vehicles, which serves as the basis of the estimate of mi/10^6-BTU for all vehicles, must already account for evaporative losses of fuel from vehicles, because the “gallon” in “mi/gal” is measured going into the vehicle.

There is, however, a complication. The fuel loss rate differs slightly across vehicle and fuel types, and in principle these differences should be accounted for, just as differences in thermal efficiency are accounted for. However, because the differences are with respect to an extremely small baseline (e.g., for a gasoline vehicle, evaporative losses are about of fuel consumption), they have been ignored.

Purging losses. The fuel cell stack in a fuel-cell vehicle may not consume all of the hydrogen supplied to it. Some or all of this unreacted hydrogen may be purged and vented to the atmosphere. On average such purging losses can be expected to be small, on the order of 1% of the hydrogen fuel on board the vehicle. I assume this rate here.

Note that these purging losses are in addition to the ordinary fuel-system leaks discussed above.

Emissions of refrigerant

The main source of CFCs and HFCs from highway vehicles is the air conditioning system. There are four ways that the refrigerant can be emitted. First, the refrigerant charge can be completely vented in the event of a collision that damages the air-conditioning system. Second, air-conditioning systems can malfunction or fail over time, resulting in partial or complete venting. Third, even though the EPA has enacted rules that require the recovery and recycling of O_3-depleting refrigerants (Walsh, 1993), there undoubtedly is some illegal scrappage in which the refrigerant simply is vented. Finally, a very small amount of refrigerant is released during the refrigerant reclamation or recharging process itself because the gas in a few inches of hose (between the hose valves and the ends of the connectors) is released when the hoses are disconnected.

Thus, some vehicles will never completely vent the refrigerant charge to the atmosphere, while others, with malfunctioning air conditioner systems or that are involved in collisions, may completely vent the refrigerant charge more than once. All vehicles will vent at least trace quantities of refrigerant at various times when their air conditioning systems are recharged, or permanently decommissioned for vehicle scrappage. However with modern reclamation systems these emissions are negligible.

DeLuchi (1993, Appendix Q) estimated that three 2.6-lb charges of CFC-12 were emitted over the 108,000-mile life of an LDV. This emission rate of 31.5 mg / mi, multiplied by a GWP of 7,300 (in Table 8 of DeLuchi [1991]), resulted in CO_2-equivalent emissions of 230 g / mi (Table B.2 of DeLuchi [1993]), a sizable fraction of total fuelcycle emissions from a gasoline LDV. However, Ford (Wallington, 1996), the EIA (*Emissions of Greenhouse Gases in the United*, 1997), and Bates and Harnishc (2001) indicate that since 1991, the typical vehicle has had a 2.0-lb charge of refrigerant, and that it is more likely that the equivalent of only one charge is lost over the life of the vehicle (which, as discussed elsewhere, we now assume to be more than 108,000 miles), resulting in an emission rate of approximately 7 mg / mi. If the refrigerant is HFC-134a, with a CEF of 2,000, then the result is 14g / mi CO_2-equivalent emissions. This is over an order of magnitude lower than the originally estimated CO_2-equivalent emissions, albeit still not entirely trivial.

Older vehicles with CFC-12 probably emit more, on account of the larger refrigerant reservoirs of vehicles made before efforts to phase-out CFCs commenced

Accounting for the lower efficiency of HFC-134a compared with CFC-12. A more complete lifecycle analysis of the effect of substituting HFC-134a for CFC-12 accounts not only for the mass emissions of each refrigerant and the CEF of the emissions, but also for the energy efficiency of the refrigerant. The efficiency is defined as $A \cdot C / E$, where A is the amount of air cooled, C is the magnitude of cooling, and E is the quantity of energy consumed. Although HFC-134a is thermodynamically similar to

CFC-12, it is not miscible with the mineral oils currently in use and is somewhat less efficient with most substitute lubricants (Fischer and McFarland, 1992a) . Fischer et al. (1992b) account for this lower efficiency, along with the lower GWP for HFC-134a, and the mass emission rate, in their estimate of the "total equivalent warming impact" (TEWI) of refrigerants. They find that the TEWI value of HFC-134a would be only 16% of that of CFC-12 (17,000 lbs of CO_2 equivalent emissions versus 108,000 lbs), based on 500-year GWP values (Fischer, et al., 1992b)[26] .

However, faced with the lower theoretical efficiency of HFC-134a, Ford Motor Company redesigned the heat exchanger units and other components of automobile air conditioners (Wallington, 1996) . As a result of these improvements in system efficiency (about 5% from improved heat exchanger design alone), the amount of HFC-134a used now is the same as the amount of CFC-12 used immediately before the transition (about 2 lbs) , and the energy-use difference between new HFC-134a systems and the previous late-model CFC-12 systems is not detectable (Wallington, 1996) . Therefore, I assume that there is no change in motor-vehicle fuel consumption on account of the switch to HFC-134a, and do not have any such adjustment factor in the LEM

Alternative-fuel vehicles. This refrigerant emission rate is, in theory, a function of the type of coolant used, charging and maintenance practices, and the life of the vehicle, but presumably not a function of the type of fuel or engine used in an LDV. Thus, it seems reasonable to assume that all AFVs using the same type of cooling system will be responsible for the same amount of refrigerant-caused global warming.

These refrigerant emissions now are included as part of the full lifecycle emissions from heavy-duty trucks used to deliver feedstocks, end-use fuels, chemicals, fertilizers, and so on. The full lifecycle includes fuel production and use, materials manufacture and assembly, and refrigerants.

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[26] This analysis used direct rather than net GWP values, so it is reasonable to assume that the 16% figure could be revised upward to about 20%, based on the net GWP values reported by the IPCC (1996a).

PETROLEUM REFINING

Refinery energy use: meaning of BTU/BTU measure

In the previous version of the model, the measure of refinery energy use, BTU-refinery-energy / BTU-gasoline, was with respect to BTUs of complete gasoline product. This includes anything produced outside of the refinery, such as MTBE, not to just the refinery-produced hydrocarbon portion of the gasoline. The meaning of the measure has now been changed to BTUs-refinery-energy / BTU-gasoline-HC, where the denominator includes only the refinery-produced hydrocarbon-portion of the gasoline and not , for example, the energy value of any MTBE or ethanol produced outside of the gasoline. Conceptually, I now assume that in effect any methanol or ethanol (as such, or in MTBE or ETBE) is added to the gasoline outside of the refinery gates.

Presently, I do *not* have a different BTU / BTU measure for different "base" reformulated gasolines. I do not distinguish the hydrocarbon "base" for RFG with ethanol from the hydrocarbon base for RFG with MTBE. Although it is possible that BTU / BTU energy intensity of the hydrocarbon base depends appreciably on the final overall composition of the RFG, I could not find any estimates of this dependency[27].

BTUs of refinery energy per BTU of each major refinery product

The LEM takes as inputs estimates of the refinery energy intensity of producing conventional gasoline (CG), reformulated gasoline (RFG), conventional diesel fuel (CD), ultra-low-sulfur diesel fuel (ULSD), residual fuel oil (RFO), and liquefied petroleum gases (LPG). The estimates are in units of BTUs of total refinery energy (including steam, at 1400 BTUs-NG / lb-steam [Kadam et al., 1999], and electricity, at 3413 BTU / kWh) per BTU of product.

The BTU / BTU estimates in DeLuchi (1991, 1993) are:

CG	RFG	CD	LSD	RFO	LPG
0.182	0.145	0.058	0.065	0.045	0.054

where LSD is low-sulfur diesel (specified in DeLuchi, 1991, 1993), not ultra-low sulfur diesel (specified in the present LEM). Because petroleum refining usually is the second-largest source of emissions in the petroleum lifecycle (after end use), it is important to have good estimates of the refinery energy intensity. To check the original energy intensity assumptions shown above, the total U. S. refinery fuel use was calculated based on the energy intensities shown above, and the results were compared to the actual total U. S. refinery fuel use, as reported by the EIA's *PSA*:

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[27] Hadder (1997) used the Oak Ridge National Laboratory Refinery Yield Model (ORNL-RYM), an enhanced personal-computer version of the Refinery Evaluation Modeling System used by DOE, to estimate the impacts of ethanol use on refinery inputs and outputs, but he does not model the difference in energy intensity as a function of the type of oxygenate used.

$$CTRF_Y = \sum_p RP_{p,Y} \cdot HHV_p \cdot RFI_p$$

$$ATRF_Y = \sum_p RF_{p,Y} \cdot HHV_p \qquad \text{eq. 49}$$

where:

subscript Y = year of analysis (1990, 1996, and 1998).
subscript p = fuel or product type p (coke, residual fuel oil, gasoline, distillate fuel, etc.).
$CTRF_y$ = calculated total refinery fuel in year Y (BTUs).
$RP_{p.y}$ = refinery production of P in year Y (usually 10^3 bbl; EIA, *PSA).*
HHV_p = the higher heating value of product P (usually BTUs/10^3-bbl; EIA, *PSA*, and other sources).
RFI_p = the refinery fuel intensity of producing P (BTUs-refinery-fuel/BTU-P; as above).
$ATRF_y$ = actual total refinery fuel used in year Y (BTUs).
$RF_{p,y}$ = refinery use of fuel P in year Y (usually 10^3 bbl; EIA, *PSA).*

The results, for the years 1990 (with no LSD, and no RFG), 1996, and 1998 are:

	1990	**1996**	**1998**
Calculated refinery fuel (quads)	2.89	3.26	3.40
Actual refinery fuel (quads)	2.89	3.04	3.12
Calculated/actual	1.00	1.07	1.09

The energy intensities assumed by DeLuchi (1991, 1993) are consistent with the actual fuel usage reported by refineries in 1990, but overestimate fuel usage in 1996 and 1998. It appears that the error increases with time.

I suspect that there are two reasons for the discrepancy between calculated and actual fuel usage in 1996 and 1998. First, I believe that over the past decade refineries have become more energy efficient. Second, I believe that my original estimate of the energy intensity of producing reformulated gasoline is too high (I assumed that there was no reformulated gasoline in 1990).

The trend in the overall refinery energy intensity (i.e., the energy intensity of producing all products) supports the first explanation. Using EIA *PSA* data on total refinery production, and total use of process fuels, I calculate the following overall refinery energy intensity (BTUs-refinery-fuel/BTUs-all-products):

1990	1991	1994	1996	1998
0.096	0.096	0.095	0.094	0.093

Note that, in spite of increasing output of reformulated gasoline, which has the highest energy intensity of any product, the overall refinery energy intensity has declined slightly. This implies that refineries have been becoming more energy efficient[28].

Evidence for the second proposition -- that the energy intensity of producing reformulated gasoline was over-estimated -- is thinner. In Table H.6 of DeLuchi (1993), refineries consumed 0.145 BTUs of process energy to produce 1.0 BTU of conventional gasoline. That estimate was based mainly on the following estimates (BTU-refinery energy / BTU-product; see DeLuchi [1993] for details):

Estimate based on:	conventional gas	distillate	residual fuel
Lawrence et al. (1980)	0.146	0.072	0.068
Lawrence et al. (1980)	0.162	0.039	0.036
Haynes (1976)	0.148	0.077	0.052
Mertes and Hurwicz (1980)	0.156	0.043	0.034
White et al. (1982)	0.145	0.064	n.e.

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[28] The EIA's *AEO* projections imply increasing overall energy intensity from 1998 to 2002, constant energy intensity from 2002 to 2008, and decreasing energy intensity thereafter.

Recently, Stork and Singh (1995) reported that a linear programming model of a complex refinery estimated that summer conventional gasoline requires 0.155 BTUs/BTU, and winter conventional gasoline 0.141 BTUs/BTU. The simple average, 0.148, is very close to the 0.145 value assumed here. However, Stork and Singh (1995) estimate that reformulated gasoline requires essentially the same amount of energy to produce as does conventional gasoline. It is not clear why this should be so.

Using an input/output model of refinery processes, GM et al. (2002c) estimate that a European refinery requires 0.055 to 0.120 BTUs-process-energy (including electricity) per BTU-ULSD, and 0.102 to 0.208 BTUs-process-energy/BTU-gasoline (with ultra-low sulfur), depending on the sulfur content of the crude oil input, and whether a partial oxidation plant is included to produce hydrogen from the visbreaker residue.

Finally, there is the question of the energy intensity of producing ULSD versus conventional diesel. Analyses by GM et al. (2002c) indicate that the production of ULSD (10 ppm S) requires 0.02 to 0.03 BTUs-process-energy (including electricity and hydrogen) per BTU of diesel produced, depending on whether the distillates are "straight run" or from crackers (which increase the hydrogen requirement), and about 0.02 BTUs/BTU if the hydrogen energy requirement is ignored. Fredriksson et al. (2000) use linear programming models to estimate the effect of sulfur reductions on refinery costs and emissions in the European Union (EU). They plot incremental increases in CO_2 from EU refineries against the log of the suflur content of diesel fuel, and show that reducing the sulfur content of diesel fuel from 350 ppm to 10 ppm would increase CO_2 emissions from European-Union (EU) refineries by 4.8 Mt/year. Extending their plot to 5000 ppm (they stopped at 350 ppm), I estimate that going from 5000 ppm to 10 ppm diesel fuel would increase CO_2 emissions by about 10 Mt/year. Elsewhere, Fredrikkson et al. (2000) report that total CO_2 emissions from all sources in EU refineries (producing 350 ppm diesel fuel) are about 100 Mt/year, and that diesel fuel is about 25% of the output of EU refineries. All of this suggests the production of diesel fuel with 5000 ppm would be responsible for a total of 25-30 Mt-CO_2/year, and that the production of 10 ppm S would increase CO_2 emissions by 10 Mt-CO_2/year, or 35-40%.

In consideration of the foregoing, the original assumptions are modified and qualified as follows:

- RFG requires 0.170 BTU-process/BTU-RFG;
- ultra-low sulfur diesel (USLD) with 5 ppm S by weight requires 40% more energy to produce than does conventional diesel with 5000 ppm S by weight;
- the original and modified assumptions regarding energy intensity apply to the year 1990; thereafter, all of the energy intensity values decrease by 0.25%/year, in relative terms.

With the new assumptions, calculated values match actual total refinery consumption:

	1990	**1996**	**1998**
Calculated refinery fuel (quads)	2.89	3.04	3.12
Actual refinery fuel (quads)	2.89	3.04	3.12
Calculated / actual	1.00	1.00	1.00

Refinery energy use in other countries

As discussed at the beginning of this report and in Appendix B, the LEM represents trade in crude oil and petroleum products. Specifically, for any designated consuming country the LEM estimates the sources of petroleum – the particular petroluem-producing countries that supply the target country -- and then calculates lifecycle emissions based on the energy-use parameters for these major petroleum producing and refining countries.

The LEM has two parameters that vary from one major refining center to another: the BTU / BTU energy intensity, by type of product, and the mix of fuels used to generate electricity used by refineries. Generally, I assume that refining process technology is the same everywhere, so that it takes the same amount of refinery energy to make a particular product in, say Europe, as it does in the United States. However, I do adjust for significant differences in the quality of input crude oil: if a country tends to process especially heavy crude oil, then the energy intensity of refining likely will be higher. I assume that this is the case for Canada (about 30% higher energy requirements than the U. S.), Venezuela (about 15% higher), and Carribean heavy crude (50% higher). I also assume that energy requirements are 10% higher in the Former Soviet Union and in less-developed countries on account of the relative inefficiency of the industrial sector in these places.

My assumptions regarding the mix of fuels used to generate electricity used by refineries are as follows:

Petroleum refiner	**generation mix by type**					**notes**
	coal	*oil*	*gas*	*nuke*	*hydro*	
U. S.	31%	5%	33%	23%	7%	analysis of actual generation mix for refineries (DeLuchi, 1993)
Canada	31%	0%	6%	21%	41%	regional analysis of Canadian power mix (see App. B)
N. Europe	25%	3%	62%	4%	0%	IEA (2002c) data for the Netherlands year 2000
S. Europe	6%	16%	46%	0%	16%	IEA data for Italy (see App. B; calculated mix in target year)
OPEC			0%			
Venezuela	0%	10%	16%	0%	74%	IEA (2002c) data for Venezuela year

						2000
N. Africa	0%	3%	95%	0%	0%	IEA (2002c) data for Algernai year 2000
Nigeria	0%	6%	57%	0%	36%	IEA (2002c) data for Nigeria year 2000
Indonesia	31%	22%	36%	0%	10%	IEA (2002c) data for Indonesia year 2000
Persian Gulf	0%	70%	30%	0%	0%	IEA (2002c) data for Persian Gulf countries, year 2000
Caribbean	6%	51%	40%	0%	2%	IEA (2002c) data for countries of the Carribean, year 2000
Other Asia	48%	4%	9%	37%	2%	IEA data for Korea (see App. B; calculated mix in target year)
Other	38%	10%	14%	18%	19%	my estimates
FSU	25%	2%	39%	17%	17%	IEA data for Russia (see App. B; calculated mix in target year)
generic developed	50%	5%	15%	20%	10%	my estimates
generic LDC	62%	5%	15%	0%	15%	my estimates

BTUs of refinery energy per BTU of diesel fuel

In the real world and in the LEM, the energy intensity of producing diesel fuel depends on the sulfur content of the finished fuel. In the LEM, this dependency is handled by estimating the energy intensity of producing reference diesel fuels, and then relating the sulfur content of the user-specified diesel fuel to the sulfur content of the reference fuels. Specifically, the LEM first estimates the energy intensity of producing a conventional diesel (CD) fuel with a relatively high sulfur level (5000 ppm S) and the energy intensity of producing an ultra-low-sulfur-diesel (ULSD) fuel with almost no sulfur (5 ppm S). Then, the energy intensity of producing the actual diesel fuel specified in the model, with a sulfur content corresponding to the target year and target country specified by the user (as discussed above), is calculated by multiplying the CD and ULSD energy intensities by weighting factors. These weighting factors are estimated on the basis of the sulfur content of the actual fuel relative to the sulfur content of the reference CD and ULSD. In effect, the model interpolates between the CD and ULSD energy intensities according to where the sulfur content of the actual fuel lies with respect to the sulfur content of the reference fuels. (See the section on the sulfur content of diesel fuel for further discussion.)

Projections of the mix of refinery fuels

In the model, the user enters the refinery energy intensity of producing each major kind of petroleum product, and the breakdown of that refinery energy by type of fuel (refinery gas, natural gas, petroleum coke, electricity, etc.). Previously, the user of the model input one fuel breakdown for all years, on the basis of historical use (e.g., Table H.4 of DeLuchi [1993]) and considerations of future trends. Now, the model has a detailed projections of refinery fuel use, from 1990 to 2050. The user specifies the year of analysis, and the model selects the appropriate data series and calculates the breakdown of refinery fuel use. Fuel-use data for the years 1990-1999 are from annual issues of the *PSA*. The projections of the amount of each kind of fuel used by refineries are from the EIA's *AEO*. Projections of the energy content of purchased steam have been added assuming 1400 BTUs-NG/lb-steam[Kadam et al., 1999] and hydrogen.

The model also projects, just for reference, the overall energy intensity of refinery output, expressed as total BTUs of refinery energy consumed per BTU of product output. For each year, this is calculated as follows:

$$REI = \frac{\sum_{p} RF_p}{\left(C_I + NGL_I + O_I + VG\right) \cdot \frac{E_{PS}}{B_{PS}}} \qquad \text{eq. 50}$$

where:

subscript p = types of refinery fuel (fuel oil, natural gas, refinery gas, etc. -- see above).

REI = the overall refinery energy-use intensity (BTU-process-fuel/BTU-product-output.

RF_p = refinery fuel type P (BTUs).

C_I = input of crude oil to refineries (bbls) (projected as "total crude oil supply" by the EIA).

NGL_I = input of natural-gas liquids to refineries (bbls) (unpublished projections available from the Energy Information Administration).

O_I = input of other liquids and feedstocks to refineries (bbls) (my extrapolation of historical data).

VG = volumetric gain of refineries (bbls; the difference between the volume of input, which we know, and the volume of output, which we are interested in) (projections by the EIA).

E_{PS} = the total energy content of petroleum products supplied (BTUs) (projections by the EIA).

B_{PS} = the total volume of petroleum products supplied (bbls) (projections by the EIA).

Again, this quantity is not used in the model; it is provided just for information.

Allocation of refinery energy to specific products

There is a typesetting error in Table H.5, page H-20 of DeLuchi (1993). On the "Desulfurization" line, the values 0.454, 0.302, and 0.070 should be shifted over to the right by one column, so that the 0.454 is under "Gasoline," the 0.302 is under "Dist.", and the 0.070 is under "Residual". There should be a blank (zero) under "Haynes". This is typesetting error only; the values were entered correctly in the model.

Sale or transfer of electricity

According to the EIA's *Manufacturing Energy Consumption Survey 1991* (1994), petroleum refineries on average sell or transfer out about 10% of the amount of the electricity that they purchase. This sold or transferred power should be deducted from electricity purchases, to arrive at a "net purchase" figure for calculating greenhouse gas emissions due to electricity use. Deluchi (1991; Table 4) did not account for this. Now, the EIA's projections of refinery electricity use is multiplied by 0.90.

Crude used as fuel gas or petroleum coke in refineries

The previous version of the model did not account for emissions from the production and transport of the portion of the crude oil that ends up being used as fuel gas or petroleum-coke fuel in refineries. This has been corrected. Refinery gas and petroleum coke have been added in the appropriate places (Tables 3, and 5).

Emissions of pollutants from refinery process areas

The estimation of g/BTU emissions from process areas, such as catalytic cracking units has been completely overhauled. Now, emissions are estimated separately for each pollutant (NMOCs, CH_4, CO, N_2O, NO_X, SO_2, PM, and CO_2) and each major type of product (gasoline, distillate fuel, residual fuel, and LPG). The basic input data are controlled and uncontrolled emissions of each pollutant from each process area in a refinery. Specific inputs are the fraction of throughput that is controlled, and the amount of throughput of each type of product in each process area. From these data, the model calculates emissions of each pollutant per unit output of each type of product. Formally:

$$GBTU_{P,F,T} = K_F \cdot \sum_{A} CEM_{P,A} \cdot FC_{A,T} \cdot FA_{F,A} + UEM_{P,A} \cdot \left(1 - FC_{A,T}\right) \cdot FA_{F,A}$$

eq. 51

$$K_F = \frac{453.6}{1000 \cdot EBBL_F}$$

where:

subscript P = pollutant types (NMOCs, CH_4, CO, N_2O, NO_X, SO_2, PM; CO_2 is estimated slightly differently, as discussed below).

subscript F = refinery product categories (conventional gasoline, reformulated gasoline, conventional diesel fuel, low-sulfur diesel fuel, residual fuel, and LPG).

subscript A = refinery process areas:

process area:	*applies to:*	*pollutants:*
Vacuum distillation	all products	NMOC, N_2O, CH_4, CO_2
Blowdown systems	all products	CO, NMOC, NO_X, N_2O, CH_4, CO_2
Fluid-bed catalytic cracking units (FCCUs)	distillate, gasoline	PM, CO, NMOC, NO_X, N_2O, CH_4, CO_2
Moving-bed catalytic cracking units (MCCUs)	distillate, gasoline	PM, CO, NMOC, NO_X, N_2O, CH_4, CO_2
Thermal cracking (coking)	distillate, gasoline	PM, NMOC, N_2O, CH_4
Oil / water separators	all products	NMOC
Cooling towers	all products	NMOC
Valves, seals, flanges, drains	dist., gasoline, LPG	NMOC

$GBTU_{P,F,T}$ = emissions of pollutant P emitted per energy unit of product type F produced by refineries, in year T (g / 10^6 BTU) (results shown in Table 14).

$CEM_{P,A}$ = controlled emissions of pollutant P from process area A (lbs / 10^3-bbl-throughput or feed) (discussed below).

$FC_{A,T}$ = the fraction of process areas A with controls, in year T (discussed below).

$FA_{F,A}$ = throughput of product type F in process area A (bbls-F-throughput-area-A / bbl-F-output-from-refinery) (discussed below).

$UEM_{P,A}$ = uncontrolled emissions of pollutant P from process area A (lbs / 10^3-bbl-throughput or feed) (discussed below).

K_F = factor to convert from lbs / 10^3-bbl to g / 10^6-BTU, for product type F.

$EBBL_F$ = average energy content of a barrel of refinery output of product type F (10^6 BTU / bbl) (from DeLuchi [1993] and revisions there to this report.

Controlled and uncontrolled emissions from refinery process areas (CEM and UEM). EPA’s (1995) AP-42 reports controlled and uncontrolled emissions of PM, CO, VOCs, SO_X, and NO_X from vacuum distillation, blowdown systems, FCCUs, MCCUs, and fluid coking units (thermal cracking). They also report controlled and uncontrolled "fugitive" NMOC emissions from oil / water separators and cooling towers, and uncontrolled fugitive NMOC emissions from valves, flanges, seals, and drains. I use EPA factors for controlled and uncontrolled emissions, with the following exceptions:

- I assume 17 lbs-PM / 10^3-bbl-fresh-feed rather than the EPA's 45 lbs-PM / 10^3-bbl-fresh-feed, from FCCUs with controls, because New Source Performance Review Standards adopted before 1980 limit PM emissions from FCCUs to 19 lbs-PM / 10^3-bbl-fresh-feed. (I assume that refineries meet the standard with some margin of safety.)
- NO_X emissions from FCCUs are assumed to be controlled from 71 lbs / 10^3-bbl to 20 lbs / 10^3-bbl fresh feed (the EPA reports that NO_X emissions from controlled FCCUs are the same as emissions from uncontrolled FCCUs, presumably because the controls in FCCUs are meant for PM and CO).
- The EPA does not report controlled fugitive NMOC emissions from valves, seals, flanges, and drains; I assume that controlled emissions are 20% of uncontrolled emissions (DeLuchi et al., 1992).
- Rather than use the EPA's reported SO_X emission factors, I apportion SO_X emissions to products on the basis of the difference between the sulfur content of the product and the sulfur content of the crude oil, accounting for the efficiency of sulfur control:

$$LBBBL_{SOx,F,T} = \frac{GGAL_F \cdot \left(SF_{oil,T} - SF_F\right)}{453.6} \cdot 42 \cdot 1000 \cdot \frac{MW_{SO2}}{MW_S} \cdot \left(ER_{SOx,T}\right) \qquad \text{eq. 52}$$

where:

$LBBBL_{SOx,F,T}$ = refinery emissions of SO_X attributable to product F in year T (lbs-SO_X / 10^3-bbl-F).

$GGAL_F$ = the energy density of product F (g / gal; see DeLuchi [1993] and revisions thereto in this report).

$SF_{oil,T}$ = the sulfur weight fraction of crude oil in year T (elsewhere in this report).

SF_F = the sulfur weight fraction of product F (elsewhere in this report).

453.6 = g / lb

42 = gal / bbl

MW_{SO2} = the molecular mass of SO_2 (64 g / mol).

MW_S = the molar mass of S (32 g / mol).

$ER_{SOx,T}$ = the emission reduction factor, due to emission controls, for SO_X emissions in year T (the ratio of controlled or post-control emissions to uncontrolled emissions); estimated using Equation 6 with the following parameter values:

V_U = the upper limit = 0.20
V_L = the lower limit = 0.01
V_{T_B} = the base-year value = 0.02 (i.e., 98% control efficiency)
k = the shape or steepness factor = 0.08
T_B = the base year = 1990

These assumptions give results that are consistent with estimates and projections of sulfur control and sulfur emissions from refineries (EPA, AP-42, 1995; EPA, *National Air Pollutant Emission Trends 1900-1996*, 1997; DeLuchi et al., 1992).

Note that in the case of diesel fuel, we perform this SOx-apportionment calculation for a reference conventional diesel fuel (5000 ppm S) and a reference ultra-low-sulfur-diesel fuel (5 ppm S). Then, as explained in the section on refinery BTU/BTU energy intensity and the section on the sulfur content of diesel fuel, we estimate the SOx emissions attributable to producing the actual, user-specified diesel fuel on the basis of the sulfur content of the user-specified diesel fuel relative to the sulfur content of the reference fuels.

The EPA's AP-42 does not report CH_4 or N_2O emissions from refinery process areas. However, the EPA's *Inventory of U. S. Greenhouse Gas Emissions and Sinks: 1990-1998* (2000a) has a comprehensive set of CH_4 emission factors for refinery process and combustion areas. Virtually all of the refinery emissions come from system blowdowns and asphalt blowing. The emission factor for system blowdowns is 5.8 lb/1000-bbl feed. I adopt this here. I also adopt the EPA's (2000a) emission factors for other process areas. The EPA's (2000a) estimates result in about 1 g-CH_4/10^6-BTU production, which is consistent with the range of 0.24 to 2.4 g/10^6 BTU cited by DeLuchi (1993; Table A.1), and the emission factor of 2.4 g/10^6 BTU cited by the EIA's *Emissions of Greenhouse Gases in the United States 1987-1994* (1995).

Low-temperature combustion processes, such as in fluidized-bed combustion, can produce significant amounts of N_2O. Fluidized-bed combustion is used at one stage in the production of petroleum products: when the fluidized-bed catalytic cracker, which breaks the large hydrocarbon molecules of crude oil into the smaller molecules of gasoline or diesel fuel, becomes coated with coke residue from the crude oil, the coke is burned off the catalysts by fluidized bed combustion (called in this case "regeneration") (Cooper and Emanuelsson, 1992). Hence, this step in the refining process may produce non-trivial amounts of N_2O (Lyon, et al., 1989) . The one test of which we are aware measured 3-26 ppm N_2O and about 400 ppm NO from a fluidized-bed catalytic cracker with a zeolite catalyst, in a modern Swedish refinery (Cooper and Emanuelsson, 1992) . This concentration is lower than the N_2O concentration measured in other fluidized-bed combustors (See Appendix F to this report), perhaps owing to the type of catalyst used. According to Cooper and Emauelsson (1992), this emission rate is equivalent to 0.6 to 5.0 grams N_2O per barrel of oil. I assume 11 lbs-N_2O/10^3-

bbl-fresh feed from FCCUs, MCCUs, and fluid cokers, and an order of magnitude lower emissions from vacuum distillation and blowdown systems. The contribution of 11 lbs-N_2O / 10^3-bbl-fresh feed (5 g / bbl) is less than 1% of fuel cycle CO_2-equivalent emissions.

Technically, there should be a process area called "steam reforming of natural gas to produce hydrogen," to distinguish this use of natural gas, as a feedstock to make hydrogen, from the use of natural gas as a boiler fuel (Kadam et al., 1999). However, the process area emissions from reforming are similar to those from combustion, and so for simplicity I assume that all natural gas input to refineries is used in boilers.

The use of controls in process areas. Eq. 6 was used to estimate the parameter FA (the fraction of process areas with controls) in Eq. 51 above. I pick the values of the parameters in Eq. 6 so that the resulting estimates are consistent with estimates and projections of emissions from refineries by EPA (*National Air Pollutant Emission Trends 1900-1996, 1997*):

V_U = the upper limit on FA = 0.99
V_L = the lower limit on FA = 0.30

Process area	T_B	V_{T_B}	k
Vacuum distillation	1986	0.750	0.12
Blow down systems	1986	0.750	0.12
Fluid bed catalytic cracking	2006	0.985	0.12
Moving bed catalytic cracking	2006	0.985	0.12
Thermal cracking (coking)	2006	0.985	0.12
Oil / water separators	1986	0.600	0.16
Cooling towers	1986	0.850	0.10
Valves, seals, flanges, drains	1986	0.850	0.10

The EPA (*National Air Pollutant Emission Trends 1900-1996, 1997*) estimates process-area emissions of CO, NMOCs, NO_X, SO_X and PM_{10} from "petroleum refineries and related industries", from 1970 to 1996, and projects emissions from 1999 to 2010. (These estimates do not include emissions from fuel combustion in refinery boilers.) The projected emissions are equal to estimated emissions in 1995 multiplied by the ratio of projected earnings to estimated earnings in 1995, with adjustments for changes in process efficiency, and emission controls. Generally, the EPA assumed changes in control of ozone precursors, NO_X and NMOCs, but no change in CO, SO_X, or PM_{10} controls [EPA, *National Air Pollutant Emission Trends Procedures Document*, 1998].) In the EPA projections, total refinery emissions of all pollutants except NMOCs decline until the mid 1990s, and rise thereafter; emissions of NMOCs decline uniformly through 2010. I scaled my and the EPA estimates and projections to 1996 refinery output levels for comparison.

Product throughput through process areas. On the basis of my understanding of the use of each process area (see DeLuchi et al., 1992; Hadder, 1997), I estimate the following throughput for each type of product, in bbl of product to each process area, per bbl of product output:

Process area	*CFG*	*RFG*	*diesel*	*ULSD*	*resid.*	*LPG*
Vacuum distillation	0.30	0.30	0.45	0.45	1.00	0.00
Blowdown systems	1.00	1.00	1.00	1.00	1.00	1.00
FCCUs	0.45	0.52	0.30	0.30	0.00	0.00
MCCUs	0.02	0.02	0.02	0.02	0.00	0.00
Thermal cracking (coking)	0.02	0.02	0.02	0.02	0.00	0.00
Oil / water separators	1.00	1.00	1.00	1.00	1.00	1.00
Cooling towers	1.00	1.00	1.00	1.00	1.00	1.00
Valves, seals, flanges, drains	0.80	0.92	0.40	0.40	0.00	1.00

Results. FCCUs, which use heat, pressure and catalysts, convert heavy oils into lighter products, such as gasoline and distillate blending components. FCCUs account for most refinery emissions from process areas: about 85% of the total refinery emissions of PM_{10}, 65% of the SO_X emissions, and 95% of the CO emissions. FCCUs and controls on blowdown systems account for most of process-area NO_X emissions, and a variety of sources emit NMOCs (EPA, *National Air Pollutant Emission Trends 1900-1996, 1997*; DeLuchi et al., 1992).

CO_2 emissions from the control of CO and NMOC emissions from process units

Some refinery units, most notably fluid catalytic cracking units, produce large amounts of CO and NMOCs. Most of the CO and NMOC emission is controlled by burning the CO or NMOC to CO_2. In this section, CO_2 emissions from the control of CO and NMOCs at petroleum refineries in the U. S are estimated.

CO_2 emission from emission control is based on the difference between uncontrolled and controlled carbon emissions, for those process areas where the control is oxidation (e.g., flaring) rather than emission prevention. Thus, one first must identify which kinds of controls result in CO2 emissions. As just noted, not all do -- for example, the control of "fugitive" NMOC emissions, such as leaks from valves, generally involves reducing leakage rate. Such emission prevention does not result in CO_2 emissions. However, CO_2 is produced by the boilers used to control emissions from FCCUs, and the incinerators and flares used to control emissions from vacuum distillation and blowdown systems.

Formally:

$$CO2_{A,T} = \left(UEM_{CO,A} - CEM_{CO,A}\right)\cdot FC_{A,T}\cdot CC_{CO}\cdot 3.6641 + \left(UEM_{NMOC,A} - CEM_{NMOC,A}\right)\cdot FC_{A,T}\cdot CC_{NMOC}\cdot 3.6641 \qquad \text{eq. 53}$$

where:

UEM, CEM, and FC are as defined above
$CO2_{A,T}$ = CO_2 emissions from the control of CO and NMOC emissions from process area A in year T (lb / 10^3-bbl) (relevant process areas and emissions are: NMOCs from vacuum distillation and blowdown systems, and NMOCs and CO from FCCUs and MCCUs).
CC_{CO} = the carbon weight fraction of CO (0.43).
3.6641 = the ratio of the molecular mass of CO_2 to the molar mass of carbon.
CC_{NMOC} = the carbon weight fraction of NMOCs (assume 0.85).

Feedstock carbon lost in emissions: the effect on crude oil throughput

Total carbon: As discussed above, refinery process areas, such as catalytic crackers, emit CH_4, CO, NMOCs, and CO_2. I assume that carbon in these process-area emissions comes from the petroleum feed. This means that emissions of carbon from refinery process areas constitute lost crude-oil feedstock. The more crude oil lost, the greater the throughput of crude oil required to produce a given amount of gasoline, diesel fuel, etc. The greater the throughput, the greater the use of energy to recovery and transport crude oil, and hence the greater the emissions of greenhouse gases.

The model now accounts for this effect of lost crude oil, by incorporating emissions from the recovery and transportation of the amount of crude that ends up being lost in carbon emissions from process areas. Formally, the grams of CO_2-equivalent GHG emissions due to the recovery and transport of crude oil lost at the refinery, per 10^6 BTU of product out of the refinery is estimated as follows:

$$\frac{GHG_{RLO}}{BTU_F} = \frac{GHG_{RLO}}{BTU_{LO}} \cdot \frac{BTU_{LO}}{BTU_F}$$

$$\frac{GHG_{RLO}}{BTU_{LO}} = \frac{GHG_{RO}}{BTU_O}$$

$$\frac{BTU_{LO}}{BTU_F} = \frac{gC_{LO}}{BTU_F} \cdot \frac{BTU_{LO}}{gC_{LO}}$$

$$\frac{BTU_{LO}}{gC_{LO}} = \frac{BTU_O}{gC_O}$$

$Hence$:

$$\frac{GHG_{RLO}}{BTU_F} = \frac{GHG_{RO}}{BTU_O} \cdot \frac{gC_{LO}}{BTU_F} \cdot \frac{BTU_O}{gC_O}$$

And:

$$\frac{GHG_{RO}}{BTU_O} = \frac{GHG_{RR}}{BTU_R} \cdot \frac{D_R}{D_O}$$

$$\frac{gC_{LO}}{BTU_F} = \frac{gCH4_{LO}}{BTU_F} \cdot 0.75 + \frac{gNMOC_{LO}}{BTU_F} \cdot 0.90 + \frac{gCO_{LO}}{BTU_F} \cdot 0.428 + \frac{gCO2_{LO}}{BTU_F} \cdot 0.273 \qquad \text{eq. 54}$$

where:

GHG_{RLO}/BTU_F = grams of CO_2-equivalent GHG emissions from the recovery and transport of crude oil lost at the refinery, per BTU of product F output from the refinery.

GHG_{RLO}/BTU_{LO} = grams of CO_2-equivalent GHG emissions from the recovery and transport of crude oil lost at the refinery, per BTU of oil lost at the refinery.

BTU_{LO}/BTU_F = BTUs of crude oil lost at the refinery per BTU of product F output from the refinery.

GHG_{RO}/BTU_O = grams of CO_2-equivalent GHG emissions from the recovery and transport of any crude oil, per BTU of any crude oil.

gC_{LO}/BTU_F = grams of carbon in crude oil lost at the refinery per BTU of product F output from the refinery.

BTU_{LO}/gC_{LO} = BTUs of crude oil lost at the refinery per gram of carbon in crude oil lost at the refinery.

BTU_O/gC_O = BTUs of crude oil per gram of carbon in crude oil (about 49; the amount varies slightly over the projection period 1994 to 2010).

GHG_{RR} / BTU_R = grams of CO_2-equivalent GHG emissions from the recovery and transport of residual fuel oil, per BTU of residual fuel oil (calculated by the model, which does not calculate emissions for crude oil itself).

D_R / D_O = the ratio of the density of residual fuel oil to the ratio of the density of crude oil (this adjustment is necessary because recovery and transport emissions are assumed to be proportional to the mass of the material).

$gCH4_{LO} / BTU_F$ = grams of CH_4 emissions from process areas in the refinery, per BTU of product F output from the refinery (see DeLuchi [1993], and revisions in this report).

0.75 = weight fraction of carbon in methane.

$gNMOC_{LO} / BTU_F$ = grams of NMOC emissions from process areas in the refinery, per BTU of product F output from the refinery (see DeLuchi [1993]).

0.90 = weight fraction of carbon in NMOCs.

gCO_{LO} / BTU_F = grams of CO emissions from process areas in the refinery, per BTU of product F output from the refinery (see DeLuchi [1993] and revisions in this report).

0.428 = weight fraction of carbon in CO.

$gCO2_{LO} / BTU_F$ = grams of CO_2 emissions from process areas in the refinery, per BTU of product F output from the refinery (see DeLuchi [1993] and revisions in this report).

0.273 = weight fraction of carbon in CO_2.

The effect of this change is quite small, because less than 1% of the crude input is lost.

Note that a similar accounting is not required for the production of alternative fuels, for which the feed-input/fuel-output ratios are assumed to be based on output net of any losses in the plant.

Comparison of our estimates of refinery emissions with those of GM et al. (2002c)

A lifecycle emissions study by General Motors et al. (2002c) uses input/output models of of refinery process areas to estimate refinery pollutant emissions attributable to individual products. The method is similar to but in some respects more detailed than the method used by DeLuchi et al. (1992), which serves as the basis of the estimates in the LEM.

Table 14a compares the GM et al. (2002c) estimates of refinery emissions with the LEM's estimates.Details of the comparison are given in the notes to Table 14a. The LEM estimates of CO_2 emissions fall between the GM et al. (2002c) low and high estimates, but the LEM estimates of emissions of other pollutants are quite a bit higher than even the high estimates of GM et al. (2002c). The main explanation for this difference appears to be that the LEM estimates cover more pollutants from more sources than do the GM et al. estimates:

Source of emissions in refinery	Pollutants estimated in LEM	Pollutants estimated by GM et al. (2002c)
Fuel combustion	CO_2, CH_4, N_2O, NO_x, SOx, PM, CO, NMOCs	CO_2 only?
Process areas	CO_2, CH_4, N_2O, NO_x, SOx, PM, CO, NMOCs	CH_4, N_2O, NO_x, SOx, PM (CO_2?)
Electricity generation for refineries	CO_2, CH_4, N_2O, NO_x, SOx, PM, CO, NMOCs	not counted as refinery emissions

Thus, GM et al. (2002c) apparently do not include emissions of CH_4, N_2O, NO_x, SOx, PM, CO, NMOCs from fuel combustion. This would explain much of the difference between the LEM estimates and the GM et al. (2002c) estimates.

ELECTRICITY GENERATION

Efficiency of electricity generation

Table 6a of DeLuchi [1991]) projected the efficiency of electricity generation from coal, oil, natural gas, methanol, and hydrogen, in the year 2000. In the revised model, the efficiency of coal, natural gas, oil, or biomass generation, in any year from 1970 to 2050, is equal to the BTU equivalent of the net generation in that year divided by the higher-heating value of the fuel input in that year. The net generation and the fuel input, for coal, natural gas, oil, and biomass electric generators (utility and non-utility providers), are projected by the EIA's *AEO* . Fuel used to generate electricity for internal use by non-utilities and co-generators has been ignored. The efficiency for methanol and hydrogen has been estimated. The model looks up the calculated efficiency for the target year, and uses it in all calculations of emissions from electricity generation.

The projected efficiencies are generally higher than the originally assumed fixed values. Thus, this change has caused a significant decrease (>5%) in CO_2-equivalent emissions from electricity fuel cycles, including the EV fuel cycle.

National average mix of fuels used to generate electricity

The projected national-average mix of fuels used to generate electricity in the year 2000 (Table 6b of DeLuchi [1991]) has been replaced with year-by-year projections of electricity generation by fuel type. The EIA's *AEO* projects generation by utilities, non-utilities, and co-generators, from coal, petroleum, natural gas, nuclear power, geothermal, hydropower, waste, biomass, solar thermal, solar photovoltaic, wind, and other sources. These data were used to calculate generation shares by fuel type. (I allocate total natural gas generation to natural-gas boilers and natural turbines, on the basis of installed capacity, and ignore generation by non-utilities and co-generators for their own use.) The model looks up the calculated average fuel mix for the target year, and applies this average mix to natural gas compressors, hydrogen compressors and liquefiers, and "generic" electricity end uses.

This change has a negligible effect on the results.

Marginal mix of power used to recharge electric vehicles

There are two major changes regarding the marginal mix of power used to recharge EVs. First, the national marginal recharging power mix (Table 6b of DeLuchi [1991]) has been changed. Second, the model now has the marginal recharging power mix in each of six regions of the U. S., as well as for the whole U. S. The six are the regional power systems of the Electric Power Research Institute (EPRI): Northeast (mainly New England), East Central (Ohio and neighboring states), Southeast (Tennessee and North Carolina and south), West Central (mainly Minnesota and neighboring states, South Central (Texas, Oklahoma, Louisiana, Arkansas), and West (the Rocky Mountain states and west). A macro, "EVs_by_region," calculates and presents g/mi results by stage of fuel cycle for each of the six regions and the whole U. S.

The basis for these changes is the analysis of EV charging in Yao et al. (1993). Yao et al. (1993) describe the method:

> The generation dispatch scenario...consisted of performing a regional power system hourly operation simulation for peak weekdays, average weekdays, and average weekend days in each month of the year using economic dispatch techniques employed by electric utilities. This accounts for regional differences in electric utility generation mix, daily and seasonal end-use load shapes, and hourly time-of-day impacts. The Zaininger Engineering Company's chronological production simulation program was used to perform the power system dispatch calculations in each of the six EPRI regional power systems (p. 3-1).

Yao et al. (1993) presented electricity use and recharging mix for weekdays and weekend days, in each region (Table 15). Given those results, I calculated the overall recharging mix (weekdays and weekends combined) in each region and for the U. S. as a whole (Table 15).

Mix of power used at aluminum production plants

Previously, I assumed that aluminum production plants drew from the national-average power mix, which is mainly coal-fired. However, Alcoa aluminum (1994) points out that a substantial number of aluminum smelters have been built in conjunction with hydro-electric power plants, and that as a result, hydropower is the primary source of electricity for aluminum plants. According to Alcoa (1994), the International Primary Aluminum Institute (IPAI) tracks and publishes the sources of energy used in the aluminum industry. The IPAI's web site has documents that show the following sources of electrical power used by aluminum producers worldwide in 1997 (gigawatthours) (http://www.world-aluminium.org/iai/stats/es002.html):

	Africa	N. America	S. America	Asia	Europe	Oceania	World	% of Total
Hydro	6,986	65,313	31,898	4,317	25,767	7,340	141,621	55.90
Coal	9,530	28,555	0	8,741	11,595	19,633	78,054	30.81
Oil	0	0	0	81	1,264	0	1,345	0.53
NG	0	77	1,036	13,956	3,632	446	19,147	7.56
Nuclear	200	869	116	2	12,013	0	13,200	5.21
Total	16,716	94,814	33,050	27,097	54,271	27,419	253,367	100

In North America, the mix is 69% hydropower, 30% coal, and 1% nuclear. The mix in the U. S. might be a bit lower, because Canada has a considerably higher proportion of hydropower in its overall mix than does the U. S. the following assumptions were used:

	Coal	Oil	NG boiler	NG turbine	Nuclear	Biomass	Hydro
U. S.	34%	0%	2%	0%	2%	0%	62%
Canada	24%	0%	0%	0%	1%	0%	75%

This change reduces emissions from materials manufacture by a few percentage points.

High-renewables generation scenario

A high-renewables generation scenario has been added, in which less fossil fuel, and more biomass, solar, hydro, wind, and geothermal power is used than in the conventional scenarios. These high-renewables generation mixes are used in the hydrogen and biomass fuel cycles, on the grounds that any large-scale production of renewable transportation fuel is likely to be complemented by a shift to renewable fuels for electricity generation. Thus, for example, the generation mix for power used to compress synthetic gas for transportation has more renewable fuel, and less fossil fuel, than the generation mix for power used to compress fossil natural gas. In the high-renewable scenarios renewable is about 30% of the generation mix, as compared with about 10% in the conventional scenarios.

Uncontrolled emissions from utility boilers

The factors for uncontrolled emissions from utility boilers firing coal, fuel oil, and natural gas, were updated with values from the fifth edition of AP-42 (EPA, 1995, including supplements through 2003) (cf. Table D.4 of DeLuchi [1993]). The changes are insignificant. Emission factors for aldehydes (formaldehyde), PM_{10}, and $PM_{2.5}$ (EPA, 1995, AP-42) have been added.

Criteria pollutant emission factors for coal, oil, and NG-fired utility boilers. AP-42 presents emission factors for different types of combustion technologies, and different types of coal and oil. Ideally, one would represent the actual mix of fuel types and combustion technologies (and emission controls) in use now and projected to be in use in the future. However, although the data are available to do this, they are difficult to obtain. Therefore, the most representative emission factors were used:

- In the case of coal, emission factors for dry-bottom boilers firing pulverized bituminous coal were used.
- In the case of oil, emission factors for "normally" fired (as opposed to tangentially fired) utility boilers burning Number 6 oil were used.
- In the case of CO and NO_X from natural gas, the average of the factors for uncontrolled emissions from large wall-fired boilers and tangential-fired boilers were used. (AP-42 does not distinguish CH_4, NMOC, PM, or SO_X emissions by technology.) .

AP-42 presents the emission factors for NG boilers in units of lbs / 10^6 SCF. To convert these units to lbs / 10^6 BTU, one must divide by BTU / SCF of NG. In the previous version of the model, I used 1031 BTU / SCF (HHV) which is the average heat content of NG in the U. S. However, the fifth edition of AP-42 (EPA, 1995) states that the EPA emission factors are based on a HHV of 1020 BTU / SCF. The model now uses this instead of 1031 BTU / SCF.

The AP-42 emission factors for PM_{10} and $PM_{2.5}$ sometimes do not state whether they include condensable PM. It appears that they do not, so I have added emissions of condensable PM (which is less than 1.0 µm).

Emission factors for biomass-fired utility power plants. The biomass emission factors were adapted from the study by Mann and Spath (1997), which used the ASPEN simulation model to estimate all emissions from a biomass gasification combined-cycle power plant. All of the uncontrolled emissions are below the relevant New Source Performance Standards, and consideraby lower than the updated AP-42 emission factors for wood residue combustion.

Mann and Spath (1997) report total PM. AP-42 emission factors indicate that PM_{10} is 90% of PM, and that $PM_{2.5}$ is 78% of PM.

CH_4 and N_2O emission factors. The most recent supplements to AP-42 include N_2O and CH_4 emission factors for utility boilers, differentiated by type of fuel and firing configuration. The IPCC (1997) summarizes the AP-42 emission factors in its "detailed" emission inventory guidelines. In its "simple" guidelines, the IPCC (1997) uses its judgment to "average" across fuel and boiler varieties and establish generic emission factors for the use of coal, oil, or gas, in what it refers to as the "energy industry" (summarized presented in EPA [1998c]). The AP-42 emission factors for fuel combustion by electric utilities, the IPCC "generic" emission factors, and my assumptions, are shown in Table 16.

The AP-42 N_2O emission factors appear to be consistent with the N_2O emission tests summarized in Delucchi and Lipman (1997). In Table 16, the IPCC (1997) generic factors for oil use in the "energy industry" differ from the AP-42 factors for utility boilers burning fuel oil (which is what are shown in Table 16) because the IPCC (1997) apparently includes and gives great weight to the emission factors for large diesel engines, which factors are, according to AP-42, quite a bit higher than the factors for fuel-oil boilers, and which I exclude from the AP-42 estimates shown in the table.

The assumptions for coal, oil, and NG utility power plants are based on the AP-42 emissions factors. In the case of N_2O from coal-fired plants, I have allowed for the possibility of elevated emissions from the few fluidized-bed combustion plants. In the case of N_2O from NG-fired plants, the average of the factors for controlled and uncontrolled burners were used.

Assumptions for wood-fired power plants are based on the Mann and Spath (1997) study mentioned above. Note, though, that there estimates are orders of magnitude lower than the IPCC recommended emission factors for wood-fired power plants.

Emission-reduction factor due to emission controls

In the GHG model, stack emissions from power plants are estimated simply as:

$$EM_{S,kWh} = \frac{EM_{U,input} \cdot ER}{EFF} \qquad \text{eq. 55}$$

where:

$EM_{s,kWh}$ = emissions from the stack, per unit of power output (g / kWh).

$EM_{u,input}$ = uncontrolled emissions per unit of fuel input (g / 10^6-BTU; see discussion above).

ER = the emission reduction factor due to emission controls; equal to the ratio of controlled emissions to uncontrolled emissions, on average.

EFF = the efficiency of electricity generation (kWh / 10^6-BTU).

Originally, a single value of ER was specified for the year 2000. Now the model estimates ER for SO_2 and NO_2 emissions for the period 1970 to 2050, on the basis of emissions estimates and projections by the EIA and EPA.

The EIA's *AEO* projects total emissions of SO_2 and NO_2 from utility and non-utility generators in the U. S. through the year 2020. The EIA's projections of SO_2 emissions are based on the requirements of the 1990 Clean Air Act Amendments that utilities reduce their SO_2 emissions by 10 million tons, in two phases, in 1995 and 2000 (EIA, *AEO 1996*, 1996). The EIA's projections are consistent with the EPA's independent projections of SO_2 emissions through the year 2010 (EPA, *National Air Pollutant Emission Trends, 1900-1994,* 1995).

With the EIA's projections of emissions, fuel input to power plants, and the sulfur content of coal, one can estimate the ER implicit in the EIA's projections of SO_2 emissions:

$$ER_{SO2,T} = \frac{\left(\frac{TEM_{SO2,Y=T}}{2} - NG_T \cdot K_{NG} \cdot SF_{NG}\right)}{COAL_T \cdot K_{coal,T} \cdot SF_{coal,T} + OIL_Y \cdot K_{oil} \cdot SF_{oil}} \qquad \text{eq. 56}$$

where:

$ER_{SO2,T}$ =the average effective emission reduction factor for SO_2 emissions from coal and oil-fired power plants, in year T.

$TEM_{SO2,T}$ = total emissions of SO_2 from utility and non-utility power generators in year T, as projected by the EIA's *AEO* (tons).

NG_T = total quads of natural gas used by utility and non-utility power generators in year T, as projected by the EIA's *AEO* .

K_{NG} = mass/energy conversion factor for NG (22,321,719 tons-NG/quad-NG; assumed to be the same for all years).

SF_{NG} = the weight fraction of sulfur in natural gas (see discussion elsewhere in this report).

$COAL_Y$ = total quads of coal used by utility and non-utility power generators in year T, as projected by the EIA's *AEO.*

$K_{coal,T}$ = mass/energy conversion factor for coal in year T (tons-coal/quad-coal; projected for different years, as described elsewhere in this report).

$SF_{coal,T}$= the weight fraction of sulfur in coal in year T (projected for different years, as described elsewhere in this report; see Table 4).

OIL_T = total quads of fuel oil used by utility and non-utility power generators in year T, as projected by the EIA's *AEO.*

K_{oil} = mass/energy conversion factor for oil (26,325,634 tons-oil/quad-oil; assumed to be the same for all years).

SF_{oil} = the weight fraction of sulfur in fuel oil (0.0099; assumed to be the same for all years[29]).

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[29] Estimated on the basis of imports and production of residual fuel oil by sulfur-content category in 1996 (million bbl) (EIA, *PSA 1996*, 1997):

	less than 0.31% S	0.31% to 1.00% S	more than 1.00% S
imports	18.9	21.7	50.3

This method assumes that SO_2 emissions from natural-gas fired plants are not controlled (which is reasonable given the extremely low level of uncontrolled emissions), and that SO_2 emissions from oil and coal plants are controlled to the same degree. It accounts for emissions reductions due to the projected decline in the sulfur content of coal as well as reductions due to the use of sulfur removal from the flue gases.

Similarly, the ER implicit in the EIA's projections of NO_2 emissions are estimated as follows:

$$ER_{NO2,T} = \frac{TEM_{NO2,T}}{COAL_T \cdot EM_{U,coal} + OIL_T \cdot EM_{U,oil} + NG_T \cdot \left(EM_{U,NGB} \cdot F_{NGB} + EM_{U,NGT} \cdot (1 - F_{NGB})\right)}$$

eq. 57

where:

$ER_{NO2,T}$ = the average effective emission reduction factor for NO_2 emissions from coal and oil-fired power plants, in year T.

$TEM_{NO2,T}$ = total emissions of NO_2 from utility and non-utility power generators in year T, as projected by the EIA's *AEO* (lbs).

$COAL_T$ = total amount of coal used by utility and non-utility power generators in year T, as projected by the EIA's *AEO* (10^6 BTU).

$EM_{U,coal}$ = uncontrolled emissions of NO_2 from coal-fired plants (lb / 10^6-BTU) (EPA's AP-42; see discussion elsewhere in this report).

OIL_T = total amount of oil used by utility and non-utility power generators in year T, as projected by the EIA's *AEO* (10^6 BTU).

$EM_{U,oil}$ = uncontrolled emissions of NO_2 from oil-fired plants (lb / 10^6-BTU) (EPA's AP-42; see discussion elsewhere in this report).

NG_T = total amount of NG used by utility and non-utility power generators in year T, as projected by the EIA's *AEO* (10^6 BTU).

$EM_{U,NGB}$ = uncontrolled emissions of NO_2 from natural-gas-fired boilers (lb / 10^6-BTU) (EPA's AP-42; see discussion elsewhere in this report).

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refinery production	25.7	71.8	168.0

and assuming 0.2% for the less-than-0.31% category, 0.65% for the 0.31-1.00% category, and 1.3% for the greater-than-1.00 category.

$EM_{U,NGT}$ = uncontrolled emissions of NO_2 from natural-gas-fired turbines (lb / 10^6-BTU) (EPA's AP-42; see discussion elsewhere in this report).

F_{NGB} = of total natural-gas used by power plants, the fraction used in boilers (based on EIA projections and other data; see discussion elsewhere in this report).

This method results in ERs in the range of 0.60, which implies an average effective reduction of 40%, which seems reasonable[30]. This has been applied to uncontrolled NO_2 emissions from all fossil-fuel combustion.

The EIA does not project emissions of PM. The EPA (*National Air Pollutant Emission Trends 1900-1996, 1997*) does, but somewhat implausibly assumes that no new particulate matter controls will be applied to power plants. As a result, the EPA projects increasing total PM emissions from power plants through the year 2010. Given the recent proposed tightening of the ambient air quality standard for PM, it seems unlikely that PM emissions from power plants escape further controls.

The emission reduction factor, ER, can be estimated for the year 1994:

	Coal	Oil	Gas boiler	Gas turbine
PM	0.015	0.25	0.10	0.10
PM_{10}	0.030	0.25	0.10	0.10
$PM_{2.5}$	0.050	0.33	0.10	0.10

These base-year (1994) values applied to AP-42 uncontrolled emission factors approximately reproduce the EPA's estimates of total PM and PM_{10} emissions from power generation in 1994 (EPA, *National Air Pollutant Emission Trends 1900-1994*, 1995). I assume that the reduction factors for coal and oil decline by 1.5% per year.

NMOCs, CO, CH_4, and N_2O emissions were assumed to remain uncontrolled indefinitely.

Fuel cycle emissions due to the use of limestone to scrub sulfur from flue gases of coal-fired power plants

In Appendix D of DeLuchi (1993), the formation of CO_2 from the use of limestone ($CaCO_3$) to scrub sulfur from flue gases of coal-fired power plants is accounted for:

$$CaCO_3 + SO_2 \text{-->} CaSO_3 + CO_2$$

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[30] Most NO_x controls reduce emissions by 30-50% (EPA, AP-42, 1995; EIA, *Electric Power Annual 1995*, 1996).

CO_2 from the disposal of the $CaSO_3$ sludge is also included . However, I did not account for emissions from the production and transport of the rather substantial amount of limestone required for the scrubbing. The model now includes CO_2-equivalent emissions from the limestone fuel cycle, per 10^6 BTU of coal input:

$$GHGSC_{ls,T} = \frac{\left(\frac{MW_{CO2}}{AW_S} \cdot 453.6 + \frac{MW_{CaCO3}}{AW_S} \cdot GHG_{ls} \cdot AUF_{ls} \right) \cdot FSC_{ls,T} \cdot \left(1 - ER_{SO2,T}\right) \cdot SF_{coal,T} \cdot 2000}{HHV_{coal,T}}$$

eq. 58

where:

$GHGSC_{ls,T}$ = CO_2-equivalent GHG emissions from the use of limestone to scrub sulfur from the flue gases of coal-fired power plants in year T (g / 10^6-BTU-coal).

MW_{CO2} = the molecular mass of CO_2 (Table 5).

MW_{CaCO3} = the formula mass of limestone (calcium carbonate) (100 g / mole).

AW_S = the molar mass of sulfur (32 g / mole).

GHG_{ls} = fuel cycle CO_2-equivalent GHG emissions from the production and transport (but not use) of limestone (g-CO_2-equivalent / lb-limestone; estimated to be about 80).

AUF_{ls} = the ratio of the actual to the theoretical (stoichiometric) use of limestone to scrub sulfur (Spath et a. [1999] report that an average U. S. coal-fired plant that uses coal with 4% S by weight, and limestone to scrub the flue gases, consumes 448,171 kg-coal / gWh and 90,704 kg-limestone / kWh, and emits 6,400 kg-SO_X / gWh. This implies about 6-g-$CaCO_3$ / g-S-scrubbed, about twice the stoichiometric ratio of 100 / 32 or about 3:1. Therefore, the ratio of the actual to the theoretical use is assumed to be 2.0.).

$FSC_{ls,T}$ = of plants that control sulfur emissions, the fraction that do so with limestone (The value assumed by DeLuchi [1993] of 0.50 has been used).

$ER_{SO2,T}$ = the average effective emission reduction factor for SO_2 emissions from coal and oil-fired power plants, in year T (see discussion of emission controls, above).

$SF_{coal,T}$ = the sulfur weight fraction of coal in year T (see Table 4).

453.6 = g / lb

2000 = lbs / ton

$HHV_{coal,T}$ = the higher heating value of coal in year T (10^6-BTU / ton-coal; see Table 4).

Note that this includes the CO_2 from the scrubbing reaction. These emissions are included as "upstream" emission in the fuel cycles in which coal is used as a fuel for utility or industrial boilers[31].

Nuclear fuel cycle

The LEM distinguishes three sources of fuel for nuclear power plants:

i) natural uranium, from mines;
ii) uranium or plutonium recycled from spent nuclear reactor fuel, as mixed oxides;
iii) ex-military weapons grade uranium

Each of these has a different fuelcycle, represented in the LEM (in the case of the U. S.) as follows:

Stage	Natural uranium	Recycled nuclear fuel	Military uranium
Uranium production	mining (use actual U. S. energy data; discussed more below)	reprocessing (estimated relative to uranium mining)	reprocessing (estimated relative to uranium mining)
Conversion to UF_6	combined conversion, fabrication, disposal stage (simple energy inputs)	same as (not distinguished from) natural uranium	same as (not distinguished from) natural uranium
Enrichment	enrichment (detailed representation of energy requirements, by technology and country; discussed more below)	enrichment (estimated relative to requirements for natural uranium)	enrichment (estimated relative to requirements for natural uranium)
Fabrication	see "conversion"	same as (not distinguished from) natural uranium	same as (not distinguished from) natural uranium

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[31] In the previous version of the LEM, there was an error in the calculation of CO_2 emissions from the scrubbing process: the oxidation of C to CO_2 was counted twice.

Disposal	see “conversion”	same as (not distinguished from) natural uranium	same as (not distinguished from) natural uranium
Transportation	all steps characterized in detail, for U. S. conditions	same as (not distinguished from) natural uranium	same as (not distinguished from) natural uranium

The most energy-intensive stages are production and enrichment, which the LEM characterizes in some detail.

Uranium production. The LEM represents the energy requirements of uranium production in each country C as follows:

$$EUP_C = EUM_{US} \cdot UR_C \cdot \sum_{UPS} UP_{UPS,C} \cdot AEUP_{UPS} \qquad \text{eq. 59a}$$

where:

subscript UPS = the uranium production source (U. S. mines, Canadian mines, FSU mines, Australian mines, South African mines, other mines, reprocessed tails or spent fuel, military high-enriched uranium)

subscript C = the nuclear-power-consuming country targeted for analysis

EUP_C = the energy requirements of producing uranium for nuclear power reactors in country C (BTUs-production-energy / gWh-power-generated)

EUM_{US} = the energy requirements of uranium mining in the U. S. (BTUs-mining-energy / ton-U_3O_8-equivalent produced) (see DeLuchi [1993] and the “Energy Used in Mining” section of this report)

UR_C = the uranium requirements of nuclear reactors in country C (tons-U_3O_8-equivalent / gWh-net-nuclear-power-generated) (discussed below for the U. S., and in Appendix B for other countries)

$UP_{UPS,C}$ = uranium produced from source UPS for reactors in country C, as a fraction of the total uranium required by reactors in country C (discussed below for the U. S., and in Appendix B for other countries)

$AEUP_{UPS}$ = the average energy intensity of producing uranium from source UPS *relative* to EUM_{US}, the energy requirement for mining uranium in the U. S. (unitless) (discussed below)

The uranium requirements of nuclear reactors. The production of electricity by nuclear power plants ultimately is a fairly direct function of the amount of the fissile isotope of uranium – U-235 – consumed. Naturally occurring uranium contains only about 0.7% U-235, but through a process called “enrichment” this is increased to 3% to

4%. Nuclear power plants “burn” enriched uranium fuel until there is only 0.5% to 0.8% U-235 left in it. The greater the percentage of the U-235 in the input fuel and the lower the percentage of U-235 in the spent fuel the greater the amount of U-235 “burned” and hence the greater the electricity generation, per ton of uranium oxide input. The uranium requirement of nuclear reactors can range from 0.029 tons U_3O_8 per gWh (when relatively highly enriched uranium is burned until the U-235 level is relatively low) to 0.039 tons U_3O_8 per gWh (when less highly enriched uranium is not burned as long) (World Nuclear Association, 2002). Worldwide, nuclear reactors in recent years have required about 0.035 tons U_3O_8 per gWh (World Nuclear Association, October 2002, December 2002; EIA, *AEO 1999*, 1998; EIA, *AER 1997*, 1998; EIA, *Nuclear Power Generation and Fuel Cycle Report 1996*, 1996; EIA, internet projections, 2003).

The World Nuclear Association (October 2002) states that from 1970 to 1990 the ton/gWh uranium requirement of nuclear reactors in Europe declined by 25% due to the use of more highly enriched fuel and longer burn up of the fuel (to lower levels of U-235 in the depleted fuel). It also shows a graph that projects that this trend will continue worldwide through 2010. The EIA projections of ton/gWh uranium requirements for nuclear reactors worldwide through the year 2025 do show a decrease in uranium requirements in Western Europe (EIA, internet projections, 2003). More detailed projections for the U. S. also indicate a slight decrease (EIA, internet projections, 2003).

Given these data and projections, I assume a value of 0.035 tons U_3O_8 (equivalent) per gWh net nuclear power generated in the U. S. in 2000, decreasing by 0.2% per year. Assumptions for other countries are given in Appendix B.

Sources of uranium. The EIA’s *Uranium Industry Annual 2001* (2002) reports sources of uranium required by U. S. nuclear utilities, the World Nuclear Association (October 2002) projects sources of uranium supply for the world through 2010, and other World Nuclear Association papers (July 2002 and August 2002) show uranium production from world mines. The World Nuclear Association (October 2002) projects that in 2010 mine production will satisfy 75% of world uranium demand, military uranium will satisfy 20%, and reprocessed fuel and re-enriched tails about 5%.

I use the EIA data on sources of uranium to U. S. utilities in 2001 (*Uranium Industry Annual 2001*, 2002), along with my assumptions regarding the use of military uranium and reprocessed fuel in the U. S., to estimate the following:

Source of uranium	Contribution to U. S. utility uranium requirements
U. S. mines	0.24
Canada mines	0.27
Former Soviet Union mines	0.17
Australia mines	0.17
South Africa mines	0.05
Other mines	0.00

reprocessed fuel	0.01
military uranium	0.10

Relative energy intensity. The LEM requires as an input the energy intensity of uranium production (BTUs / ton-uranium) for each production source *relative* to the energy intensity of production from uranium mines in the U. S. I assume that this relative intensity is 1.0 for all mine production worldwide, 0.50 for reprocessed tails and spent fuel, and 0.30 for military high-enriched uranium.

Uranium enrichment. The energy requirement of uranium enrichment, which is by far the most energy-intensive step in the nuclear fuelcycle, is now modeled in the LEM in more detail. Because there is international trade in uranium enrichment services (measured in separative work units, or SWUs), the LEM now represents, for each country that provides enrichment services: i) its contribution to the total SWU requirement of nuclear power plants in any one of the consuming countries that can be targeted for analysis; ii) the fraction of SWUs provided by different enrichment technologies (gaseous diffusion, centrifuge, molecular laser [SILEX]); and iii) the MWh of electrical energy required per SWU. The U. S. A., France, Germany, the Netherlands, the U. K., Japan, Russia and China provide the world's uranium enrichment services. With these data, and an estimate of the SWUs required per ton of natural uranium to be enriched, the model calculates the figure of interest: the energy efficiency of uranium enrichment, in mWh-enrichment-energy / mW-power-generated.

Formally:

$$EEU_C = SWUU_C \cdot SWUU_C * \cdot UR_C \cdot \sum_{EC} SWUP_{EC,C} \cdot AESWU_{EC}$$

$$SWUU_C * = \sum_{UPS*} UP_{UPS*,C} \cdot SWUUR_{UPS*}$$

$$AESWU_{EC} = \sum_{ET} SWUF_{ET,EC} \cdot ESWU_{ET,EC}$$

eq. 59b

where:

subscript EC = the enriching country (U. S. A., France, Northern Europe, Japan, Former Soviet Union, China, and other)
subscript C = the nuclear-power-consuming country targeted for analysis
subscript ET = the uranium enriching technologies (gaseous diffusion, centrifuge, AVLIS)
EEU_C = the energy efficiency of uranium enrichment for nuclear power produced in country C (mWh-enrichment-energy / mWh-power-generated)

$SWUU_C$ = the enrichment-service requirement of nuclear utilities in country C (SWUs / ton-U_3O_8-from mines) (discussed below)

$SWUU_C*$ = adjustment to account for the enrichment service requirement of uranium from secondary sources (i.e., non-mine sources: military high-enriched uranium, reprocessed spent fuel)

UR_C = the uranium requirements of nuclear reactors in country C (tons-U_3O_8-equivalent / mWh-net-nuclear-power-generated) (discussed above)

$SWUP_{EC,C}$ = SWUs produced by enriching country EC for consuming country C, as a fraction of the total SWUs required by C (values for U. S. discussed below; values for other countries discussed in Appendix B)

$UP_{UPS*,C}$ = uranium produced from secondary source UPS* (reprocessed spent fuel, military high-enriched uranium) for reactors in country C, as a fraction of the total uranium required by reactors in country C (discussed above)

$SWUUR_{UPS*}$ = the SWUs required to enrich a ton of uranium from secondary source UPS* *relative* to that required to enrich a ton of uranium from mines (the latter being parameter SWUUc) (assumed to be 0.9 for spent fuel [The EIA *Nuclear Power Generation and Fuel Cycle Report 1996*, 1996, reports that the use of mixed-oxide fuel reduces the enrichment work required, by on the order of 10%], and 0 for high-enriched uranium, which in fact has to be diluted rather than enriched)

$AESWU_{EC}$ = the weighted-average energy-intensity of SWU production in enriching country EC (mWh-enrichment-energy / SWU-produced)

$SWUF_{ET,EC}$ = of total SWUs produced by enriching country EC, the fraction produced by enriching technology ET (discussed below)

$ESWU_{ET,EC}$ = the energy intensity of SWU production by technology ET in enriching country EC (mWh-enrichment-energy / SWU-produced) (discussed below)

SWUs required per ton of uranium (from mines) enriched in country C. The amount of work required to enrich the U-235 content of a ton of uranium is a function of the initial concentration of U-235, the concentration of U-235 in the enriched stream, the concentration in the waste stream, and the mass of the streams (EIA, *Uranium Industry Annual 2001*, 2002). Generally, the greater the desired concentration of U-235 in the nuclear fuel, the more SWUs required.

Given this, one would expect that the EIA and World Nuclear Association projections of the use of more highly enriched uranium fuel (see discussion above) would be accompanied by projections of greater SWU requirements per ton of uranium. This does indeed seem to be the case: the EIA's most recent projections (2003) indicate that SWUs / ton-U_3O_8 increase at about 0.25% / year, which is about the same rate that projected uranium requirements (in tons / gWh) decrease.

I assume that the U. S. requires 480 SWUs / ton-U_3O_8 natural uranium in the year 2000, and that the requirement increases at 0.25% / year.

Source of SWUs required by U. S. utilities. The EIA's *Uranium Industry Annual 1998* (1999) and *Uranium Industry Annual 1998* (2002) show U. S. utility purchases of enrichment services by country of origin and delivery year (expressed here as a percentage of the total enrichment services, in SWUs, provided):

Enrichment plant location	1994	1995	1996	1997	1998	1999	2000	2001
United States	82%	71%	72%	68%	56%	46%	44%	12%
France	6%	9%	14%	8%	7%	8%	9%	13%
Japan	0%	0%	0%	0%	0%	0%	0%	0%
Germany, Netherlands, U. K.	5%	9%	5%	4%	13%	10%	20%	16%
Russia	5%	12%	10%	20%	23%	34%	25%	56%
China	3%	0%	0%	0%	0%	1%	2%	3%?
Argentina, Pakistan, S. Africa	0%	0%	0%	0%	0%	0%	0%	0%

Note that Russia has supplied an increase share of the enrichment service for U. S. utilities. (This is consistent with the implication in the EIA's *Nuclear Power Generation and Fuel Cycle Report 1996*). The LEM uses the actual distributions estimated above for the years 1994-2001, and assumes that 1992 to 1994 is the same as 1994, that post-2001 is in between 2000 and 2001, that 1985 to 1991 is 88% U. S. and 12% France (based on EIA data discussed in DeLuchi, 1993, Appendix I), and that 1970 to 1985 is 100% U. S.

SWU production by technology and enriching country. Presently, the U. S., France, and China operate gaseous diffusion plants, and the other countries operate centrifuge plants (EIA, *Nuclear Power Generation and Fuel Cycle Report 1996* , 1996). The U. S. is considering a centrifuge plant, and has an interest in the SILEX process, and Japan and Western Europe are planning additional centrifuge plants (EIA, *Nuclear Power Generation and Fuel Cycle Report 1996* , 1996; World Nuclear Agency, 2003. Given this, my assumptions are shown in the table below.

Energy requirement of SWU production, by technology. DeLuchi (1993) reports that gaseous diffusion requires 2.40 mWh / SWU; centrifuge and AVLIS (a predecessor of SILEX), 0.10 mWh / SWU. The World Nuclear Association (2003) reports 2.4 or 2.5 mWh / SWU for gaseous diffusion and "as little as" 0.05 mWh / SWU for "modern" gas centrifuge plants. Actual data on the electricity consumption of the old gaseous diffusion plants in the U. S. indicate that they consume more than 2.4 mWh / SWU (DeLuchi, 1993). Given this information, I assume the following:

Enriching country	**Source of SWUs**			**mWh/SWU**		
	diffusion	*centrifuge*	*SILEX*	*diffusion*	*centrifuge*	*SILEX*

U. S.	difference	0.00	0.2 after 2012	3.00	0.06	0.05
France	difference	0.2 after 2006	0.00	2.50	0.06	0.05
N. Europe	0.00	1.00	0.00	2.40	0.06	0.05
Japan	0.00	1.00	0.00	2.40	0.06	0.05
Former Soviet Union	0.00	1.00	0.00	2.40	0.06	0.05
China	difference	0.2 after 2010	0.2 after 2015	2.40	0.06	0.05
Other	0.00	1.00	0.00	2.40	0.06	0.05

Emissions from the use of electricty to enrich uranium. Given an estimate of the energy efficiency of uranium enrichment, from above (MWh-enrichment-power / MWh-nuclear-power-generated), the model calculates emissions, in g-CO_2equivalent / MWh-nuclear-power, by multiplying the energy efficiency figure by an aggregate emission factor, in g-CO_2equivalent / MWh-enrichment-power. This aggregate emission factor, in turn, is calculated in the normal manner in the LEM, using the following parameters:

i) uncontrolled emission rates per unit of fuel input for each type of power plant;
ii) the energy efficiency of electricity generation; the generation mix;
iii) emission control extent and effectiveness; and
iv) CO_2-equivalency factors (Appendix D).

In this calculation, the model uses the actual generation mix in the producing countries that are enriching the uranium (for use in the target or consuming country), but uses the generation efficiency values and emission control parameters for the *target* or consuming country. (Ideally, one would use generation efficiency and emission control parameters as well as generation mix parameters specific to the actual producing countries, but for simplicity I chose to use producing-country-specific values for only the most important of these – generation mix.)

The generation mix of each uranium producing country is weighted by its contribution to the total SWU requirements of the target consuming country. The source of SWUs for the U. S. is given above; the source of SWUs for other countries is given in Appendix B. The generation mix in uranium producing countries is assumed to be as follows (year 2020, except as noted):

Enriching	**generation mix by type**	**notes**

country	*coal*	*oil*	*gas*	*nuke*	*hydro*	
U. S.	88%	0%	0%	8%	3%	analysis of actual generation mix for enrichment (DeLuchi, 1993)
France	6%	1%	3%	77%	13%	IEA (2002b) data for year 2000
N. Europe	51%	1%	10%	30%	5%	IEA data for Germany (see App. B; calculated mix in target year)
Japan	19%	15%	26%	30%	9%	IEA data for Japan (see App. B; calculated mix in target year)
FSU	20%	4%	42%	15%	19%	IEA data for Russia (see App. B; calculated mix in target year)
China	78%	3%	0%	1%	16%	IEA data for China (see App. B; calculated mix in target year)
Other	60%	3%	21%	0%	15%	my assumption

Uranium transportation. DeLuchi (1993) estimates the energy requirement of transporting uranium and nuclear fuel in the U. S., and finds it to be a negligible fraction of nuclear power output. Because of this, I do not model the international transport of uranium.

Standby diesel generators. To estimate emissions from standby-diesel generators, the model now uses the emission factors for large rather than small stationary diesel engines. (EPA's AP-42 states that large stationary diesel engines are used for standby generation, and to operate emergency cooling-water pumps at nuclear power plants.) (This change is utterly insignificant.) Also, the fuel consumption of the standby generators, in gallons-diesel fuel per million BTU of nuclear power generated has been made into a separate input variable.

Greenhouse-gas emissions at hydropower facilities

Flooded land at hydropower facilities can produce greenhouse-gas emissions, as inundated soils and organic matter degrade and their carbon content becomes mineralized to CO_2 and CH_4. (These emissions are analogous to emissions of CO_2 and CH_4 from natural processes in pristine lakes and wetlands.) The emissions, in grams-CO_2 equivalent / kWh-generated, can be estimated simply as the product of the emission rate per unit area (g-CO_2-equivalent / ha), and the areal intensity of power generation (ha / kWh). However, it is difficult to estimate any sensible average worldwide or U. S. emission rate, because areal emissions have been measured only at but a few sites in Canada, and the areal intensity of generation varies by orders of magnitude (Gagnon and van de Vate, 1997). Gagnon and van de Vate (1997) speculate that the worldwide average might be on the order of 20 g-CO_2-equivalent / kWh, including emissions from construction, which appear to be on the order of 5 g / kWh.

St. Louis et al. (2000) review data available in 2000 and estimate gross and net fluxes of CO_2 and CH_4 from surface reservoirs globally: 7 to $15 \cdot 10^{14}$ g/yr of CO_2, and about $0.7 \cdot 10^{14}$ g/yr CH_4. (The gross fluxes are similar to the net fluxes.) Assuming that only 1/3 of the total is from reservoirs which would not have been built were the production of hydropower not desired, and given $0.7 \cdot 10^{14}$ kWh of hydropower produced in 1999 (EIA, *International Energy Annual 1999*, 2001) the result is about 5 g CO_2/kWh and 0.3 g CH_4/kWh.

On the basis of information presented above and reviewed in more detail in Appendix E to this report, I assume average "net" emissions of 0.3 g-CH_4/kWh, and 5 g-CO_2/kWh, excluding emissions from construction, which in this analysis are not counted for any power generation facilities. "Net" emissions are equal to total ("gross") emissions from hydropower facilities less the emissions that would have come from the area had it not been flooded. (Measurements by Kelly et al. [1997] suggest that emissions prior to inundation are small compared to emissions measured after inundation.)

PRODUCTION OF ALTERNATIVE FUELS

Feedstock and process energy use of alternative-fuel production plants

My previous estimates of the use of feedstock and process energy by methanol, ethanol, and SNG plants in the year 2000 have been replaced with estimates of:

i) inputs of specific fuels and feedstocks, per unit of output, in a base year (usually 1994); and

ii) the annual percentage change in the inputs through the projection period. This method allows the calculation of feedstock and energy use in any year, but anchors the calculation to the presumably reasonably well-known data on the feedstock and energy use of current-technology plants. Of course, this does not eliminate uncertainty in projecting energy use; rather, it locates the uncertainty in a single, explicit parameter: the annual percentage change in energy use.

For many of the input/output values shown in Table 17, I have estimated one annual rate of change from the base year until 2020, and a lower rate of change in feedstock use after 2020. I do this because I expect that many alternative-fuel production processes will develop rapidly over the next 20 years or so and then settle into a more mature development phase thereafter. The formulae are:

if T < T_C

$$IN_T = IN_B \cdot \left(1+\frac{PCY1}{100}\right)^{T-T_B}$$

else

$$IN_T = IN_B \cdot \left(1+\frac{PCY1}{100}\right)^{T_C-T_B} \cdot \left(1+\frac{PCY1 \cdot R_{PCY2}}{100}\right)^{T-T_C} \quad \text{eq. 60}$$

where:

T = the target year of the analysis.
T_C = the year in which the annual rate of change in the energy input/output parameter of interest changes (from PCY1 to PCY2; assumed to be 2020)
T_B = the base year (corresponding to IN_B).
IN_T = the value of the energy input/output parameter (e.g., lbs of wood in per gallon of ethanol out) in target year T.
IN_B = the value of the energy input/output parameter (e.g., lbs of wood in per gallon of ethanol out) in base year B (see the discussion in the text, and Table 17).
PCY1 = the annual percentage change in the value of the energy input/output parameter IN, up to the time T_C (see the discussion in the text, and Table 17).
PCY1 = the annual percentage change in the value of the energy input/output parameter IN, after time T_C.
R_{PCY2} = the ratio of PCY2 to PCY1 (assumed to be 0.30).

These formulae are used whenever PCY1 is greater than or equal to 0.6%/year. If PCY1 is less than 0.5%/yr, I assume that PCY2 = PCY1 (i.e., that R_{PCY2} = 1.0)

Table 17 presents the new parameter values (cf. Tables J.1, J.3, J.4, K.7, and K.11 of DeLuchi [1993], and Table 3 of DeLuchi [1991]). The estimates for ethanol and methanol from wood, and ethanol from corn, have been updated on the basis of a review of recent literature (see the discussions below).

I emphasize that mine are meant to be projections of actual energy use and emissions, not best-case or worst-case scenarios. Marland (1994) properly points out that some of the differences between past estimates of GHG emissions from the corn-to-ethanol fuel cycle are due to the difference between assuming “best practice” (e.g., the use of the most efficient conversion technology) and “typical practice” (the use of the average conversion technology). Here, I wish to project what is most likely to occur, not what might occur under the best circumstances.

Note that I have added grass as a feedstock for the production of ethanol. The energy inputs and outputs of the grass-to-ethanol process are taken from NREL's detailed evaluation of the biomass fuel cycle (Riley and Schell, 1992). I also have added biodiesel from soybeans, with the input/output parameters estimated on the basis of the data reviewed in Appendix A to this report.

Note, too, that Table 17 shows only purchased electricity inputs; it does not show any excess power marketed to the grid. The [negative] emissions related to any electricity sales are calculated separately.

Finally, I have added emissions from the lifecycle of chemicals (sulfuric acid, lime, nitrogen, phosphate, solvents, catalysts, miscellaneous chemicals) used in the wood/ethanol, grass/ethanol fuel cycles, soy/biodiesel fuel cycles, and corn/ethanol fuel cycles.

$$GHGCH = (1+FL)\cdot U \cdot \sum_{C} Q_C \cdot EF_C$$

$$EF_C = \sum_{E} Q_E \cdot EF_E \qquad \text{eq. 61}$$

where:

GHGCH = lifecycle-CO_2-equivalent emissions due to the use of chemicals in the fuel production stage (g-CO_2-equivalent/10^6-BTU-net-fuel-output).

U = conversion factor (e.g., grams/gallon to grams/10^6-BTU).

FL = fraction of fuel production lost due to evaporation or spillage (Appendix B of DeLuchi [1993], and updates thereto in this report).

Q_C = quantity of chemical C used per unit of fuel output (e.g., gallons of solvent per gallon of biodiesel produced) (Riley and Schell, 1992; Ahmed et al., 1994).

EF_C = the emission factor for the production of chemical C (g-CO_2-equivalent/unit-chemical-C; e.g., grams per gallon of solvent).

Q_E = BTUs of energy source E used to make a unit of chemical C (e.g., BTUs of NG per gallon of solvent) (Appendix H; Ahmed et al., 1994; my estimates).

EF_E = the fuel cycle emission factor for energy source E (g-CO_2-equivalent/BTU-E; e.g., grams per BTU natural gas) (Table A.2 of DeLuchi [1993], and updates thereto in this report).

In the soy/biodiesel process, a petroleum solvent, n-hexane, is used to extract the oil from the soybeans (see Appendix A to this report for details). A small fraction of this solvent evaporates. I assume that these evaporative emissions will be controlled as fugitive NMOC emissions, and that the controls will capture 85% of the evaporated solvent (this is towards the upper end of the range of the effectiveness of controls on fugitive emissions at refineries [DeLuchi et al., 1992]), which leaves 15% as an emission to the atmosphere.

All of the evaporated fuel is counted is counted as fuel consumption.

Feedstock and process energy use of biomass/alcohol plants

Methanol from wood. Newer data from Stone and Lynd (1993), the U. S. DOE (1990), and Wyman et al. (1993) are consistent with the data in Table K.11 of DeLuchi (1993). The new assumptions are based on the data of the U. S. DOE (1990) and Wyman et al. (1993):

	As in U. S. DOE (1990)		kJ/kJ-MeOH	
	dry ton wood/gal	*kWh/gal*	*Wood*	*Power*
Koppers-Totzek (K-T) low-pressure oxygen gasification	0.0078	0.5382	2.06	0.028
Institute of Gas Technology (IGT) high-pressure oxygen gasification	0.0066	0.7064	1.75	0.037

	As in Wyman et al. (1993)		kJ/kJ-MeOH	
	Mg wood/Mg-MeOH	*GJ-elec./Mg-wood*	*Wood*	*Power*
Indirectly heated gasification	1.63	0.647	1.60	0.052

According to the U. S. DOE (1990), the low-pressure Koppers-Totzek process is commercially available today, the high pressure IGT process will be available by the year 2000, and the indirectly heated gasification process will be available in the long run. Thus, I assume that the near-term technology is low-pressure oxygen gasification, and that in the longer term the technology evolves toward indirectly heated gasification.

Methanol also can be synthesized from the products of the gasification of grass, providing the grass is harvested late and has a low protein content (Lynd, 1997). However, I do not have process data for this, and so have not included a grass-to-methanol pathway in the LEM.

Ethanol from wood or grass. Wooley et al. (1999) perform a detailed engineering and economic analysis of an ethanol production plant, based on “technology that has been developed or is currently researched and close to completion” (p. 56). The modeled uses “co-current dilute acid prehydrolysis of the lignocellulosic biomass with simultaneous enzymatic saccharification of the remaining cellulose and co-fermentation of the resulting glucose and xylose to ethanol" (p. 4). The feedstock is yellow poplar hardwood. The model estimates the following inputs and outputs, in pounds per gallon ethanol unless noted otherwise:

dry feedstock	29.41
sulfuric acid	0.66
lime	0.25
ammonia	0.50
corn steep liquors	0.70
nutrients	0.06
sulfate	0.14
antifoam (corn oil)	0.08
diesel (for bulldozers handling feedstock)	0.16
makeup water	65.90
BFW chemicals	0.00
cooling water chemicals	0.00
waste-water treatment nutrients	0.08
waste-water treatment chemicals	0.00
solids disposal	1.27
electricity credit (kWh / gal)	-1.76

They also project that the feedstock requirement (lbs / gal) will decline over time, as the technology improves:

	base case	2005	2010	2015
feedstock lbs / gal (gal / ton)	29.4 (68)	24.7 (81)	21.3 (94)	20.2 (99)
(excess) power (kWh / gal)	-1.76	-2.80	-1.22	-0.00

Wooley et al. (1999) also note that NREL has contracted to Dartmouth University to investigate long-term, advanced ethanol production technologies. Lynd (1996a) from Dartmouth projects the following:

	Values from Lynd (1996a)		Calculated BTU / BTU- output	
	gal/ton	*kWh/gal*	*Wood*	*Power*
Current technology, Rankine cycle	91.3	-2.24	2.16	-0.0903
Advanced technology, Rankine cycle	107.5	-3.06	1.84	-0.1234
Advanced technology, BGCCGT	107.5	-5.13	1.84	-0.2069
Best Parameter, Rankine cycle	127.7	-3.16	1.55	-0.1274

(BGCCGT = biomass gasification combined-cycle gas turbine.)

There is a rather considerable discrepancy between gal/ton estimates of Wooley et al. (1999) and those of Lynd (1996a). For example, Lynd's (1996a) "current technology" gal/ton estimate is much higher than the current-technology base-case gal/ton estimate of Wooley et al. (1999). We assume that the Wooley et al. (1999) parameters apply to the year 2000, and that Lynd's (1996a) "advanced technology, BGCCGT" estimates applies to the year 2025, and then estimate an annual percentage change accordingly.

There also is a considerable discrepancy in the estimates of excess power generated (in kWh/gal). Note that Wooley et al. (1999) project no excess power in 2015, (because the process is optimized to maximize the gal/ton output), whereas Lynd (1996a) projects substantial excess power even as the gal/ton output is increased considerably beyond that projected by Wooley et al. (1999). However, Wooley et al. (1999) provide information that at least partially explains the discrepancy: in their analysis the boiler, burner, and turbogenerator are not [yet] optimized. I assume that the change in kWh/gal follows a two-side logistic curve (Eq. 3a), with the following parameter values:

	grass-to-ethanol	wood-to-ethanol
year 2000 value	0.55 kWh/gal	1.10 kWh/gal
lower limit	0.00 kWh/gal	0.00 k Wh/gal
upper limit	6.00 kWh/gal	7.00 kWh/gal
"k" exponent	0.15	0.15

Elsewhere in this report, I discuss my treatment of the fate of the excess electricity produced.

Wooley et al. (1999) do not project the use of inputs other than feedstocks, beyond the base case. In the absence of data, I assume that inputs per unit of wood feedstock input (rather than per unit of ethanol output) remain constant.

Kadam et al. (1999) use NREL's "Aspen" model to estimate inputs and outputs for enzymatic hydrolysis of rice straw, forest residue, and chaparral. See Wyman (1999) for a discussion of the technology, economics, and commercialization potential of producing ethanol from lignocellulose.

Feedstock and process energy use of natural-gas to hydrogen plants

In at least the near term, natural gas will be the cheapest source of hydrogen. Hence, I have added to the model natural gas as a hydrogen feedstock.

The conventional way to produce hydrogen from natural gas is to reform methane with steam at high temperature, to produce a mixture of carbon monoxide and hydrogen:

$$CH_4 + H_2O \text{ --> } CO + 3H_2$$

The CO / H_2 mixture is “shifted” to CO_2 and H_2 by low-temperature reaction of the CO with steam:

$$CO + H_2O \text{ --> } CO_2 + H_2$$

Hence, the overall reaction is:

$$CH_4 + 2H_2O \text{ --> } CO_2 + 4H_2$$

Finally, the CO_2 and the hydrogen are separated is a pressure-swing adsorption unit, from which the CO_2 is vented to the atmosphere.

Note that half of the hydrogen comes from natural gas, and half comes from water.

The best estimates of the energy inputs and outputs of conventional reforming are consistent, and indicate an energy-out / energy-in ratio of on the order of 85%[32]. Rosen and Scott (1998) used the ASPEN PLUS process simulator to estimate the energy efficiency of several hydrogen production processes. Assuming that natural gas is pure methane, and including in the energy input the energy needed to generate electricity, they estimated that 100 BTUs of natural gas produce 86 BTUs of hydrogen.

Similarly, Katofsky performed a detailed thermodynamic analysis of an efficient steam reforming process, and estimated 1.11 BTUs- NG / BTU-H_2 (HHV), and 0.029 $BTU_{electric}$ / BTU-H_2 (Blok et al, 1997), resulting in 88% efficiency with electricity at 3412 BTU / kWH, and about 85% efficiency with electricity at 8600 BTU / kWH.

Spath and Mann (2001) report energy balances estimated by SRI for refinery production of hydrogen by catalytic steam reforming of natural gas. The plant consumes 159.6 MJ-NG / kg-H_2 (LHV), which corresponds to an efficiency of about 80% on a HHV basis.

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[32] NREL (1992) cites an estimate of 68%thermal efficiency in 1990, but the estimate is undocumented. It is likely that NREL has confused the overall thermal efficiency (H_2-out / NG-in) with what might be called a natural-gas conversion ratio: NG-feedstock-to-H_2 / total-NG-in, which ratio, according to Rosen and Scott (1998), is 67%. (In other words, 33% of the input NG is used as a process fuel rather than a chemical feedstock.) But the energy output of the plant is much greater than the energy content of the 67% of the gas that is a feedstock, because half of the hydrogen comes from the decomposition of water.

Marquevich et al. (2002) estimate that 2.4 kg of NG are input to a steam reforming plant for every kg of H_2 produced. This corresponds to a thermal efficiency (HHV basis) of at least 85%, depending on the heating values of the input and output streams.

Steinberg (1998) estimates that steam reforming requires 71.9 kcals of total energy (feedstock + process) per mole of H_2 produced, indicating an efficiency of about 95%, apparently excluding electricity input.

There are other ways to produce hydrogen from natural gas. Bromberg et al. (1998) project that plasma reforming of methane would require about 40 kWh-electricity/kg-H_2 and 1.22 BTUs-NG/BTU-H_2, considerably more than in conventional steam reforming.

Several schemes for disposing or using rather than venting the CO_2 have been proposed. Blok et al. (1997) analyze a scheme whereby 70% of the vented CO_2 (the concentrated stream from the separation plant) is compressed and injected back into the depleted gas field, perhaps to enhance recovery of the last bits of gas. There is a small additional cost for compression, and some cost to transport the CO_2, which of course is minimized by having a short transport distance. Blok et al. (1997) find that CO_2 injection to enhance gas recovery adds on the order of $0.10/gJ to the cost of hydrogen. Steinberg (1998) proposes thermal decomposition of methane to C + H_2, sequestration of the carbon, and reaction of the hydrogen with CO_2 from a coal-fired power plant to produce methanol for motor vehicles.

The input/output estimates of Katofsky (Blok et al., 1997) for the year 1994 are used, with slight efficiency improvements over time.

Feedstock and process energy use of coal-to-synthetic crude oil plants

Coal can be liquefied to produce a synthetic crude oil, which then can be refined into conventional petroleum products. To represent this process in the LEM, data from South Africa was used, which is the world's largest producer of coal-based synthetic liquid fuels. South Africa's coal-to-liquid plants consume almost 20% of the country's coal output, and produce more than 25% of the total liquid fuel output (EIA, *International Energy Outlook 1999*, 1999).

The South African Department of Minerals and Energy (DME, 2001) reports energy balances for the "liquefaction" energy sector: 624.7 EJ of coal and 71.8 EJ of natural gas produced 309.3 EJ of synthetic crude oil. This gives an output/input energy ratio of 44.4%. I assume a slightly higher value of 46%.

I calculate upstream emissions of from the coal-to-oil fuel cycle as follows:

$$GHG_{syncrude} = GHG_{coal/crude} + GHG_{syncrude} + GHG_{products}$$

$$GHG_{coal/crude} = GHG_{coal} \cdot Q_{syncrude} \cdot Q_{products} \cdot (1 + FL)$$

$$GHG_{syncrude} = (1 + FL) \cdot \left(GHGCH_{syncrude} + \sum_{E} Q_{E} \cdot EF_{E} \right)$$

$$GHG_{products} = GHG_{product-refining} + GHG_{product-marketing} + GHG_{product-dispensing} \qquad \text{eq. 62}$$

$GHG_{syncrude}$ = CO_2-equivalent emissions from the coal-to-synthetic petroleum upstream fuel cycle (g-CO_2-equivalent / 10^6-BTU-net-petroleum-product-output).

$GHG_{coal/crude}$ = CO_2-equivalent emissions from coal recovery and coal delivery to the coal liquefaction (synthetic crude oil) plant (g-CO_2-equivalent / 10^6-BTU-net-petroleum-product-output).

$GHG_{syncrude}$ = CO_2-equivalent emissions from the coal liquefaction (synthetic crude oil) plant (g-CO_2-equivalent / 10^6-BTU-net-petroleum-product-output).

$GHG_{products}$ = CO_2-equivalent emissions from the production, marketing, and dispensing of petroleum products derived from synthetic crude oil (g-CO_2-equivalent / 10^6-BTU-net-petroleum-product-output) (the LEM uses values calculated for petroleum products derived from natural crude oil).

GHG_{coal} = CO_2-equivalent emissions from coal recovery and coal delivery to the coal liquefaction plant (g-CO_2-equivalent / 10^6-BTU-coal-delivered) (the LEM uses values for the typical coal-to-power plant process).

$Q_{syncrude}$ = BTUs of coal consumed per BTU of synthetic crude produced (data from DME, 2001).

$Q_{products}$ = BTUs of oil consumed per BTU of product produced (calculated as the ratio of BTU / ton-crude-oil to BTU / ton-petroleum product [gasoline or diesel]).

FL = fraction of fuel production lost due to evaporation or spillage (Appendix B of DeLuchi [1993], and updates thereto in this report).

$GHGCH_{syncrude}$ = CO_2-equivalent emissions from the lifecycle of chemicals used by coal liquefaction plants (g / 10^6-BTU-synthetic crude) (see discussion elsewhere).

Q_E = BTUs of energy source E used to make a unit of synthetic crude C (data from DME, 2001).

EF_E = the fuel cycle emission factor for energy source E (g-CO_2-equivalent/BTU-E; e.g., grams per BTU natural gas) (Table A.2 of DeLuchi [1993], and updates thereto in this report).

Feedstock and process energy use of corn-to-ethanol plants

Fuel ethanol can be produced by dry milling or by wet milling. As regards the estimation of GHG emissions, dry-mill plants differ from wet-mill plants in several key respects, and as a result it is important to determine at the outset how much future incremental ethanol supply will come from dry mills, and how much will come from wet mills. I argue that most future incremental supply will come from dry mills.

Dry-mill plants produce ethanol (about 2.7 gallons/bushel), and distillers' dried grains and solubles (DDGS) as a byproduct. Wet-mill plants produce corn oil, corn gluten meal, corn gluten feed, and, from the starch of corn, high-fructose corn syrup *or* ethanol (at about 2.5 gallons/bushel). Note that not only do wet-mill plants produce more products than do dry-mill plants; they produce ethanol *optionally,* whereas dry-mill plants do not. This means that dry-mill plants are built expressly to supply ethanol, and would not be built were there no anticipated demand for the ethanol, whereas wet-mill plants typically are built to supply other products, and in many if not most cases would be built regardless of the market for ethanol (Madson, personal communication, 1997).

Now, in 1992, wet mill plants did produce 872.0 million gallons of fuel ethanol, whereas dry mill plants produced only 174.2 million gallons (Bureau of the Census, *1992 Census of Manufactures, Industrial Organic Chemicals*, 1995). However, much of the wet mill capacity was put in place in the 1980s in order to produce high-fructose corn-syrup to replace sucrose in soft drinks (Madson, 1997). Moreover, over the past decade or so, as demand for fuel ethanol has increased roughly fourfold (ERS, *Feed Situation and Outlook Yearbook*, 1997), the majority of new ethanol plants have been dry mills (Madson, personal communication, 1997) -- probably because, as noted above, dry mill plants are built specifically to supply ethanol, whereas wet mill plants are built mainly to supply the other products (corn oil, corn meal, corn gluten feed, and high-fructose corn syrup). It therefore seems plausible that any increase in demand for ethanol will be supplied mainly by new dry mills, and for this reason, only dry-mill production are formally analyzed in the GHG emissions model.

Still, there is no doubt that wet mills will supply at least some of a large increase in demand for ethanol, because in response to an increase in demand, some existing wet mills will switch from producing corn syrup to producing ethanol, and a few new wet mills might even be built. Consequently, it is important to at least sketch out the GHG effects of producing ethanol from wet mills. I do that here.

<u>Energy use at ethanol plants.</u> The energy efficiency of corn-to-ethanol plants has improved substantially over the past 15 years, and as a result new dry milling plants

use less energy per gallon of ethanol than I assumed in DeLuchi (1993). On the basis of three recent reviews, discussed next, I have made new assumptions for energy use at corn-to-ethanol plants.

Madson (1997), an industry consultant with extensive experience, has summarized the energy requirements of new plants, and projected future energy:

	Year 1997 actual		Year 2002 projected	
	BTU / gal	kWh / gal	BTU / gal	kWh / gal
wet mill	32,000	0.5 - 0.6	29,000	0.5
dry mill with DDGS drying	44,000	1.1	39,000	1.0
dry mill without DDGS drying	31,000	0.9	27,000	0.7

For wet mills, the energy consumption is that of the processes specific to ethanol production. It appears that Madson uses HHVs.

In an earlier review of the actual energy requirements of corn-to-ethanol plants, Lorenz and Morris (1995) provide similar estimates:

	Average		Best existing		State-of-the-Art	
	wet mill	dry mill	wet mill	dry mill	wet mill	dry mill
process steam (BTU / gal)	35,400	39,000	29,200	26,500	26,000	26,500
electricity (kWh / gal)	2.07	1.20	1.05	0.60	0.90	0.60
bulk transport (BTU / gal)	1,330	1,330	1,100	1,100	800	800
other (BTU / gal)	1,450	1,450	1,282	1,282	1,050	1,050

Their estimates result in an average overall energy use of 0.60 BTU/BTU-ethanol, and a state-of-the-art energy use of 0.40 BTU / BTU-ethanol. (They apparently use lower heating values.) This is similar to the estimate of Conway et al. (1994) that efficient dry-mill and wet-mill corn-to-ethanol plants consume 0.50 BTU-coal per BTU ethanol produced. These energy-use requirements generally are lower than those of Table K.11 of Appendix K, supporting the contention of Lorenz and Morris (1995) and Madson (1997) that ethanol plants have become more efficient. Lorenz and Morris, and Madson, also believe that the efficiency will continue to improve.

Lastly, Shapouri et al. (2002) report the results of a year-2001 survey of energy and feedstock requirements of current dry-mill and wet-mill plants. The plants covered in the survey account for 65% of the industry's ethanol production capacity. The results are:

	BTU / gal (HHV)	kWh / gal	gal / bu
wet mill	51,060	n.e.	2.68
dry mill	36,000	1.09	2.64

For input to the GHG emissions model, I convert the estimates above from BTU / gal to physical units / gal. For example, the estimates of Lorenz and Morris (1995) convert to:

	Average		State-of-the-Art	
	wet mill	dry mill	wet mill	dry mill
process steam (lbs-coal / gal-ethanol)	3.54	3.90	2.60	2.65
electricity (kWh / gal-ethanol)	2.07	1.20	0.90	0.60
bulk transport (gal diesel / gal-ethanol)	0.010	0.010	0.006	0.006
other (lbs-coal / gal-ethanol)	0.15	0.15	0.11	0.11

My assumptions, shown in Table 17, are based on the data cited above. The % / change per year is picked so that by 2015 the resultant energy-use values approach those estimated for the more efficient technologies[33].

In the previous model, it was assumed that coal supplied 100% of the thermal energy at dry mill plants. However, environmental regulations and in some cases straight economics now favor natural gas over coal, with the result that most new dry mill plants use natural gas (Madson, personal communication, 1997). Therefore, the mix of fossil fuels used to provide thermal energy at dry mill plants has been changed from 100% coal to mainly natural gas (Table 17). This results in a 5% decrease in fuel cycle CO_2-equivalent emissions.

Finally, chemical use at corn-to-ethanol plants has been added. In note h to Table K.7 of DeLuchi (1993), reference is made to an ethanol dry mill plant designed to consume 3.7 tons / day of chemicals to treat wastewater. The plant was designed to produce about $0.16 \cdot 10^6$-gal / day, giving a chemical consumption of 23.3 tons-chemicals / 10^6-gal ethanol, or 0.047 lbs / gallon. This is the same as the chemical usage at biomass-to-ethanol plants, which seems reasonable. The GHG emissions associated with the 0.05 lbs / gal chemical consumption increase fuel cycle CO_2-equivalent emissions by about 2%.

Co-products of the corn-to-ethanol conversion process: conceptual background

Ethanol is made from the starch of the corn. The rest of the corn -- the protein, the oil, and the fiber -- is made into other products, such as distillers dried grains and solubles (DDGS). Because only a portion of the corn is made into ethanol, it is tempting to assign to ethanol, according to some allocation rule, only a portion of the total

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[33] I believe that the efficiency projections of Madson (1997) are too optimistic. The efficiency gains of the 1980s were spurred mainly by the high cost of fuel, and it appears that fuel prices will remain relatively low for a long time. The wellhead price of natural gas declined from \$3 - \$3.50 per 10^3 CF in the early to mid 1980s to under \$2 / 10^3 CF in the 1990s (EIA, *AER 1996*, 1997), and is projected to remain under \$2.50 / 10^3 CF through the year 2020 (1992 dollars) (EIA, *AEO 1998*, 1997). Moreover, increasingly stringent emission control requirements will tend to inhibit some efficiency gains.

emissions from the corn farming stage through the ethanol production stage. Unfortunately, such allocation schemes, whether according to the market value of the various products, their energy content, or some other rule, do not represent any reality we might wish to model. It is not true, for example, that if we increase production of ethanol from corn, we will get only some fraction of the emissions from corn through ethanol production. Rather, if we increase ethanol production, and hence increase corn production, we will get all of the emissions associated with corn through ethanol production. But -- and here is where consideration of the other products of the ethanol plant (call them "co-products) is relevant -- we also get fewer emissions in the co-product market, because we presumably will make less of the co-product substitutes.

Thus, as pointed out in Appendix K of DeLuchi (1993), the correct approach is conceptually simple: estimate emissions in the world with and without ethanol production. Quoting from Appendix K (pp. K-16 to K-17):

> ..the whole point of calculating greenhouse gas emissions from the manufacture and use of ethanol is to help answer the question, "Should we make ethanol from corn?" That is, we are interested in seeing what happens if we make ethanol from corn — compared, by default, to not making it, and using gasoline, or some other fuel, instead.
>
> We may, therefore, begin by saying, "If the United States endorses ethanol from corn, it will build and operate a large number of additional ethanol plants; if it does not endorse ethanol, it will not, and will make something else (probably gasoline) instead." We wish to compare these with and without scenarios: we wish to estimate whether the ethanol world produces more or less greenhouse-gas emissions than the gasoline world. In the "with" scenario, we have emissions from most or all of the ethanol production-and use-cycle. In the "without" scenario, we have all the emissions from the production and use of the work-equivalent amount of gasoline, plus the emissions from the production and use of the products (call these the "by-product substitutes") that would have been displaced by the by-products of the ethanol production process. The difference between the with and without scenarios is the result of the ethanol policy.
>
> Now, if we wish to compare emissions from the ethanol case with emissions from a baseline gasoline case, we must move the emissions associated with the by-product substitutes from the gasoline side of the ledger to the ethanol side of the ledger, by subtracting these emissions from the fuel-cycle totals for ethanol. To do this, one must know what the ethanol by-products would displace, and how much of what kind of energy would have been used to make the by-products.

This means that, in principle, it is not correct to estimate what we might call a "co-product displacement credit" by deducting or excluding some of the energy (for example, energy to dry co-products) used in the corn to ethanol process[34]. All energy

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[34] Conway et al. (1994) exclude the energy used to dry the germ, fiber, and gluten coproduct, on the grounds that such energy should not be assigned to ethanol production. Furthermore, they go on to suggest that there should be an additional credit given for the co-products, even after the energy used to dry the co-products has been excluded. However, if we follow this suggestion, we run the risk of double counting the credit, or of being internally inconsistent: one the one hand, we ignore the energy associated with making the coproducts -- as if the coproducts weren't made -- but then on the other hand count as a credit the foregone energy associated with the products that would have been displaced by the co-products (had they

and emissions from the ethanol production process must be counted and assigned to ethanol. The displacement credit should be calculated by estimating the emissions foregone in a world without the ethanol production.

The trick, of course, is to estimate what would happen in the co-product markets in the world without ethanol production. What would be produced if the co-products of the corn-ethanol process not available? How much would be produced? It is also important to know what would happen in agricultural markets if extra corn were not demanded for ethanol production.

Most analyses of the "co-product displacement credit" have assumed that the DDGS from dry-mill plants (recall from above that I consider only dry mill plants) displaces soy protein (e.g., Marland and Turhollow, 1990; Conway et al., 1994). However, the DDGS co-product is a more complete feed than is soy protein because it has more fat and fiber (Madson, personal communication, 1997; Madson states also that the DDGS protein is more digestible than is soy protein). Madson claims that DDGS is used mainly in feedlots, to fatten up cattle, and that the substitute for this is whole corn feed, not soy protein. This seems reasonable, and this analysis therefore assumes that the DDGS displaces whole corn feed. (Note that this assumption probably is favorable to the corn-ethanol energy and emissions balance, because one kg of DDGS displaces more than a kg of whole corn feed, but less than kg of soy protein.) The formal relationship is quantified below.

The effects of corn production on agricultural markets might be important, but are too complex to be modeled here. The shift in demand for corn, as a result of the extra demand in the ethanol production sector, will increase the price, which will reduce demand for corn for other uses, by an amount depending partly on the slope of the supply curve. A USDA study cited by Harding Lawson Associates (HLA, 1992) estimates that each additional 1 billion gallons of ethanol produced per year will increase the price of corn by $0.08 to $0.28/bushel (for the past 20 years the price has been in the range of $2-3/bu). However, the increase in the price of corn will increase demand for corn substitutes, and reduce the demand for complements, by amounts depending on the cross-price elasticity of demand. The overall effect on agricultural markets and ultimately greenhouse-gas emissions is not clear.

GHG emissions displaced by the DDGS co-product of dry-mill ethanol plants

The net co-product displacement emissions credit is equal to emissions from the production and transport of the corn feed displaced by the DDGS, less the emissions from the transport of the DDGS to end users. The emissions from the production and transport of the displaced corn feed depend on the amount of DDGS produced, the equivalency between DDGS and corn feed, the emission factors for corn production and transport, and other factors. Formally:

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actually been made). It is better to estimate emissions in the world with and without the ethanol plant and all its actual coproducts.

$$GHGDD = GHGDC - GHGTD$$

$$GHGDC = DDGS \cdot \frac{CD}{BU} \cdot NDF \cdot GHGC$$

$$GHGTD = EF_{DT} \cdot \frac{EI_{DT}}{2000} \cdot DDGS$$

$$DDGS = DDGS^{*} \cdot YE \cdot \frac{(1+FL)}{D_E}$$

$$GHGC = \sum_{S} GHGC_S$$

$$\text{assume}: EF_{DT} = EF_{CT};\ EF_{CT} = \frac{EF^{*}_{CT}}{ER_{CT}}$$

$$EI_{DT} = EI^{*}_{DT} \cdot MD;\ \text{assume} = EI^{*}_{DT} = EI^{*}_{CT};\ \text{then}$$

$$EI_{DT} = EI^{*}_{CT} \cdot MC \cdot \frac{MD}{MC} = EI_{CT} \cdot RD1 \qquad \text{eq. 63}$$

where:

GHGDD = net CO_2-equivalent GHG emissions displaced by the production of DDGS, per energy unit of ethanol made available to end users (g / 10^6-BTU).

GHGDC = CO_2-equivalent GHG emissions from the production and transport of the corn feed displaced by the DDGS, per energy unit of ethanol made available to end users (g / 10^6-BTU).

GHGDT = CO_2-equivalent GHG emissions from the transport of the DDGS to end users, per energy unit of ethanol made available to end users (g / 10^6-BTU).

DDGS =lbs of DDGS produced per 10^6-BTU of ethanol made available to end users.

CD = lbs of shelled corn (at 56 lbs / bu) equivalent as feed to one lb of DDGS (see discussion below).

BU = lbs of shelled corn per bushel (56; USDA, *Agricultural Statistics 1997,* 1997)

NDF = the net displacement fraction: of the total lbs of DDGS produced, the fraction that actually displaces existing or “old” feed (so that 1-NDF is the fraction that supplies new demand) (see discussion below).

GHGC = CO_2-equivalent greenhouse-gas emissions from the production and transport of the corn feed displaced by the DDGS (g-CO_2-equivalent / bu-corn).

EF_{DT} = GHG emissions from the use of fuel to transport of DDGS (g / 10^6-BTU-fuel; assume the same as that calculated for transporting corn).

EI_{DT} = fuel use per ton of DDGS transported (10^6-BTU-fuel / ton)

2000 = lbs per ton

DDGS* = lbs dried DDGS per bushel of corn processed at ethanol plant (see discussion below)

YE = bushels per gallon of ethanol (Table 17).

FL = fraction of fuel production lost due to evaporation or spillage (Appendix B of DeLuchi [1993], and updates thereto in this report).

D_E = the heating value of ethanol (10^6-BTU / gal).

$GHGC_S$ = CO_2-equivalent GHG emissions from source S in the production and transport of the corn feed displaced by the DDGS (g-CO_2-equivalent / bu-corn) (g-CO_2-equivalent / bu-corn; calculated by the GHG model).

S = sources of GHG emissions from the production and transport of corn feed displaced by DDGS: corn farming, manufacture of agricultural chemicals, use of chemicals, and corn transport.

EF_{CT} = GHG emissions from the use of fuel to transport of corn (g / 10^6-BTU-fuel).

EF^*_{CT} = GHG emissions from the use of fuel to transport of corn, per energy unit of ethanol made available (g / 10^6-BTU-ethanol; calculated by the GHG model).

ER_{CT} = energy use ratio for corn transport (10^6-BTU-fuel / 10^6-BTU-ethanol; calculated by the GHG model)

EI^*_{DT} = fuel use per ton-mile of DDGS transport (10^6-BTU-fuel / ton-mile; assume same as that calculated for transporting corn from farm to ethanol plant, EI^*_{CT}).

EI^*_{CT} = fuel use per ton-mile of corn transported (10^6-BTUs-fuel / ton-mile).

MD = the distance from the ethanol plant to the DDGS end user (mi).

MC = the distance from the corn field to the ethanol plant (mi).

EI_{CT} = fuel use per ton of corn transported (10^6-BTUs-fuel / ton; calculated by the GHG model).

RD1 = MD / MC = the distance from the ethanol plant to the DDGS end user, relative to the distance from the corn field to the ethanol plant (assume 1.00; i.e., the same distance).

CD: lbs of shelled corn (at 56 lbs / bu) equivalent as feed to one lb of DDGS. According to industry consultant Madson (1997), 1.0 lbs of DDGS, plus 0.4 lbs of roughage such as straw, replace 1.4 lbs of bone-dry whole-corn feed[35]. To account for the undoubtedly minor amount of GHG emissions associated with the provision of the 0.4 lbs of roughage, without having to explicitly include roughage in the GHG model, I will assume that 1.05 lbs of DDGS and 0 lbs of roughage are equivalent to 1.4 lbs of bone-dry whole corn feed, or 1.4 / 0.85 = 1.65 lbs of 15% moisture corn (which is the basis of 56 lbs / bushel metric used in this analysis). Thus, one lb of DDGS is equivalent as feed to 1.65 / 1.05 = 1.57 lbs of corn (at 56 lbs / bu).

NDF: the net displacement fraction. This is the fraction, of the total lbs of DDGS produced, that actually displaces existing or “old” feed, such that 1-NDF is the fraction that supplies new demand. Not all of the byproduct will displace feed previously produced from other sources; some will be additional, new supply that will satisfy an increased demand for feed. As shown in Figure 3 , the byproduct DDGS will shift the supply curve out, from S* to S: at any given price, the amount of feed supplied will increase by the amount of DDGS marketed as a byproduct of ethanol production. But in general, the equilibrium quantity of feed consumed will not increase by the amount of DDGS made available to the market, because the equilibrium price of feed will decline[36]. Hence, some portion of the marketed byproduct DDGS will displace marginal high-cost supply, and some will satisfy additional demand stimulated by the lower price.

The balance between displacement and additional supply depends on the slope of the supply and demand curves. Consider the extreme or boundary conditions. If demand is completely inelastic, there will be no change in consumption, and all of the marketed byproduct DDGS from ethanol plants will displace feed produced from other sources. On the other hand, if demand is completely elastic, there will be no change in price, and all of the byproduct DDGS will be additional consumption. Most likely, reality will lie between these two extremes, as indicated in Figure 3 . In Figure 3 , the amount of byproduct DDGS marketed is equal to Q-Q’. As a result of the shift in the supply curve from S* to S, the price declines from P* to P, and the equilibrium quantity increases from Q* to Q. The difference between the total byproduct quantity marketed, Q-Q’, and the equilibrium increase in quantity, Q-Q*, is the amount of previously produced [high-cost] feed displaced, Q*-Q’. In Figure 3 , the amount of displaced feed is about half of the total amount of DDGS produced.

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[35] For those interested in the equivalency of DDGS to soybeans: Marland and Turhollow (1990) estimated that one lb of DDGS is equivalent to 0.721 lbs soybean meal. Madson (1997) says that the equivalency has increased slightly, to 0.75 to 0.80, as the protein content of DDGS has increased as more of the carbohydrate is converted to ethanol.

[36] HLA (1992) note that “an increase in ethanol production would increase the quantity of corn by-products sold on the market,” and that “this increase of corn-byproducts could lower the price of these byproducts..” (p. 74).

The amount of feed displaced, Q*-Q', can be estimated as:

$$NDF \cdot (Q\text{-}Q')$$

where:

NDF is the ratio Q*-Q' to Q-Q'

Thus, if demand is relatively inelastic, NDF is close to 1; if demand is relatively elastic, NDF is close to 0. We wish to know, then, whether demand for feed elastic or inelastic. The Economic Research Service asserts that "food and industrial demand for feed grain is largely inelastic, with little or no substitution possibilities" (ERS, *Feed Situation and Outlook Yearbook*, 1997). On the other hand, the same ERS report, and the World Agricultural Outlook Board (1997) projections, indicate that demand for feed grains is fairly sensitive to price. I will assume that demand is only moderately elastic.

Theoretically, however, the story does not end here, because any net expansion of feed consumption -- might itself displace production of other kinds of food. In general, a reduction in the price of feed will reduce consumption of feed substitutes, by an amount depending on the cross-price elasticity of demand. Allowing qualitatively for such effects, and assuming only a moderately elastic demand, I assume that NDF = 0.75; that is, that 75% of the byproduct DDGS displaces previously produced feed (or feed substitutes), and that 25% satisfies additional consumption with no further substitution.

DDGS*: lbs of DDGS per bushel of corn processed. Data cited in Tables K.7 and K.8 of DeLuchi (1993) indicate 3,000 to 3,500 tons DDGS / 10^6-gal, or about 15-18 lbs / bu-corn, depending mainly on the ethanol yield. (The higher the ethanol yield per bushel, the lower the DDGS yield per bushel.) Industry consultant Madson (personal communication, 1997) confirms this range: today, the DDGS yield ranges from 16 lbs / bu, at 2.6 gal / bu, to 14 lbs / bu at 2.78 gal / bu. (The greater the ethanol output, the less the DDGS output.) The following formula reproduces the figures reported by Madson, and are used in the model:

$$DDGSY = 42 - 10 \cdot YE$$

where:

DDGSY = the DDGS yield (lbs / bu).
YE = the ethanol yield (gal / bu; Table 17).

Use of fusel oil as a boiler fuel The corn-to-ethanol conversion process produces small amounts of aldehydes and higher alcohols. In DeLuchi (1993), this so-called fusel oil was used as a supplementary boiler fuel. However, according to industry consultant Madson (personal communication, 1997), the fusel oil is left in the fuel alcohol, and is included in the gallon / bushel yield figures reported by Madson and others. Therefore, two changes have been made to the model:

1) I have added a "yes / no" switch to the calculation of the fusel oil credit: "yes" means that the fusel oil is used as a boiler fuel, "no" means it is used as product. The switch now is set to "no", with the result that fuel cycle GHG emissions increase by about 2% over the estimates of DeLuchi (1991). However, the previous model did not reduce the ethanol yield by the amount of fusel oil diverted to boilers. The present model does, and now the difference between using fusel oil in the boiler and keeping it in the methanol product is only 1% of fuel cycle GHG emissions.
2) If the switch is set to "yes," so that the fusel oil is used as a boiler fuel, then the amount of fusel oil used is deducted from the reported gal / bu yield. The fusel oil is assumed to be a mix of propanol and butanol.

Note that if in the model fusel oil is designated to be a boiler fuel, it is treated as an as a 1:1 BTU-for-BTU substitute for gas or coal at the ethanol plant, not as a marketable co-product.

Use of ammonium sulfate as a fertilizer In the previous model, ammonia was assumed to be used to scrub sulfur from coal and produce ammonium sulfate, which then was used as a fertilizer for corn. It turns out, however, that limestone is used, and that the resultant sludge is disposed of (Madson, personal communication, 1997). Because of this, and because in any event I neglected to include the emissions from the manufacture of ammonia (which emissions probably would cancel the emissions saved as a result of using the ammonium sulfate as fertilizer), I have removed the ammonium sulfate credit from the model. To account for emissions from use of limestone to scrub sulfur, I have added to emissions from coal-fired industrial boilers the same limestone-related emissions estimated for coal-fired utility boilers (see Appendix D of DeLuchi [1993]).

The co-product displacement credit for wet-mill plants

It will be apparent from the discussion above that the proper way to analyze GHG emissions from a corn / wet-mill / ethanol fuel cycle depends in the first instance on whether the wet mill plant would have been built had there been no ethanol policy. If a wet mill plant is built specifically to supply ethanol, and would not have been built had there been no incremental demand for ethanol, then fuel cycle GHG emissions are analyzed as in the dry-mill case: one first estimates total "gross" emissions from the production, transport, and processing of *all* the corn input to the wet mill plant, and then deducts the GHG emissions that would have been generated by the production displaced by the co-products (corn meal, corn oil, and corn gluten) of the wet-mill process. (Because there are several co-products, the analysis of the co-product displacement credit is complicated.) In this case, one starts with total emissions from the processing of all corn input because one would not have processed any of the corn had there been no ethanol policy.

If, however, an ethanol policy induces an existing wet-mill plant (one that would exist, or would have been built, and would be in operation regardless of the ethanol policy) to switch starch conversion from corn syrup to ethanol, then the GHG emissions attributable to the ethanol policy are equal to:

1) the emissions from the starch-to-ethanol conversion process in the wet -mill plant, plus the emissions from transporting and using the ethanol product, *minus:*
2) the emissions from the now abandoned starch-to-corn-syrup conversion step in the wet-mill plants and the emissions from transporting and using the corn syrup, *plus:*
3) the emissions from the production, transport, and use of whatever is made to replace the corn syrup formerly produced.

If, to a first approximation, the emissions from the conversion of starch to ethanol (in #1) are canceled by the emissions foregone from the conversion of starch to corn syrup (in #2), and if emissions foregone from the transport and use of the corn syrup (in #2) at least cancel the emissions from the transport and use of whatever replaces the foregone corn syrup (#3), then the net GHG emissions attributable to the ethanol policy are the emissions from transport and end use of ethanol, plus the emissions from the production of stuff to replace the corn syrup formerly made. This makes sense: if an ethanol policy has no effect on the use of corn, and no effect on the output of wet mill plants other than to switch starch from corn syrup to ethanol, then the only things changed in the ethanol-policy world are the transport and use of the ethanol, and the production of whatever makes up for the loss of corn syrup. These emissions will total to much less than the emissions from the corn / dry-mill / ethanol process, because there are no net emissions from corn farming or ethanol production. (To put it yet another way, ethanol in this scenario is almost a “free” byproduct.) I estimate that fuel cycle GHG emissions (including emissions from end use, but not from vehicle manufacture) from switching wet mill plants to ethanol production are on the order of 100-150 g-CO_2-equivalent / mi -- well less than half of the emissions in the dry mill case.

Co-products of wood-to-alcohol production

Von Sivers and Zacchi (1996) state that wood-to-ethanol plants produce marketable chemicals, lignin fuel, and electricity, in addition to ethanol, and estimate that the $/gallon-ethanol value of these co-products is as much as 50% of the $/gallon production cost of ethanol. However, in the wood-to-ethanol process assumed here (Table 17; Lynd, 1996a), the lignin is used within the plant as a boiler fuel, and there is no significant chemical co-product. As discussed elsewhere, the excess power produced is given an appropriate GHG emissions credit.

In the absence of data to the contrary, we assume that there are no significant co-products from wood-to-methanol plants either.

Electricity displaced by electricity exported from wood-to-ethanol and grass-to-ethanol plants

Mix of fuels displaced. The GHG model now requires that you to specify the mix of electricity that is displaced by the excess power generated by wood-to-ethanol plants[37]. (In the previous version, the model assumed that the U.S. average power mix was displaced.) The excess power made available to the market will displace electricity generated at a high variable cost. Compared to the national average mix, the high-variable-cost mix has a relatively large amount of gas.

To quantify the mix of electricity-generation fuels displaced by electricity exported from ethanol plants, the U. S. DOE ran the electricity module of the National Energy Modeling System (NEMS) with and without electricity from ethanol plants, and reported the change in the dispatch of power plants (Conti, 1999). The DOE assumed that marginal cost of power from ethanol plants is zero, so that they would run whenever they are available. Also, DOE didn't allow for any other expansion of capacity, or any change in demand, even though these would occur in reality, because they could not be modeled reliably given the tiny change in supply due to power from ethanol plants (Conti, 1999).

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[37] An ethanol plant can generate more power than it needs internally by burning lignin , the component of the wood that cannot be converted easily to ethanol. Under any conceivable regime of electricity prices, it probably will be more economic for wood-ethanol producers to buy electricity- generation equipment, burn the unusable lignin to produce power, and sell the excess power to the grid, rather than to buy electricity from the grid (and perhaps attempt to find some other market for the lignin). I have assumed therefore that ethanol producers will in fact burn lignin to produce power, and that all of the excess power that *can* be produced (which is what Lynd [1996a], cited in the notes to Table 17, actually estimate) will in fact be sold to the grid.

The DOE analysis indicates that every kWh of electricity displaced by a kWh of power from ethanol plants is distributed as follows (generation rather than fuel-input basis):

	2010	2015	2020
Coal	100%	53%	12%
Gas	0%	40%	82%
Oil	0%	7%	6%
Hydro	0%	0%	0%
Nuclear	0%	0%	0%
Renewables	0%	0%	0%

The DOE model results also indicate that the introduction of ethanol results in a slight increase in generating efficiency for most fuel types. However, the effect is small, and so I ignore it.

Given this, I assume that oil's share of the displaced power increases 0.5% per year from 0% in 2010 to 8% in 2026 (and remains at 8% thereafter), that the share of coal declines 9% per year (absolute terms) from 100% in 2010 to 10% in 2020 (and remains at 10% thereafter), and that the balance is natural gas. I assume that the gas is split 90% turbines, 10% boilers.

Formerly, I assumed that electricity from ethanol plants displaced the national average mix of power. The change to a displaced mix with a higher share of natural gas reduces the displacement credit by a little less than 10%.

Net displacement. The model now asks you to specify the fraction of the byproduct power that actually displaces generated power. Not all of the byproduct power will displace power previously generated from other sources; some will be additional, new supply that satisfies an increased demand. The effect is illustrated in Figure 3 , and is discussed above in regards to the DDGS co-product of ethanol production from corn. As discussed there, the balance between displacement and additional supply depends in the first instance on the slope of the demand curve. If demand is relatively inelastic, the net displacement factor NDF (the ratio of Q*-Q' to Q-Q') is close to 1; if demand is relatively elastic, NDF is close to 0. I will assume that demand is moderately elastic, an assumption consistent with the EIA's *Analysis of Electricity Prices in a Competitive Environment* (EIA, 1997), and the finding of Silk and Joutz (1997) that a 1% decrease in electricity price causes an 0.6% increase in electricity consumption in the long run. (In Figure 3, a 1% decrease in price is associated with approximately a 0.6% increase in consumption.)

Theoretically, however, the story does not end here, because any additional electricity consumption most likely will affect energy use in other sectors. For example, some of the additional consumption of electricity might reflect a switch, in the long run, from gas to electricity for heating or cooking. In this case, the exported power displaces gas indirectly, rather than previously generated power directly. In general, the reduction of the price of electricity will reduce consumption of substitutes for electricity, by an amount by the long-run cross-price elasticity of demand[38]. However, if the use of substitutes, such as natural gas, is determined more by availability (of gas infrastructure) than by price, as Silk and Joutz (1997) maintain, then a change in price of electricity is not likely to have much effect on the use of natural gas.

Therefore, allowing for these probably minor second-order effects, I assume that $NDF_{power} = 0.75$; that is, that 75% of the exported power displaces previously generated high-cost power (or the equivalent amount of power substitutes) and 25% satisfies net additional consumption, in the general equilibrium[39].

Co-products of the soy-diesel production process

The soy-diesel manufacturing process produces substantial amounts of glycerine and soy meal, along with fuel (see Appendix A to this report). Ahmed et al. (1994) assume that the soy meal is used in place of barley as an animal feed (see Appendix A to this report). The situation with glycerine is more complicated, because there are many of sources of glycerine, and hundreds of uses (Economic Research Service, 1993, 1996).

It is difficult to estimate the GHG emissions displaced by the co-products of the biodiesel production process for two reasons. First, as just noted, there are two major co-products, one of which, glycerine, can be made from a variety of sources, and is used in many applications. Also, it is not clear that the other major co-product, soy meal, necessarily replaces barley feed, as Ahmed et al. (1994) assume. Second, the extra demand for soy oil, to be made into biodiesel, will affect the markets for a variety of farm products. Raneses et al. (1996) use the Food and Agriculture Policy Simulator (FAPSIM) to track the economic impacts of biodiesel production over a broad range of agricultural commodities. They simulate the production of biodiesel by shifting the demand for soybean oil. This shift increases the price of soy oil; the price increase, in turn, causes a decrease in demand for soy oil in other uses, but an increase in demand for raw soybeans used by processors, because of the greater profitability brought about by the higher price of the oil. As more soybeans are crushed, more soy meal is

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[38] In the short run, of course, there is little opportunity for end users of, say, natural gas, to switch to electricity, because it is not economical to replace natural-gas heaters, dryers, stoves, and so on before the end of their useful life. Rather, in theory, one would expect that permanently lower electricity prices might induce some home builders to equip houses for electric rather than gas appliances. But as Silk and Joutz (1997) note, we must back up one more step, because a builder can choose between gas and electricity only if gas is locally available. Silk and Joutz (1997) maintain that “availability, not price, has caused shifts between natural gas and electricity in new houses” (p. 498). In their own analysis of the reverse effect (of the price of alternatives on demand for electricity), they find that a 1% increase in the price of fuel oil leads to only a 0.25% increase in electricity consumption.

[39] This exposition assumes that the market for electricity is classically competitive. If it is not -- if price and quantity are determined by mechanisms other than the free market, then Figure 3 does not apply.

produced, and as a result the price of soy meal falls. Because meal is a major input to the production of livestock, the decline in the price of soy meal leads to an increase in production of livestock. Also, the lower price of soy meal causes livestock producers to feed more soy meal and less corn; as a result, corn production declines.

All of this is too complicated to model here. Instead, as a crude, first-cut approximation of the emissions displaced by soy diesel co-products, I rely on the estimates of Ahmed et al. (1994; Appendix A to this report) of the energy required to make the displaced products displaced by the soy diesel co-products. Formally:

$$GHGD = \frac{ED \cdot HHVF}{D_B} \cdot NDF \cdot EE \qquad \text{eq. 64}$$

where:

GHGD = greenhouse gas emissions from products displaced by the co-products of the biodiesel process (g-CO_2-equivalent / 10^6-BTU biodiesel produced).

ED = energy required to make products displaced by biodiesel co-products (Ahmed et al. [1994] estimate about 98,000 BTUs / gal-biodiesel, LHV [see Appendix A to this report]).

HHVF = conversion from LHV basis of Ahmed et al. (1994) to HHV basis of this report (assume 1.05).

D_B = the heating value of biodiesel fuel (128,200 BTUs / gal, based on data from EPA, 2002a [see also Appendix A]).

NDF = of total co-product output, the fraction that actually displaces existing products (assumed to be 0.75; see discussion above in regards to Figure 3).

EE = fuel cycle CO_2-equivalent GHG emissions per unit of energy used to make products displaced by biodiesel-production co-products (150,000 g / 10^6-BTU is assumed, including emissions from non-combustion sources such as fertilizer use).

The estimated displaced emissions, GHGD, or deducted from “gross” (pre-credit) emissions from the biodiesel production stage.

The assumptions shown here result in a displacement “credit” that is a little more than half of the gross (pre-credit) emissions from the biodiesel production cycle excluding end-use combustion in vehicles (feedstock recovery through product distribution). However, the parameters ED, NDF, and EE are quite uncertain, and different assumptions can lead to significantly different results.

Diesel fuel produced from F-T conversion of natural gas

A high-quality, clean-burning, diesel like fuel can be made from natural gas, via the F-T conversion process. The Sasol Slurry-Phase Distillage Process (Sasol, n.d.) has three steps. First, natural gas is reformed into a synthesis gas composed of carbon monoxide and hydrogen. In the second step, the synthesis gas is converted into waxy hydrocarbons, with a small amount of light hydrocarbons and water. Finally, the waxy hydrocarbons are upgraded into middle distillate fuels. See Knott [1997] for a recent review of projects.) Sasol states that their conversion process is about 60% efficient on a lower-heating-value basis.

Stork (1997) of Argonne National Lab has provided a complete efficiency and carbon-balance analysis of an F-T diesel plant, based on the work of Choi et al. (1997). The inputs and outputs are as follows:

Inputs	
natural gas	$0.412 \cdot 10^9$ SCF / day
n-butane	14,280 gal / day
Outputs	
gasoline	$0.714 \cdot 10^6$ gal / day ($2.02 \cdot 10^6$ kg / day)
distillate	$1.100 \cdot 10^6$ gal / day ($3.21 \cdot 10^6$ kg / day)
propane	$0.071 \cdot 10^6$ gal / day ($0.14 \cdot 10^6$ kg / day)
electricity	$592 \cdot 10^3$ kWh / day

The natural gas input is allocated to propane, gasoline and distillate in proportion to the mass output of each. This results in 224 SCF per gallon of diesel fuel. By comparison, the EIA's *International Energy Outlook 1999* [1999] reports that Chevron and Sasol [the South African oil company] estimate it will take 238 SCF to produce a gallon of middle distillates. Given this, my assumptions are shown in Table 17. I assume that the small amount of output electricity is sold to the grid, and to some extent (75%) displaces existing power generation. This electricity-co-product emissions displacement credit is relatively small (on the order of 1% of fuel cycle energy use).

I assume that F-T diesel plants are located in the same places as the plants that produce methanol from natural gas. Hence, the natural-gas feedstock part of the F-T diesel fuel cycle is modeled to be the same as the natural-gas feedstock part of the NG-to-methanol fuel cycle, and the fuel-distribution part of the F-T diesel fuel cycle is modeled to be the same as the fuel-distribution part of the NG-to-methanol fuel cycle.

Hydrogen produced from biomass: process energy requirements

I assume that the energy requirements are 1.30 BTUs-wood / BTU hydrogen and 0.065 BTU-electricity / BTU-hydrogen, in the biomass-to-hydrogen path, partly on the basis of data in Katofsky (1993).

Hydrogen produced from water: energy efficiency of electrolysis

The energy efficiency of water electrolyzers is projected using Eq. 3 (see above), with the following parameter values:

$V_U = 0.93$
$V_L = 0.72$
$V_{TB} = 0.76$
$k = 0.150$
$T_B = 1996$

Estimates of V_U and V_{TB} are based on the statements by Ogden and Nitsch (1993), Rosen and Scott (1998), and Kreuter and Hofmann (1998). In 1993, Ogden and Nitsch (1993) stated that then-present electrolyzers were 73% efficient (HHV), and that "future" electrolyzers will be 90% efficient. In 1998, Rosen and Scott (1998) state that current-technology water electrolysis is 77% efficient, and that advanced-technology water electrolysis is 92% efficient. Kreuter and Hofmann (1998) write that "a lot of emphasis has been put in the past on improving the efficiency [and] these efforts have resulted in a gross increase in the efficiency from 65% to 85%" (p. 665). Berry (1996) assumes 68% (LHV) for polymer membrane electrolyzers, and 93% (LHV) solid-oxide electrolyzers.

Source of LPG

The model reports grams-emitted per 10^6-BTU-LPG, by stage of the fuel cycle, for three LPG pathways: LPG from NGL plants, LPG from refineries, and LPG from a combination of refineries and NGL plants. It reports g / mi results only for the combined pathway, which I assume draws from NGL plants and refineries in proportion to their total annual national output of propane and butane. However, a recent report by the U. S. General Accounting Office (GAO) (1998) notes that the EIA believes that an increase in propane production due to an increase in transportation demand for LPG motor fuel probably will be supplied by refineries. The GAO (1998) goes on to state that "according to EIA analysts, the effect [of the increased demand by the transportation sector] would not be sufficient to cause natural gas processing plants to increase their production because overall natural gas production would likely not be affected" (U.S. GAO, 1998, p. 8).

If at the margin LPG for transportation does indeed come exclusively from refineries, then total fuel cycle CO_2-equivalent emissions will be higher than estimated here for the combined refinery / NGL-plant pathway, because CO_2-equivalent emissions from the "upstream" refinery pathway, from feedstock production up to but not including vehicle end use, are 35%-40% higher than emissions from the upstream combined refinery / NGL-plant pathway. This increase in upstream emissions increases total fuel cycle CO_2-equivalent g / mi emissions (including end use) by on the order of 4%.

Emission factors for plants that produce ethanol or methanol

The emission factors for NG-to-methanol, coal-to-methanol, wood-to-methanol, and corn-to-ethanol plants have been revised on the basis of a reconsideration of my original data, and new data from Ismail and Quick (1991), Ecotraffic AB (1992), EPA (1994), Darrow (1994), and other sources (Table 18). PM, PM_{10}, and SO_2 emission factors have been added. The CH_4 exhaust emission rate from NG-to-methanol plants has been increased from 0.4 to 10 g / 10^6-BTU-NG on the basis of new data from the Texas Air Control Board (1990), Ecotraffic AB (1992), and the IPCC (1997) (Table 18). My assumptions are shown in Table 18.

Formerly, emissions from wood-to-ethanol plants were estimated by multiplying the energy content of feedstock used as a process fuel by the g / 10^6-BTU-feedstock-input emission factors for wood fluidized bed combustors. Now, NMOC, NO_2, CO, PM, and SO_2 emissions are estimated by multiplying the *total* 10^6-BTU-feedstock-input of the plant by *total* plant emission factors in g / 10^6-BTU-feedstock-input. These total-plant emission factors are from NREL (Riley and Schell, 1992) Emissions from grass-to-ethanol plants are estimated in the same way, also using NREL (Riley and Schell, 1992) estimates (Table 18). Emissions of CH_4 are estimated by scaling NMOC emissions by the CH_4 / NMOC ratio for wood-waste combustion (see below), and emissions of N_2O are estimated by scaling the NO_X emissions by the N_2O / NO_X ratio for wood-waste combustion.

Emission factors for plants that produce hydrogen from natural gas

Spath and Mann (2001) use EPA's AP-42 emission factors to estimate emissions of about 0.6 g-CO / 10^6-BTU-NG, 0.2 g-PM / 10^6-BTU-NG, and 6.7 g-NO_X / 10^6 BTU. They also assume zero CH_4, NMOCs, and N_2O. Their estimates for CO and NO_X are several-fold lower than our estimates of CO and NO_X emissions from NG-to-methanol plants. We assume that emissions from NG-to-hydrogen plants (in g / BTU-NG-feed) are one-half of those from NG-to-methanol plants, on account of the less intensive processing required to make hydrogen.

Emission factors for wood-waste combustion in boilers

See the section on emissions from industrial boilers.

PRODUCTION OF CORN, SOYBEANS, TREES, AND GRASSES

In this section, earlier estimates of the energy and chemical inputs to the production of energy crops have been revised for corn for ethanol, soy for biodiesel, and trees and grasses for ethanol. (Note that perennial grasses as feedstock for ethanol production have been added.) I discuss first the inputs to corn and soybean farming. Because the amount of energy and chemical input per bushel of soybean or corn varies considerably from place to place, it is important to determine at the outset if it is possible to identify the marginal corn and soybean production for energy. I believe that this is not possible, and that it is acceptable to estimate inputs on the basis of national average trends.

Where will the marginal corn come from?

It is difficult to determine where the corn used for ethanol might come from. Historical data on acres of corn harvested are not of much help. From 1975 to 1994 there were slight regional shifts in production, mainly from the corn belt and southern states to the plain states (Economic Research Service, *Feed Situation and Outlook Yearbook*, 1995), but in 1996 at least part of this slight trend was reversed, as plantings in the south increased dramatically (see below).

The problem, of course, is that corn plantings in a particular area depend very much on local weather conditions, soil conditions, input costs, and the expected price of corn relative to that of competing crops, considerations regarding crop rotation, conservation requirements and other factors that are very difficult to predict (Economic Research Service, *Feed Situation and Outlook Yearbook*, 1995). In the past, when planting decisions were influenced heavily by Federal farm programs, it may have been easier to predict regional cropping patterns. However, the Federal Agriculture Improvement and Reform Act of 1996 removed many of the old constraints, and made farmers much more responsive to market conditions. In fact, according to the Economic Research Service (ERS, *Feed Situation and Outlook Yearbook,* 1997), the 1996 Farm Act:

> provides producers with almost complete planting flexibility by decoupling planting decisions from program payments and by eliminating annual supply control programs. Target prices and deficiency payments are eliminated and replaced by fixed contract payments that are independent from market prices...In addition to a more market-oriented commodity policy, reduced trade barriers through passage of GATT and NAFTA are leading to freer trade and closer linkage of commodity prices between domestic and world markets. Under the old program rules, acreage response largely depended on program rules and planting restrictions.

The 1996 Farm act had immediate effect on corn planting decisions. At the beginning of the 1996 season, demand and prices for corn were high, but adverse conditions in the some of the major corn-producing states in the Midwest prevented some plantings. In response the resultant sustained high prices, farmers in the smaller producing states of the South -- no longer constrained by base acreage considerations under the old farm program -- shifted much land into corn. Production records were set in Missouri, North Dakota, Louisiana, and Mississippi (ERS, *Feed Situation and Outlook Yearbook,* 1997).

These sorts of effects obviously are difficult to predict. The ERS is developing econometric models to predict the supply response of corn (ERS, *Feed Situation and Outlook Yearbook,* 1997), but long-term projections of regional planting have not been published.

Nor is it any easier to base projections of corn planting on projections for ethanol production. In the first place, total annual demand for corn for ethanol has fluctuated considerably since 1993, in response to fluctuations in the cost of corn and the price of products that compete with ethanol (ERS, *Feed Situation and Outlook Yearbook,* 1997). For example, in 1994/95, "ethanol producers were caught between higher costs for inputs and competing products that limited raising prices and suspended operations to do maintenance on their plants. Ethanol producers then found many petroleum firms had committed to MTBE when ethanol prices were not competitive as they made plans for the winter oxygenate season" (ERS, *Feed Situation and Outlook Yearbook,* 1997).

State and Federal policies regarding ethanol production also can play a role in determining in state and regional ethanol production. Again, according to the ERS (*Feed Situation and Outlook Yearbook,* 1997):

> Increased grain prices have caused four dry mill ethanol plants to close since May 1995, and some may not reopen...One of the plants closing was in North Dakota where the State legislature has limited funding for ethanol subsidies. Minnesota and Nebraska have incentives to encourage production of alcohol and new plants have opened in these States. In an effort to encourage ethanol use, EPA announced a proposed rule change permitting 10-percent ethanol blends in reformulated gasoline year-round.

If history is any guide, then, it will not be easy to predict where marginal ethanol supplies will come from in the future. Finally, even if it were possible to predict the future of the ethanol market, one still would have to predict how corn plantings would respond to regionally specific changes in ethanol production.

The same arguments apply to soybean farming. Consequently, in the absence of compelling reasons to do otherwise, I specify the model with national-average data on energy and chemical inputs to corn and soybean farming.

Note, though, that, as shown below, the national average total energy use -- fertilizer energy plus on farm fuel and-power -- on corn farms in 1991 was less than the average for Illinois, which produces the most ethanol, and less than the average for Nebraska, which has one of the fastest-growing corn outputs. Thus, a marginal analysis might conceivably come up with higher total energy inputs (fertilizer plus on-farm use) than the estimated national average inputs here.

Use of fertilizer for corn and soybeans

As discussed in Appendix K of DeLuchi (1993), it is best to estimate fertilizer use on an input/output basis, as lbs of fertilizer used per bushel of crop produced[40]. Thus, for fertilizer use, we have simply:

$$F_B = \frac{F_H}{Y_H}$$

where:

F_B = the amount of fertilizer applied per bushel harvested (lb/bu)
F_H = the amount of fertilizer applied to harvested acres (lb/harvested-acre) (discussed below)
Y_H = the crop yield (bushels per harvested acre) (data for corn, 1951-1996, from ERS, *Feed Situation and Outlook Yearbook*, 1997; data for soybeans, 1955-1996, from *Oil Crops Situation and Outlook Yearbook*, 1996)

Note that this calculation calls for the application rate per *harvested* acre. This is not the same as the application rate per *planted* acre, which since 1986 has been the basis of the published fertilizer-use data. We are interested here in the rate of fertilizer use on harvested acres specifically because, obviously, corn or soybeans to be used for fuel or feed must harvested. Planted acreage is equal to harvested acreage plus acreage that is planted but eventually abandoned and not harvested. Given that farmers probably apply relatively little fertilizer to acreage that is planted but eventually abandoned (Taylor, 1994), the rate of fertilizer use per harvested acre probably exceeds the rate per planted (harvested plus not-harvested) acre.

Taylor (1994) reports the total amount of fertilizer applied to corn and soybeans from 1964 to 1993. This total amount is calculated as the application rate per acre multiplied by the total number acres. Now, from 1964 to 1985, the fertilizer-use surveys collected data on harvested acreage only, and hence the reported application rate was the rate per harvested acre specifically. Thus, the parameter F_H for the years 1964 to 1985 can be estimated directly from the data in Taylor (1994).

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[40] Actually, we are concerned ultimately about lbs applied per bushel of material actually delivered to the fuel-production plant gate, after post-harvest storage, transfer, and transportation losses. I make this adjustment later.

Calculation of F after 1985. However, after 1985, the ERS surveyed and reported the average rate of fertilizer use per planted (and fertilized) acre, rather than per harvested acre (Taylor, 1994).Thus, the published data on fertilizer use from 1986 to 1995 must be adjusted to account for the greater use of fertilizer on acreage that eventually is harvested than on acreage that is planted but not harvested.

Given the rate of application of fertilizer on all planted acres, the amount of acres planted, and the amount of acres harvested, and an assumption regarding the application rate on non-harvested acres relative to that on harvested acres, the rate of fertilizer use on harvested acres can be calculated as:

$$F_H = \frac{F_P \cdot P}{H + R \cdot (P - H)} \qquad \text{eq. 65}$$

where:

F_H = the amount of fertilizer applied to harvested acres (lb / harvested-acre).
F_P = the amount of fertilizer applied to planted acres (i.e., all acres, harvested and non-harvested) (lb / planted-acre) (Taylor reports $F_P \cdot P$ for corn and soybeans from 1986 to 1993; the Agriculture Chemical Usage reports [National Agricultural Statistics Service [NASS], annual] reports data that can be used to calculate F_P for 1994, 1995, and 1996).
P = all planted acres, harvested plus non-harvested (see data sources for yield, Y_H, above).
H = the amount of acres harvested (see data sources for yield, Y_H, above).
R = the fertilizer application rate on non-harvested acres, relative to the application rate on harvested acres (see the discussion below).

The same adjustment is made to the original USDA data on pesticide use, which are reported in lbs of pesticide per planted acre (Lin et al., 1995).

Finally, essentially the same adjustment must be made to the data on fuel and electricity use per acre, which are reported per planted acre (Ali and McBride, 1994a, 1994b) and discussed below. The adjustment equation is of the same form as that given above for fertilizer; substitute “fuel and electricity” for “fertilizer,” and the appropriate energy units in place of lbs of fertilizer.

The R factor. The rate of energy use, fertilizer use, and pesticide use on non-harvested relative to harvested acres depends on how the use of energy, fertilizers, and pesticides are distributed over the growing season, and at what point non-harvested acreage is left alone. I assume that the bulk of fertilizer is applied relatively early in the season, and that energy and pesticides are used more uniformly throughout the season. I assume that non-harvested acreage is abandoned relatively early. On this basis, the R factor values are:

Fertilizer	R = 0.60
Pesticides	R = 0.40
Energy	R = 0.40

Historical and projected fertilizer use. Table 19 summarizes fertilizer/bu input-output for the period 1964 to 1996. Over the long term, nitrogen use per bushel of output has declined slightly; phosphate and potash use have declined more significantly. For my base-year values I use the averages from 1990 to 1996. Then, I project that fertilizer use per bushel will continue to decline slightly (Table 21), because of economic and environmental pressure to reduce nitrogen inputs to agriculture.[41] For lime (CaO), I use the value calculated from data in Ali and McBride (1994a, 1994b). (Data in the ERS' *Agricultural Resources and Environmental Indicators*(1994), indicate that the use of lime on corn fields has been declining.)

The new base-year assumptions for corn compared with previous assumptions, and those of Conway et al. (1994) are shown below:

	N	P_2O_5	K_2O	CaO	S	Total
Present assumptions (base year 1994)	1.100	0.420	0.510	0.330	0.010	2.370
Conway et al. (1994)	1.097	0.575	0.496	2.690	0.000	4.858
App. K of DeLuchi (1993)	1.325	0.500	0.677	2.692	0.013	5.207

This reduction in fertilizer use results in about a 5% reduction in fuel cycle GHG/mi emissions.

Use of pesticides on corn and soybeans

Previously, emissions associated with the manufacture and use of pesticides (herbicides, insecticides, fungicides, and related products) were accounted for by

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[41] Galloway [1998] states that "certainly, N fertilizer could be used more efficiently since on the order of 50% of the N applied is not taken up by the crop" (p. 23). However, he also notes that more efficient application requires "highly managed agricultural practices," which may be slow to be implemented, especially in developing countries. Vitousek et al. (1997) discuss a comparison of a "knowledge-intensive" fertilization regime, involving several small applications of fertilizer timed to the requirements of the growing crops, with a traditional regime involving a few large applications of nitrogen. The knowledge-intensive system used 1/3 less N per crop, had 10-fold lower emissions of NO and N_2O, generated higher yields, and was more profitable. Similarly, Mosier et al. (2002) state that "it is clear from many reports that when fertilizer N is applied in an amount needed by the crop for near optimum production, and at the time that the plants use the N, that N lossess are relatively small" (p. 491). However, they also note that "significant N losses through denitrification and leaching can be expcted even at 'optimal' rates" (p. 493). Finally, the IPCC (2001a) also sounds a cautionary note, stating that while "model simulations have demonstrated large potential for mitigating N_2O emissions by changing management practices...farmers will first need to accept [these practices" (p. 224).

multiplying fertilizer-related emissions by 1.20[42]. Now, emissions related to pesticide use are modeled explicitly.

Lin et al. (1995) report pesticide use per planted acre of corn and soybeans for various years from 1964 to 1992, and the *Agricultural Chemical Usage* series (NASS, annual) reports pesticide use planted acre in 1994, 1995, and 1996. With these data, and data on yields and planted and harvested acreage, and an assumption regarding the application rate on non-harvested acres relative to that on harvested acres, pesticide use per bushel were calculated, with the equation used to calculate fertilizer use per bushel. (As noted above, I assume that for pesticides, R = 0.40.) The calculated pounds of active pesticide ingredient per bushel harvested is:

Year	*Corn*	*Soybeans*
1964	0.011	0.013
1966	0.016	0.015
1971	0.021	0.036
1976	0.035	0.070
1982	0.032	0.067
1990	0.029	0.038
1991	0.030	0.035
1992	0.025	0.031
1994	0.022	0.027
1995	0.026	0.031
1996	0.025	0.033

It appears that pesticide use per bushel has dropped slightly since 1990, most likely on account of higher yields. With consideration of these results, the assumptions shown in Table 21 were used.

Energy inputs to corn and soybean farming

The model estimates of energy inputs to farming have been revised on the basis of data from the USDA's *Farm Costs and Returns Survey* (FCRS), which gathers information on the use of fuel and electricity inputs on a sample of corn and soybean farms. The FCRS reports data on hours of machine usage, acreage covered, type and size of machine, and type of fuel used (Ali and McBride, 1994a, 1994b). USDA analysts use these data "to support technical relationships that describe fuel consumption, repair requirements, and replacement costs. Engineering formulas are modified to reflect technological advances as they occur" (Ali and McBride, 1994a, p. 3). The result

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[42] Conway et al. (1994) also show that the energy embodied in "other chemicals" (pesticides, herbicides) used in corn farming is 20% of the energy embodied in the fertilizer.

is an estimate of the average use of fuel (gal / acre diesel, LPG, and gasoline; 1000 SCF / acre natural gas) and electricity (kWh / acre) in 10 major corn-producing states in 1991 and 14 major soybean producing states in 1990 (Ali and McBride, 1994a, 1994b)[43]. In several steps, I derive from these FCRS-based data an estimate of the national-average energy use per bushel of corn and soybeans.

1) First, I convert the FCRS data energy / use per acre from the reported per-planted-acre basis to a per-harvested-acre basis. As discussed above in regards to fertilizer use, I need data per harvested acre, because any crop to be used as a fuel must be harvested, and generally energy use per harvested acre will be greater than energy use per planted acre. To convert all of the data in the FCRS to a per-harvested-acre basis, I use the method described above for fertilizer use.
2) Second, I convert the data from a per-acre to a per-bushel basis, for each state:

$$E_{B,S} = \frac{E_{A,S}}{Y_{P,S} \cdot \frac{P_S}{H_S}} \qquad \text{eq. 66}$$

where:

$E_{B,S}$ = the energy use per bushel in state S.
$E_{A,S}$ = the energy use per harvested acre in state S from step 1 above.
$Y_{P,S}$ = the yield per planted acre in state S, for the farms in the FCRS survey.
P_S = the planted acreage in state S, for the farms in the FCRS survey.
H_S = the harvested acreage in state S, for the farms in the FCRS survey.

The result is the average energy use per bushel, for each state.

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[43] The FCRS estimates are consistent with independent, direct estimates of expenditures on fuels, reported in the 1992 Census of Agriculture. The FCRS data imply on the order of 7-9 gal-diesel / harvested-acre, and 4-4.5 gal-gasoline / harvested-acre, for soybean and corn (Tables 19 and 21). The *1992 Census of Agriculture* (Bureau of the Census, 1994) reports that in 1992, cash-grain farmers (SIC 011: wheat, rice, corn, soybeans, and cash grains not elsewhere classified) paid $1.034 billion for diesel fuel, and $0.536 billion for gasoline and gasohol, and harvested 159.4 million acres. (Note that the Census quite clearly instructs farmers to include only those expenses related to the farm business: “DO NOT include expenses connected with custom work for others; operation of nonfarm activities, business, or services; or household expenses not related to the farm business” [p. D-10; emphasis in original].) In 1992, the average price of diesel fuel used on farms was $0.77 / gallon ($2.69 billion spent on diesel fuel by all farms [*1992 Census of Agriculture]* divided by 3.5 billion gallons of diesel fuel used by all farms in 1992 [EIA, *Fuel Oil and Kerosene Sales 1994*, 1995]), and the average price of gasoline bought by farmers probably was around $0.85 / gallon [EIA, *Petroleum Marketing Annual 1994*, 1995]. Dividing total expenditures by the price and the harvested acreage results in 8.4 gal-diesel / harvested acre, and 4.0 gal-gasoline / harvested acre, for all cash grains -- consistent with the ranges estimated from the FCRS data for two major cash grains, corn and soybeans.

3) Third, I calculate the bushel-weighted average energy use per bushel for all of the states in the survey:

$$E_B = \frac{\sum_S E_{B,S} \cdot B_S}{\sum_S B_S} \qquad \text{eq. 67}$$

where:

E_B = the average bushel-weighted energy use per bushel for all of the states in the FCRS.
$E_{B,S}$ = the energy use per bushel in state S, from step 2 above.
B_S = the total bushel yield from all farms (not just those in the FCRS) in state S (USDA/NASS crop production data by state; available from the USDA/NASS website: www.usda.gov/nass).

At this stage, the results of the analysis for corn, expressed in 10^6 BTU of energy embodied in fertilizers, and 10^6 BTU of farm fuel and power, per bushel of production, are:

	CO	IL	IN	IA	MI	MN	NE	OH	SD	WI	Ave.
Fuel, power	0.021	0.013	0.013	0.012	0.018	0.013	0.039	0.015	0.021	0.016	0.018
Fertilizer	0.021	0.040	0.036	0.027	0.035	0.018	0.022	0.039	0.016	0.024	0.029
Total	*0.042*	*0.053*	*0.049*	*0.039*	*0.053*	*0.030*	*0.061*	*0.055*	*0.036*	*0.040*	*0.047*

4) I adjust average energy use per bushel from step 3 to account for the likely underestimation of relevant average energy use in the FCRS. For three reasons, the bushel-weighted average energy use per bushel for the farms in the 10 states in the FCRS probably slightly underestimates the national average energy use per bushel. First, it appears that the farms in the survey were a bit more productive than the average farm. The acre-weighted average bu/acre yield of the farms in the survey was about 7% higher than the average bu/acre yield of corn farms nationally. And bushel-weighted fertilizer use per bushel on the farms in the survey was 10-20% less than the national average use. Second, the FCRS estimates of fuel use do not include the minor amount of “other” fuels -- coal, kerosene, and wood -- reported in the USDA’s *Farm Production Expenditures* report (see Appendix K of DeLuchi [1993]). In Table K.6, I estimate that these other fuels supply 1-10% of total energy use per acre on corn farms. Third, the

FCRS estimates of fuel use probably do not include energy use by purchased service providers, such as crop dusters.

To account for these sources of underestimation, I multiply the calculated FCRS energy-use rates per bushel by 1.10.

5) Finally, the per-bushel rates estimated to this point are multiplied by the ratio of bushels harvested to bushels actually input to the fuel-production plant. This last step accounts for the small losses (I assume 2%) during storage, transfer, transportation, and pre-preprocessing. The net results are values of Table 21.

The revised analysis of Table 21 results in an energy consumption of about 19,000 BTU/bushel-corn in the year 2000, which is a bit less than the 22,000 BTU/bushel-corn assumed in the DeLuchi (1993). This reduction in farm energy consumption results in about a 3% reduction in fuel cycle GHG/mi emissions.

In modern intensive agricultural systems, energy inputs per unit of output (e.g., GJ per bu or ton) have tended to decline, mainly because of increasing yields derived from a given level of input (IPCC, 2001a). Whether this trend will continue depends on the balance of competing social forces. On the one hand, the development of biotechnology and gene technology could increase yields, increase resistance to pests, and increase the efficiency of nutrient and water uptake (IPCC, 2001a), all of which will reduce inputs per unit of output. However, in many parts of the world there has been considerable resistance to the adoption of genetically modified crops. Moreover, because of public concern for animal welfare, pressure for reduced chemical inputs, and increasing demand for organically grown food (all probably deriving from the same social trends that fuel resistance to genetically modified crop), the current trend in OECD countries is towards less intensive, lower input farming systems, which might result in lower yields and hence constant rather than declining GJ/ton input-output ratios (IPCC, 2001a, p. 225).

The IPCC (2001a, p. 226) notes that the extent of uptake of new bio- and genetic technologies will depend in part on public perceptions, and hence is difficult to predict. I assume that in the future there will be limited adoption of these new technologies (where the adoption is constrained by public concerns). I also assume that demand for low-input agriculture will continue to grow. Considering these forces, I assume that GJ/ton or GJ/bu input/output ratios will decline, albeit at relatively modest rates. My assumptions are shown in Table 21.

Note on the impacts of conservation tillage

In order to reduce soil erosion, some farms practice what is called "conservation tillage," in which some of crop residue is left on the soil after planting (Uri, 1998). Farms that practice conservation tillage use less energy (diesel fuel) because they don't till the soil as much, but they also use more pesticides and fertilizer (IPCC, 2001a; Uri, 1998). Also, as discussed in Appendix C, conservation tillage increases the carbon content of soils, but may increase N_2O emissions. The IPCC (2001a) lists conservation tillage as a major method of reducing greenhouse-gas emissions in the agricultural sector. However, the LEM does not formally model the adoption or impacts of conservation tillage.

Seeds

The FCRS reports lbs of soybean seeds used per planted acre, and total corn seeds used per planted acre, for the FCRS states in 1991. The application rate for soybeans appears to be quite high -- around 60 lbs / acre.

Assuming that seed usage per harvested acre is the same as seed use planted acre, and then following the method outlined in the previous section, I calculate a bushel-weighed national-average rate of 1.8 lbs-soybean-seeds / bushel-soybean, and 220-corn-seeds / bu. Assuming 0.06 g / seed-corn, the result is 0.03 lbs-corn-seed / bu. I cannot explain the large difference between the amount of soy seed and the amount of corn seed used. Instead, I assume that the data for soybeans are in error by an order of magnitude, and use a rate of 0.2 lbs-soybean-seeds / bu-soybeans.

Collection, grinding, baling, and transport of corn residue

In Table K.13 of DeLuchi (1993), I estimated that the collection, grinding, and baling of corn residue, for use as a fuel, required 0.28 to 0.56 million BTU of diesel fuel per ton of residue, and assumed a value of 0.42. In comparison with the energy requirements for grass harvesting, this seems high. I now assume a value of 0.30 million BTU per dry ton of residue.

Table K.13 of DeLuchi (1993) does not specify whether the residue tons are dry tons or wet (with moisture tons). I assume here that they are dry tons. However, in the calculation of the energy required to transport the residue, I assume that the residue is only partially dried, and weighs 25% more than when dried.

Production of cellulosic biomass: hybrid poplar, and switch grass

Most of the assumptions about productivity, fertilizer user, and N_2O emissions of SRIC (short-rotation intensive-cultivation) woody-biomass systems have been changed. The model structure also has been changed: as indicated by Table 21, the inputs are now treated explicitly, in terms of lbs of fertilizer or gallons of fuel (and so on) per ton of wood.

Perennial grasses have been added as a feedstock for ethanol production. In the model output, grass and wood feedstocks are combined into a single “biomass” fuel cycle. Thus, the user specifies the proportion of ethanol derived from grasses, and the proportion derived from trees, and the model calculates the weighted-average fuel cycle GHG emissions. Results for 100% grass or 100% wood are shown in the summary tables.

Productivity. In 1997, scientists from Oak Ridge National Laboratory (ORNL) and elsewhere reviewed the available data and field results, and projected yields through the year 2020 in every state with land suitable for SRIC wood or switchgrass plantations (Walsh, 1997a). Weighting the yield projected for each state by the state's share of national suitable acreage, I estimate the national-average dry harvest yields, shown in Table 20.

The estimates of Table 20 are bone-dry harvest yields, which means that they are net of harvest losses of some 1-2% (Walsh, 1998a), but not net of storage, transportation, transfer, and pre-processing losses. They are consistent with other assumptions and estimates in the literature[44]. The standing yields (SY), before harvest, would be 1-2% greater than these harvest yields. The effective yield into the fuel production plant, after storage, transfer, transportation, and pre-processing losses (PY), probably would be on the order of 4% (poplar) or 8% (switchgrass) less than these harvest yields (Walsh, 1998a; Perlack et al., 1992).

The ORNL estimates of Table 20 assume that plantations are managed to provide optimal yields. It is likely that in many situations other considerations, such as the need to promote wildlife diversity, will dictate a management regime different from the one that produces maximum yields. For example, Walsh (2003) notes that to promote wildlife diversity in switchgrass plantations less fertilizer would be used and plants would be harvested only every other year, effectively cutting yields in half. I account for this possibility by applying another scaling factor (0.90) to the ORNL "optimal-yield" estimates.

The harvest yield itself is not actually used in any calculation in this analysis; rather, it is used to calculate the standing yield (SY) or the into-the-plant yield (PY). SY is then used to calculate the carbon content of the standing biomass (in turn, a part of the calculation of carbon changes due to land-use changes), and PY is used to calculate fertilizer and fuel input data per dry ton of wood into the plant (dtp).

In the calculation of the energy requirements of biomass transportation, the actual weight of the material as transported, including moisture, rather than the dry

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[44] Graham et al. (1992) estimated that present wood plantations yield 11.3 metric tons of dry wood per hectare, after harvesting and transportation losses (5.0 short tons / acre), and that future plantations will yield 18.5 metric tons / ha (8.3 short tons / acre). On the basis of the work by Graham et al., Mann et al. (1995) assume 5 tons / acre / year, and Mann and Spath (1997) assume 5.7 tons / acre / year, after losses. Perlack et al. (1992) assume 5.9 tons / acre / year, after harvesting and transportation losses, for several sites. Fang et al. (1999, p. 421) reported a yield of 10-13 tonnes / ha / yr, or 4.5 to 5.8 tons / ac / yr, for poplar. These figures are consistent with the estimates of Table 20.

In a recent modeling exercise, Andress (2002) uses ORNL data and assumes yields of 4.6 to 6.0 tons / ac / yr for switchgrass. Lemus et al. (2002) reported a yield of 2.9 to 5.3 (average of 4.0) tons / ac / yr for 20 switchgrass populations harvested between 1998 and 2001. However, the lowest yield (2.9 tons / ac / yr) was associated with relatively low fertilizer application; when fertilizer was applied at closer to the rate recommended by ORNL, the average yield was 4.6 tons / ac / yr. Moreover, Lemus e al. (2002) suggest that particular ecosystem in Iowa is likely to produce relatively low yields.

Reynolds et al. (2000) reported relatively high yields of about 20 tonnes / ha / yr (8.9 tons / ac / yr) from experimental plots in Tennessee. The reasons for the unusually high yields are not clear. Sanderson et al. (1996) report yields from experimental plots in the southeastern U.S.; most are about 4 tons / ac / yr.

weight is used. (According to Perlack et al. [1992], biomass is partially dried in the field before transport, and then the biomass is completely dried after transport and before input to the fuel-production facility.)

Fertilizer. The ORNL data-review workshop (Walsh, 1997a)[45], mentioned above, recommended application rates for N, K, P, and lime, for different regions of the U. S. In most cases, the researchers recommended that the fertilizer by applied in only one year of the rotation. To estimate lb-fertlizer/dpt, one must average the rates over the different regions, average one-time applications over the entire life of the rotation, scale from K and P to K_2O and P_2O_5, and divide by PY, the expected into-the-plant yield per acre. Applying these transformations to the ORNL recommendations results in the following lb/ton applications, for circa 1996 yields:

	N (lbs/ton)	P_2O_5 (lbs/ton)	K_2O (lbs/ton)	Lime (lbs/ton)
Hybrid poplar	2.0	1.6	1.1	41.3
Switchgrass	20.6	0.8	1.1	31.0

The ORNL-estimate rates for poplar lower than the oft-cited estimates of Turhollow and Perlack (1991): 50-kg N/ha, 15-kg P_2O_5/ha, and 15-kg K_2O/ha -- which, assuming 5.8 t/acre, result in 7.6 lbs-N and 2.3 lbs K_2O and P_2O_5 per net ton of wood. The National Renewable Energy Lab's (NREL) detailed evaluation of the biomass fuel cycle (Perlack et al., 1992) assumed the Turhollow and Perlack values for tree plantations. And Mislevy and Fluck (1992) applied relatively high levels of fertilizer -- 168 kg N/ha, 25 kg P/ha, and 93 kg K/ha (roughly 27 lb N/ton, 9 lbs P_2O_5/ton, and 18 lbs K_2O/ton) at an experimental grass plot in Florida. On the other hand, Mann and Spath (1997) adopt values similar to the ORNL recommendations.

The ORNL-estimated rates for switchgrass are consistent with recent application rates in Iowa. Lemus et al. (2002) report on fertilizer applications and yields of 20 switchgrass populations harvested in Iowa between 1998 and 2001: 56 kg-N/ha (50 lbs/ac) and 2.9 tons/acre yield in 1998 (resulting in 17 lbs-N/ton-yield), and 112 kg-N/ha (100 lbs/ac) and 4.6 tons/acre average yield in 1999-2001, or 21.7 lbs/ton[46]. However, Reynolds et al. (2000) achieved lower nitrogen rates: they applied 89 lbs-

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[45] Walsh (1998a) has confirmed these rates in a more recent communication.

[46] Mislevy and Fluck (1992) applied relatively high levels of fetilizer -- 168 kg N/ha, 25 kg P/ha, and 93 kg K/ha (roughly 27 lb N/ton, 9 lbs P_2O_5/ton, and 18 lbs K_2O/ton) at an experimental grass plot in Florida. On the other hand, Sanderson et al. (1996) cite a 1994 study that recommended only 30 kg-N/ha/yr. I ignore these older data.

N / ac to experimental switch grass plots in Tennesse and harvested 8.9 tons / acre, giving a rate of only 10 lbs-N / ton-yield.

Why are the nitrogen requirements for poplar estimated by ORNL so low? The scientists at the ORNL data-review workshop (Walsh, 1997) recommend relatively low levels of N in part because a substantial amount of nitrogen is recycled via the leaf litter, and trees apparently don't respond to higher levels of nitrogen. Nevertheless, these recent recommendations appear to me to presume best practice under good conditions, and as a result, I assume that they apply to the year 2005 rather than the year 1996. I assume that all lb / ton application rates, except that for N applied to switchgrass, decline with the rate of increase in yield. (The application rate for switchgrass was given by the ORNL workshop in terms of lbs / ton, whereas the other rates were given in lbs / acre.)

Pesticides and seeds. Turhollow and Perlack (1991) estimate that the energy embodied in pesticides used in SRIC is 12% of the energy embodied in the fertilizer. Mann et al. (1995) report application rates on tree plantations of up to 10 lbs / acre. NREL's detailed evaluation of the biomass fuel cycle assumes 0.23 lbs / acre for tree plantations (Perlack et al., 1992), or 0.04 lbs / ton-wood, and 0.03 lbs / ton-grass. Walsh's (1997b) model of biomass production, BIOCOST, assumes 0.13 lbs / ton-wood. More recently, Walsh (1998a) estimates 1.75 lbs a.i. / acre (a.i - active ingredient) herbicide and insecticide plus 1 quart / acre of herbicide which, in year 1 or year 2. Assuming 3000 grams / gallon of the herbicide, an average 8-year rotation, and a yield of 4.5 dt / acre, Walsh's (1998a) recent estimate becomes 0.095 lbs / ton-wood. I assume 0.10 lbs / ton in 2005, declining by the rate of increase in the per-acre yield (on the assumption that the lb / acre application rate remains constant).

The ORNL data-review workshop (Walsh, 1997a), discussed above, recommended 4-5 lbs active ingredient (a.i.) / acre in the establishment year for switchgrass. Walsh (1998a) now estimates 3 lbs a.i / acre, plus 5 lbs of seeds per acre. Consistent with these recommendations, Lemus et al. (2002) applied an herbicide at the rate of 2 lbs a.i / acre in the first two years of planting, equivalent to 4 lbs in one year, in actual switchgrass plantations harvested between 1998 and 2001. Data in Sanderson et al. (1996) indicate an herbicide application rate of about 2.5 lbs a.i. / ac in one year. Given a 10-year rotation and a present yield of 5.3 tons / acre, 3-5 lbs a.i. / acre (the range of the estimates cited above) results in 0.06 - 0.09 lbs a.i. / ton, declining by the rate of increase in the per-acre yield. This is similar to the rate assumed in the BIOCOST model (Walsh, 1997b). Seed use is 0.09 lbs / acre.

Fuel use. Turhollow and Perlack (1991) use data from Blankenhorn et al. (1985) to estimate that the establishment, harvesting, and use of equipment for SRIC consumes 0.69 gJ diesel fuel per Mg of wood (4.3 gallons / ton) (excluding energy embodied in fertilizer and pesticides), and that hauling from field to production facility (40 km away) consumes 0.23 gJ diesel fuel per Mg wood[47]. Perlack et al. (1992) estimate diesel fuel use to be 2.3 gallons/dry ton, including energy for moving equipment and materials to the field. The BIOCOST model (Walsh, 1997b) assumes about 2.2 gal/dry ton for wood, and 1.6 for grass. However, Mann and Spath [1997] estimated less than 2 gal/dry ton wood using the BIOCOST model. Consistent with this lower figure for grass, Mislevy and Fluck (1992) used only 1.3 gal/ton to establish, fertilize, and harvest grass on an experimental plot in Florida. Recently, Walsh (1998) estimated diesel fuel use for poplar and switchgrass, year by year, for different regions. The average application rate is 1.74 gal/acre for switch grass and, 1.89 gal/acre for poplar.

The assumptions are shown in Table 21. Note that minor amounts of gasoline and electricity are also assumed to be used.

GREENHOUSE-GAS EMISSIONS RELATED TO CULTIVATION AND FERTILIZER USE

Overview of the method

Cultivation and fertilizer use can affect climate in many ways. A change to an agricultural ecosystem can change its primary productivity, and hence change the amount of carbon sequestered in soils and biomass. Agricultural cultivation, along with the use of fertilizer, affects nitrogen and carbon dynamics in soil and groundwater, and thereby changes fluxes of N_2O, CH_4, CO_2, and other gases that affect climate. Nitrogen can leach away from the site of application and fertilize plants, and thereby sequester carbon, in non-agricultural ecosystems. (For reviews of anthropogenic disturbances to the nitrogen cycle, see Erisman et al [1998], Galloway [1998], and other articles in the same issue of *Enironmental Pollution*.)

Our analysis attempts to account for many of the affects of cultivation and fertilizer use on climate, albeit in some instances only crudely. The method is similar to that recommended by the IPCC (1997) in its guidelines for estimating national greenhouse-gas emissions inventories. We pay special attention to the addition and fate of nitrogen fertilizer, because it is involved in so many GHG-producing pathways.

We consider the impact of changing, cultivating, and fertilizing crops, on four direct and indirect GHGs: CO_2, CH_4, N_2O, and NO_x. In the case of CO_2, we further distinguish carbon emissions or sequestration in the soil and litter from carbon sequestration in plant biomass. We note that nitrogen has impacts off the site of application (i.e., in ecosystems other than the crop system that initially receives the N) as well as on-site, but assume that cultivation per se affects only the site being cultivated. The impacts that we can examine can be summarized as follows:

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[47] Assuming 19.8 gJ/Mg-wood (dry) (Graham et al., 1992), the resultant energy-use intensity is about 0.035-BTU/BTU-wood for establishment, harvesting, and equipment, and 0.012-BTU/BTU for hauling. These are the same as the values estimated in DeLuchi (1993, pp. K-22 and K-23).

Greenhouse gas	**Impact of agricultural N input**	**Impact of cultivation**
CO_2 (C in soil)	N affects oxidation of carbon in soil	changes primary production; increases oxidation and erosion of organic matter
CO_2 (C in biomass)	N stimulates plant growth and hence carbon sequestration in biomass, on site and off site	changes primary production (amount of organic matter in plants)
N_2O	some N converted by microbial nitrification and denitrification to N_2O	accelerates mineralization of N-rich organic matter, to provide N for conversion to N_2O
NO_X, NH_3	some N volatilizes as NH_3, NO_X	not considered here
CH_4	N reduces oxidation of CH_4 in soils	reduces oxidation of CH_4 in soils

These impacts are represented in the model by the following parameters:

Greenhouse gas	**Parameter for impact of agricultural N input on gas**		**Parameter for impact of cultivation on gas**
	on-site	*off-site*	*(assume on-site only)*
CO_2 (C in soil, litter)	CO2SF	CO2NF	CO2C (A?CS)
CO_2 (C in biomass)	not modeled formally	CO2NF	CO2C (A?CB)
N_2O	GHGN2OF (N2ODF)	GHGN2OF (N2OIF)	GHGN2OS
NO_X (and NH_3)	---------------- GHGNO2F ------------------		not estimated
CH_4	----------------- GHGMF -------------------		GHGMS

Each of these impacts is discussed and in most cases estimated in separate sections below. Appendix C reviews data on GHG emissions from soils. Appendix D presents the estimation of a CO_2-equivalency factor for NO_X, which includes estimates of impact of N input on carbon sequestration, nitrous oxide emissions, and NO_X and NH_3 emissions. Note that we also include in this section an estimate of CO_2-equivlent GHG emissions from the burning of agricultural residues.

Following the IPCC (1997), we distinguish four kinds of agricultural nitrogen inputs: nitrogen in synthetic fertilizer, nitrogen in animal manure, nitrogen fixed by legumes, and nitrogen in crop residues. We assume that all four kinds of N input have all of the impacts summarized above, off-site as well as on-site, but that the fraction of biologically fixed and crop-residue N that is lost off site is less than the fraction of synthetic or animal-manure N that is lost offsite.

The parameters shown above all have the units of grams of CO_2-equivalent emissions per bushel or per ton of crop produced. To estimate the total CO_2-equivalent effect of all the parameters, per million BTU of fuel, we simply sum the estimated parameters and convert from a per-bushel or ton-basis to a per-BTU-fuel basis:

$$GHGLC_E = \left(1 + FL_F\right) \cdot YEF_{E,F} \cdot \sum Parameters \qquad \text{eq. 69}$$

where:

subscript E = energy-crop system (corn, soybeans, grass, SRIC wood, coal mining).

subscript F = fuel made from crop E (ethanol, methanol, biodiesel, SNG).

$GHGLC_{E,F}$ = CO_2-equivalent emissions related to changes in land use, cultivation, and fertilizer use in energy system E, per energy unit of fuel F delivered to consumers (g-CO_2-equivalents / 10^6-BTU-fuel).

“Parameters” = the emission impacts shown in the table above (g-CO_2-equivalents / bu or dry ton).

FL_F =fraction of production of fuel F lost due to evaporation or spillage (Appendix B of DeLuchi [1993], and updates thereto in this report).

$YEF_{E,F}$ = use of feedstock E per energy output of fuel F from the production plant (bu / 10^6 BTU-fuel in the case of soy and corn feedstocks; tons / 10^6-BTU-fuel in the case of wood and grass feedstock) (calculated based on data of Table 17).

Figure 5 illustrates some of the nitrogen flows and associated GHG emissions in a corn system.

Nitrogen input in energy crop system E

As indicated above, some greenhouse-gas emissions are a function of the amount and kind of nitrogen input. Analytically, this nitrogen input should be understood to be the difference between the total amount of nitrogen input in a “base case” world and the total amount used in an alternative world in which the production of crop E changes.

Following the IPCC, we consider four kinds of agricultural nitrogen input: in synthetic fertilizer, in animal manure, from nitrogen fixed by plants, and in crop residues. The units of input are grams of N per bushel or ton of crop. Synthetic fertilizer input is discussed elsewhere in this report (see Table 21 and related text); the other three kinds of inputs are discussed next. (N input via deposition of atmospheric nitrogen formed by fossil fuel combustion is estimated as part of the CO_2-equivalency factor for NO_X, in Appendix D.)

Animal manure ($F_{N,AM,E}$). There is an important difference between the use of animal manure and the use of synthetic fertilizer: synthetic fertilizer is manufactured expressly for the purpose of fertilizing crops, with the result that there is an incremental amount of fertilizer that would not have been produced had additional crops not been grown, whereas manure is a byproduct which, by and large (price and demand effects aside), would be available even if it were not used as fertilizer. In principle, we should determine the fate of nitrogen in animal manure *not* used as fertilizer, and attribute to energy-crop system E the difference between the nitrogen cycle with manure, and the nitrogen cycle with manure in its alternative use. However, at the moment, we are unable to do this. Consequently, for now, we assume that the use of manure or crop residue as fertilizer in energy-crop system E does not appreciably affect the nitrogen and carbon cycle (compared to the manure its alternative use), and hence that with animal manure there is no *net* addition of nitrogen in the world. Thus:

$$\textit{Assume:} \qquad F_{N,AM,E} \approx 0$$

Nitrogen fixed by plants ($F_{N,FX,E}$). Beans, pulses, alfalfa and other plants fix nitrogen in the soil. (See e.g. Vitousek, et al. [2002] and Shantharam and Mattoo [1997] for general discussions of biological nitrogen fixation.) The IPCC (1997, p. 4.90) suggests that atmospheric N_2 fixed by plants can be nitrified and denitrified and produce N_2O in the same ways that synthetic N can, and furthermore, that the Rhizobia living in root nodules are able to denitrify and produce N_2O. Although there are few data, the IPCC (1997) recommends assuming that the rate of production of N_2O from biologically fixed N is the same as the rate from synthetric fertilizer N. See Appendix C for further discussion.

But how much N do plants like soybeans produce? Galloway (1998) states that “actual [nitrogen] fixation rates per unit area can vary substantially by cultivar, temperature, tilling conditions, method of measuring N-fixation, etc.” (p. 19), and Smil (1999) agrees. The IPCC (1997) method assumes that the amount of N fixed, and hence the amount potentially available to the plant, or for conversion to N_2O, is precisely the amount of N in the plant. If this is correct, then plants do not incorporate all of the N that they fix, because elsewhere, the IPCC (1997) states that biological fixation of N can supply 50-60% of the total nitrogen in “grain legumes” (such as soybeans) and 70-80% of the total nitrogen in “pasture legumes.” (Consistent with this, Smil [1999, p.651 estimates that all N-fixing crops derive 67% of their N from biofixation.] Hardarson and

Atkins (2003) provide a graph that indicates that soybeans derive about 55% of their N from N_2 in air, and that this amounts to about 100 kg-N / ha / ha, but they also show that the fixed-N percentage can range from less than 40% to more than 60% depending on temperature (maximum fixation occurs at about 28^o C), strain of rhizobia bacteria, use of synthetic fertilizer (generally, the more synthetic fertilizer used, the lower the fixation), and other factors. Smil (1999) reports that estimates in the literature of the amount of N fixed by soybeans spans from 14 to 450 kg-N / ha / ac; in his own analysis, he assumes a range of 60 to 100 kg-N / ha / ac.

However, the percentage of total soybean N supplied by biologically fixed N_2 can cover a wide range, from as little as 10% to more than 70% when special methods such as innoculaion with nitrogen-fixing bacteria are used (e.g., Galal, 1997). Decaying biomass in the soil, and a little bit of synthetic fertilizer, provide the remaining N in the plant. In support of this, Paustian et al. (1990) made a complete N budget for an N-fixing lucerne ley and found that N_2-fixation provided 74%, and mineral soil N 36%, of the total N input to the roots (the mineral N, in turn, came from soil litter and fauna), but that the fixed N was 86% of the total amount of N in the roots and above-ground biomass. It thus appears that plants do indeed fix nearly as much N as they contain, yet do not incorporate all of the N that they fix. The IPCC (1997) assumption that legumes like soybeans fix (but do not necessarily incorporate) as much N as they contain was adopted for the LEM.

Formally, the amount of nitrogen fixed by plants is estimated as:

$$F_{N,FX,E} = WB_E \cdot MCF \cdot RR_E \cdot NF_E \cdot NFXR_E \cdot 453.6 \qquad \text{eq. 70}$$

where:

$F_{N,FX,E}$ = Nitrogen fixed by plants in energy-crop system E (g-N / bu).

WB_E = weight per bushel of crop E (56 lbs / bu-corn, 60 lbs-bu-soybeans, at 15% moisture).

MCF = moisture correction factor, to get to dry weight (0.85).

RR_E = scaling factor to account for unharvested residue (e.g., corn stover) and roots from energy crop E, not included in the standing yield estimates (the ratio of total plant biomass to the crop mass; as discussed below, about 2.2 for corn, and 3.4 for soybeans).

NFE = nitrogen weight fraction of dry crop E (The IPCC [1997] gives default values of 0.03 kg-N / kg-dry-biomass for all nitrogen-fixing crops, and 0.015-kg-N / kg-dry-biomass for all other crops).

$NFXR_E$ = fixed-nitrogen / plant-nitrogen ratio: the ratio of N fixed by plant, to the N content of the whole plant, for crop E (assumed to be 1.0 for soybeans, 0.0 for non-fixing plants such as corn, wood, and grass; see discussion above).

453.6 = g / lb

We need to know the ratio of total plant biomass to crop biomass. Agriculture and Agri-food Canada [1997] state that a typical corn plant at maturity consists of 50% grain and 50% stover or above ground residue[48], and that roots add another 10%. Similarly, the EIA [*Emissions of Greenhouse Gases in the United States 1997,* 1998] cites an estimate that the ratio of corn residue to corn crop volume (probably excluding roots) is 1.0. The EIA (1998) also cites an estimate that residue: crop ratio for soybeans (again, probably excluding roots)[49] is 2.1. Appendix L of the EPA's *Inventory of U. S. Greenhouse Gas Emissions and Sinks: 1990-1999* (2001), also assumes that the above-ground residue: crop ratio is 1.0 for corn and 2.1 for soybeans. Thus, the scaling factor for corn is 2.2, and for soybeans about 3.4 (See also IPCC [1997].)

Credit for excess N fixed. Nitrogen that is fixed by but not incorporated into the plant is added to the soil. This excess soil N can be made available to a non-N-fixing plant, such as corn, when it is rotated with an N-fixing plant. (This sort of rotation, in fact, is common.) When this happens, the natural production of N, by biological fixation, has effectively substituted for synthetic nitrogen production[50]. Therefore, I assume that every gram of N biologically fixed but not used by soybeans displaces some fraction of a gram of synthetic fertilizer N, and assign to soybean production the following GHG emissions credit:

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[48] Consistent with this, Table K.13 of DeLuchi (1993) indicates that there are 2.5 to 2.6 tons of corn residue per acre.

[49] As some indication of the mass of roots relative to the mass of the total plant, we note that Alves et al. (2003) write that non-recoverable root N is 30-35% of total plant N.

[50] Hardarson and Atkins (2003, p. 49) write that in mixed-crop (legume/non-legume) rotations, the non-legume draings the soil of N, and thereby effectively forces the legume to fix more N_2 than it would were there no other crop. They conclude that "the legume thus 'spares' N for use by the non-legume" (p. 49).

$$NDC_{FX,E-E^*} = EN_{FX,E} \cdot RDEN_{E-E^*} \cdot GHGLN_{E^*}$$

$$EN_{FX,E} = F_{N,FX,E} - \frac{F_{N,FX,E}}{NFXR_E} \cdot BNF_E = F_{N,FX,E} \cdot \left(1 - \frac{BNF_E}{NFXR_E} \right)$$

$$GHGLN_{E^*} = GHGML + GHGN2OF^{\wedge}_{E^*,T} + GHGNO2F^{\wedge}_{E^*}$$

$$+GHGMF^{\wedge}_{E^*} + CO2SF^{\wedge}_{E^*} + CO2NF^{\wedge}_{E^*}$$

$$BNF_E \leq NFXR_E$$

eq. 71

where:

Subscript E* is crop that benefits from the excess nitrogen fixed by crop E (I assume that corn follows soybeans).

Superscript ^ means that the term is the same as the corresponding term in Eq. 69 except that the units are g-CO_2-equivalent / g-N-synthetic fertilizer instead of g-CO_2-equivalent / bu.

$NDC_{N,FX,E-E^*}$ = GHG emissions credit for displacement of N (in crop system E*) by biological fixation of excess N by crop E (g-CO_2-equivalent / bu or dt)

$EN_{FX,E}$ = excess nitrogen fixed by crop system E (g-N / bu or dt).

$RDEN_{E-E^*}$ = the ratio of nitrogen displaced (in crop system E*) to excess nitrogen fixed (in crop system E) (for want of reasons to the contrary, I simply assume 0.5:1; i.e., that every gram of excess N fixed displaces one-half grm of synthetic N

$GHGLN_{E^*}$ = lifecycle emissions from the production, application, and eventual fate of [displaced] nitrogen in system E* (g-CO_2-equivalent / g-N).

BNF_E = the ratio of biologically-fixed N incorporated within the plant to the total N content of the plant, for crop E (assumed to be 0.55% for soybeans, as per the IPCC [1997] cited above) (note the difference between this parameter, for which the numerator is N biologically fixed *and incorporated within the plant*, and the parameter $NFXR_E$, for which the numerator is just N biologically fixed).

GHGML = the greenhouse-gas emissions from the nitrogen fertilizer manufacturing lifecycle (g-CO_2-equivalent / g-N-fertilizer; calculated from data in in Appendix H.

Other terms defined in Eq. 70.

This credit turns out to be important, because it helps offset the rather large emissions presumed to result directly from the biological fixation of N in the first place.

Nitrogen in crop residue ($F_{N,CR,E}$). The IPCC (1997) assumes that all of the nitrogen in the crop residue that is left on the field eventually is available as N in the soil, and hence potentially available for conversion to N_2O. Again, they assume that the rate N-N_2O / N-residue is the same as the rate N-N_2O / N-synthetic fertilizer, which in this case seems broadly reasonable.[51] The rate of N addition from crop residue is simply:

$$F_{N,CR,E} = WB_E \cdot MCF \cdot (RR_E - 1) \cdot NFR_E \cdot CRF_E \cdot 453.6 \qquad \text{eq. 72}$$

where:

$F_{N,CR,E}$ = Nitrogen from crop residue in energy-crop system E (g-N / bu [corn, soybeans] or g-N / dt [wood, grass]).

WB_E = weight per bushel of crop E (56 lbs / bu-corn, 60 lbs-bu-soybeans, at 15% moisture; this term is not used in the case of wood or grass).

MCF = moisture correction factor, to get to dry weight (0.85; term not used in the case of wood or grass).

RR_E = ratio of total plant biomass (crop, residue, roots) to harvested crop mass (see Eq. 70 for corn and soybeans; I assume 1.05 for switchgrass, and 1.10 for hybrid poplar, on the basis of estimates in Perlack et al. [1992], and my judgment in light of data indicating significant root development for switchgrass; RR_E -1 is the ratio of residue+root mass to harvested crop mass).

NFR_E = nitrogen weight fraction of dry crop residue E (note that this is not the same as parameter NF_E, because the nitrogen content of the residue – the parameter used here – is not the same as the nitrogen content of the entire above-ground biomass) (discussed below).

CRF_E = of the total crop residue, the fraction that is left in the field (the IPCC [1997] recommends 95%; the EPA [2001] recommends 90%; I assume 92%).

453.6 = g / lb (use 907,200 g / ton in the case of wood, grass).

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[51] In support of this, Kaiser et al. (1998) found that over a year the highest N_2O emissions were associated not with the application of N fertilizer, but with the mineralization of N in crop residue. Similarly, Baggs et al. (2003) found that the addition of residues to soil not only increased total N_2O emissions, but also increased N_2O emissions from inorganic fertilizers (determined by ^{15}N labeling) under some circumstances, on account of the C from the residues stimulating denitrification in the presence of inorganic N. N_2O emissions from residues increased with N content and decomposability of the residue.

See Hood et al. (2000) for a discussion of N uptake from various organic residues.

To proceed with the IPCC method, we need to know the amount of N in plant residue. Data from the IPCC (1997) indicates less than2.5% for soybeans, and 0.94% for corn. The EIA (*Emissions of Greenhouse Gases in the United States 1997*, 1998) cites the same original sources as the IPCC (1997); both reports show 2.3% for soybeans, and 0.8% for corn. The EPA (2001) assumes 2.3% for soybeans, and 0.58% for corn. Russelle et al. (2001) report 1.45% N by weight for corn grain, 0.53% for corn stover, and 2.81% for alfalfa herbage. Smil (1999) estimates 2.5% for legume residue and 0.6% for cereal-crop residue. Hood et al. (2000) measured 0.7% for corn and 3.2% for soybean residue. I assume 2.5% for soybeans, and 0.7% for corn.

Also, I assume 0.17% for hybrid poplar [Mann and Spath, 1997] and 0.54% for switchgrass (Lemus et al., 2002; see also Reynolds et al. [2000], who show a range of 0.28% to 0.86%, and also Perlack et al. [1992] who assume that the crude protein content of switchgrass is 2.65 times that of hybrid poplar.)

Comment on the total N input. Given 60-lb-soy / bushel-soy at 15% moisture, a total-plant / crop mass ratio of 3.4:1 (see Eq. 70), and 0.03-kg-N / kg-soybean-biomass results in 5.2 lb-N (whole-plant) / bu-soybean[52], which is many times higher than the synthetic application rate of 0.1 lb-N / bu-soy (see Table 19) -- which we would expect, because nitrogen-fixing plants generally do not need synthetic nitrogen fertilizer.

In the case of soybeans, the total N input (almost entirely from biological fixation and crop residue), is on the order of 3.5 kg-N per bushel -- somewhat higher than the soybean N content of 2.3 kg-N per bu (from the 5.2 lbs-N per bu above), but several times the per-bushel input for corn. The total N input for soybeans results in enormous fuelcycle CO_2-equivalent emissions: 3.5 kg-N / bu-soybean, multiplied by the N-N_2O / N-input rate of 0.011, the gal / bu converion factor (about 1.5), the N_2O / N_2 weight factor (1.57) and the CO_2-equivalency factor for N_2O (assume 300 for the purposes of this calculation) results in 27 kg-CO_2 equivalent per gallon biodiesel, or over 4,000 g / mi! This is an *enormous* emission rate.

Is this emission rate plausible? The IPCC (1997) notes that measurements have indicated emissions of on the order of 4 kg-N-N_2O / ha / yr from legume fields. Consistent with this, the N_2O emissions model of Mummey et al. (1998) predicts 4.6 kg N-N_2O / ha / yr total N_2O emissons from conventionally tilled soy fields in the U. S. The rate implied by the parameter values above, assuming 100 bu-soybeans / ha / yr, is about 3.9 kg-N-N_2O / ha -- consistent with the rate cited by the IPCC and the simulation of Mummey et al. (1998). Still, the enormous impact of this rate on fuel cycle emissions suggests that we urgently need more research on these parameters.

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[52] A similar calculation for corn (1.5% N, 2.2 total-plant / crop ratio, 56-lbs / bu, 85% dry matter) results in 1.57 lb-N (whole plant) / bu-corn, which is just above the synthetic fertilizer application rate of 1.2 lb-N / bu-corn (Table 19), which seems reasonable.

N_2O from nitrogen input ($GHGN2OF_E$)

The fate of the nitrogen in synthetic and natural fertilizers is complex. Initially, most of the added nitrogen is taken up by plants or, in a variety of forms, retained in soils or groundwater (Perlack et al., 1992; Paustian et al., 1990). A small amount, however, is released to the atmosphere directly as N_2O, NO, NO_2, or NH_3. In addition, a substantial fraction of the nitrogen in groundwater leaches or drains offsite and later evolves into N_2O or NO. The emissions of N_2O, NO_X, and NH_3 depend on many factors, including: the type of biomass being grown; the amount, type, depth, and frequency of application of fertilizer; the temperature, water content, and acidity of the soil; agricultural and harvesting practices; and others (IPCC, 1996c; Appendix C to this report).

The most important of these emissions is N_2O, which can be a major source of CO_2-equivalent emission in fuel cycles in which a large amount of fertilizer is applied. N_2O is produced from complex microbial nitrification, denitrification, and decomposition processes in soils. Increases in the amount of N added to the soil typically increase N_2O emissions.

The emission of N_2O usually is expressed as grams of nitrogen lost as N_2O per gram of nitrogen input (usually as synthetic fertilizer). (Note that the N-N_2O emission is *not* expressed per gram of N actually retained on the site, but rather per gram of total N input.) The lost N_2O has two components: N_2O lost on site from fertilizer in the soil, and N_2O lost offsite, from fertilizer carried away in groundwater. Allowing that these rates can change over time, we have our formal model:

$$GHGN2OF_{E,T} = \left(N2ODF_{E,1990} + NLF_{E,1990} \cdot N2OIF_{E,1990}\right) \cdot \left(1 + \frac{\Delta\%NLF}{100}\right)^{T-1990}$$

$$\cdot AF_E \cdot F_{N,E} \cdot \frac{MW_{N2O}}{MW_{N2}} \cdot CEF_{N2O}$$

$$F_{N,E} \text{ is } F_{N,SF,E} \text{ or } F_{N,AM,E} \text{ or } F_{N,FX,E} \text{ or } F_{N,CR,E} \qquad \text{eq. 73}$$

where:

$GHGN2OF_{E,T}$ = N_2O emissions from nitrogen input to energy-crop system E in year T (g-CO_2-equivalent/bu or ton).

$N2ODF_{E,1990}$ = direct, on-site N_2O emissions from nitrogen input to energy-crop system E in the base year 1990 (g-N-NO_2/g-N-fertilizer; discussed below).

$NLF_{E,1990}$ = of the N input to energy-crop system E, the fraction that leaches off-site, in 1990 (discussed below).

$N2OIF_{E,1990}$ = indirect, off-site N_2O emissions from nitrogen that has leached off site of energy-crop system E, in the base year 1990 (g-N-NO_2 / g-N-input; discussed below).

T = the target year of the analysis.

?%NLF = the annual percentage change in the other parameters.

AF_E = the fraction of acreage fertilized (by convention, 1.0, because the fertilized application rates used are averages over all acres).

$F_{N,E}$ = N fertilizer input to energy-crop system E (g / bu).

$F_{N,SF,E}$ = Synthetic N fertilizer added in energy-crop system E (g / bu; based on Table 21).

$F_{N,AM,E}$ = Animal-manure N input to energy-crop system E (g / bu; assumed to be zero, as discussed above).

$F_{N,FX,E}$ = biologically fixed N input to energy-crop system E (g / bu; discussed above).

$F_{N,CR,E}$ = crop-residue N input to energy-crop system E (g / bu; discussed above).

MW_{N2O} = the molecular mass of N_2O (44 g / mole).

MW_{N2} = the molecular mass of N_2 (Table 5; note that we must use N_2, rather than just N, because N_2O has two nitrogen atoms).

CEF_{N2O} = the CO_2-mass-equivalency factor for N_2O (Appendix D)

On-site or "direct" N_2O emission rate ($N2ODF_{E,1990}$). In a recent review and analysis of the literature, Bouman (1996) estimates that the first component, "direct" emission of N as N_2O, is 1.25% of added nitrogen fertilizer, for all crops. The IPCC (1997) adopts this value. The IPCC (1997) also assumes that the rate of N_2O emission from biologically fixed N, and from crop-residue N, is the same as rate of emission from synthetic-fertilizer N. We assume the same. However, the evidence reviewed in Appendix C of this report indicates that the emission rate for corn is higher than the emission rate for other crops, and that the emission rate for wood and grass is relatively low. My assumptions are shown below.

Offsite or "indirect" N_2O emission ($NLF_{E,1990}$ and $N2OIF_{E,1990}$). Until recently, there was virtually no data on the second emission-rate component, N-N_2O / N-nitrogen from groundwater offsite. Studies reviewed in Appendix C indicate that some 20-30% of applied nitrogen leaves the site, and that some 0.05% to 5% of this off-site nitrogen evolves as N_2O. Our assumptions are shown in Appendix C.

The change in the emission rate (?%NLF). N_2O emissions from synthetic fertilizer can be reduced by improving the efficiency of plant utilization of nitrogen (IPCC, 1996c). The IPCC (1996c) and Armstrong-Brown et al. (1995) review a number of ways to mitigate N_2O emissions, and the IPCC (1996c) estimates that the mitigation measures have the potential to reduce N_2O emissions by 20%. However, they point out, properly, that “farmers...will not volunteer to implement practices proposed to mitigate greenhouse-forced climate change,” and will adopt such practices only if they are convinced that they will be profitable (p. 765). Nevertheless, many of the mitigation measures may indeed be attractive economically. I therefore assume that the N_2O emission rate from the use of synthetic fertilizer declines by 0.5% per year, with the result that emissions are reduced by 10% -- half of the “potential” estimated by the IPCC -- after 20 years.

However, the rate of N_2O emission from biologically fixed and crop residue N is assumed constant over time, because, whereas farmers can control the amount and kind of synthetic N applied, and the timing and method of application, they presumably have much less control over the of biological fixation of N, and the release of N from crop residue.

My assumptions are shown in Appendix C.

N_2O emissions related to cultivation of organic soils (independent of the use of fertilizer) ($GHGN2OS_E$)

The IPCC (1997) guidelines for estimating national GHG emission inventories note that the cultivation of histosol, which is a peat-like soil with a very high organic content, can accelerate the mineralization of old, N-rich organic matter, which in turn can lead to increased N_2O emissions, independent of any application of synthetic fertilizer. The resultant N_2O emission can be quite large: the IPCC (1997) recommends a value of 5,000 g-N-N_2O/ha/yr for temperate and boreal regions, and 10,000 g-N-N_2O/ha/yr for tropical regions.

It appears, however, that very little corn or soybean is grown on histosols in the U. S. According to the *Encyclopedia Britannica*, histosols are typical in peat bogs and swamps, and in North America occur mainly beneath the coniferous forests of the Great Lakes area. (This also does not seem a likely spot for energy grass or wood plantations.) The EPA (1999c) reports an estimate that in 1982, 843,386 hectares of histosol were cultivated in the U. S. -- less than 1% of the 120 million hectares of harvested farmland in the U. S. in 1992 (Bureau of the Census, 1994). I therefore assume that only a token amount -- 1% -- of the land cultivated for corn, soybeans, wood, or grass is histosol soil. Using the IPCC (1997) recommended emission factors, the resultant average N_2O emission rate (the parameter $N2OS_E$) is $0.01 \cdot 5{,}000 \cdot 0.405$ ha/acre = 20 g-N-N_2O/ac/yr. This adds less than 1 g/mi to fuel cycle GHG emissions from the corn/ethanol fuel cycle.

Formally:

$$GHGN2OS_E = N2OS_E \cdot HF_E \cdot \frac{MW_{N2O}}{MW_{N2}} \cdot CEF_{N2O} \cdot \frac{0.4047}{PY_E} \qquad \text{eq. 74}$$

where:

$GHGN2OS_E$ = N_2O emissions from cultivation of organic soils (g-CO_2-equivalent/bu or ton).
$N2OS_E$ = N_2O emissions from organic soil, related to cultivation per se, independent of the use of fertilizer, in energy-crop system E (g-N_2O/ha/yr) (discussed above).
HF_E = of the total acreage planted in crop E, the fraction that is planted on histosol (discussed above).
0.4047 = hectares/acre.
PY_E = the into-the-plant yield, for energy crop E (bu/acre for soy and corn, net dry tons/acre for wood and grass) (discussed above).

corn, soybeans	1990-96 harvest yield from Table 19, less post-harvest loss; 1%/year increase based onWAOB (1997) and my judgment
wood, grass	current harvest yield and projected annual change from Table 20, less post-harvest losses and with adjustments for practices that produce less-than-optimal yields

All other terms are as defined above in this major section

NO_X and NH_3 related to use of synthetic nitrogen fertilizer and animal manure ($GHGNO2F_E$)

Some of the nitrogen in applied commercial N fertilizer, biologically fixed N, or atmospherically deposited N volatilizes as N in NO_X or NH_3. Data in Stohl et al. (1996), the IPCC (1997), and other sources reviewed in Appendix C, indicate that as much as 10% or more of the N in applied fertilizer volatilizes. Although a substantial portion of this volatilized N is in NH_3, we represent all volatilized N-NH_3 as N-NO_2, because we do not have a separate CEF for NH_3, and because NH_3 has many of the same effects on climate that NO_2 does: it forms particulate nitrate, and its deposition fertilizes plants and also leads to enhanced N_2O emissions.

Formally:

$$GHGNO2F_E = NO2F_E \cdot \left(AF_E \cdot \left(F_{N,SF,E} + F_{N,M,E}\right) + F_{N,FX,E}\right) \cdot \frac{MW_{NO2}}{MW_N} \cdot CEF_{NO2} \qquad \text{eq. 75}$$

$GHGNO2F_E$ = CO_2-equivalent NO_2 emissions from nitrogen in synthetic fertilizer, animal manure, or biologically fixed N in energy-crop system E (g-CO_2-equivalent/bu or ton)

$NO2F_E$ = NO_2 + NH_3 emissions per unit N in synthetic fertilizer, manure, or biologically fixed N in energy-crop system E (g-N-NO_2/g-N-fertilizer) (Appendix C)

MW_{NO2} = the molecular mass of NO_2 (46 g/mole).

MW_N = the molecular (atomic) weight of N (Divide MW of N_2 in Table 5 by 2).

CEF_{NO2} = the CO_2-mass-equivalency factor for NO_2 (Appendix D)

All other terms are as defined above in this major section.

I assume that the emission factor, NO2F accounts for off-site as well as on-site emissions. Also, I assume that theN in crop residue is *not* converted to NO_X.

CH_4 from soil due to fertilization and cultivation (parameters $GHGMF_E$, $GHGMS_E$)

Cultivation reduces the oxidation of methane in aerobic soils, and thereby increases the concentration of methane in the atmosphere (IPCC, 1996c, 1997; Appendix C to this report). Some of the reduction in soil uptake of methane is related to the use of nitrogen fertilizer, and some is related to cultivation per se, independent of the use of fertilizer. The change in methane emissions due to cultivation is a function of *both* the type of energy crop system being put into place (corn, soybeans, etc.) *and* the type of land use being displaced (range land, forest, etc.) by the energy crop system. (For example, methane emissions from corn planted over range land are different from methane emissions from corn planted over forest land and from methane emissions from soybeans planted over range land.) The reduction in methane uptake is equivalent to an emission of methane from fertilized and cultivated soils.

Formally, we estimate these CO_2-equivalent GHG emissions of methane and CO_2:

$$GHGMS_E = \frac{CH4S_E}{PY_E} \cdot 0.4047 \cdot CEF_{CH4}$$

$$GHGMF_E = \frac{CH4F_E}{1000} \cdot AF_E \cdot F_{N,SA,E} \cdot CEF_{CH4} \qquad \text{eq. 76}$$

where:

$GHGMS_E$ = CH_4 emissions from soil, related to cultivation per se, independent of the use of fertilizer, in energy-crop system E (g-CO_2-equivalent/bu or ton).

$GHGMF_E$ = CH_4 emissions related to the use of synthetic nitrogen fertilizer or animal manure in energy-crop system E (g-CO_2-equivalent / bu or ton).

$CH4F_E$ = CH_4 emissions related to the use of synthetic nitrogen fertilizer or animal manure in energy-crop system E (g-CH_4 / kg-N-fertilizer); data discussed in Appendix C suggest the following input values:

Corn	Grass	Wood	Soy
0.1	10	10	1.0

$CH4S_E$ = CH_4 emissions from soil, related to cultivation per se, independent of the use of fertilizer, due to energy-crop system E (g-CH_4 / ha / yr); parameter values presented here and in Appendix C result in the following calculated values:

Corn	Grass	Wood	Soy
435	180	85	335

CEF_{CH4} = the CO_2-equivalency factor for methane (Appendix D)

1000 = g / kg

All other terms are as defined above in this major section.

The parameter $CH4S_E$ is calculated as the weighted-average CH_4 emission rate over all types of displaced land uses:

$$CH4S_E = \sum_D FD_{E,D} \cdot CH4S_{E,D} \qquad \text{eq. 77}$$

where:

subscript D = types of land uses displaced by energy-crop system E (forests, grassland, generic agriculture, desert, nothing).

$CH4S_{E,D}$ = CH_4 emissions from soil, related to cultivation of energy-crop system E instead of land-use type D (g-CH_4 / ha / yr; discussed in Appendix C).

$FD_{E,D}$ = acres of land-use type D ultimately displaced per acre of energy system E (discussed elsewhere in this section).

As indicated below, these methods and assumptions result in a minor contribution to total fuel cycle CO_2-equivalent GHG emissions. Although there is considerable uncertainty in these assumed parameter values, they would have to be low more than an order of magnitude in order to have a significant impact on fuel cycle emissions.

CO_2 emissions from on-site soil due to N fertilization (parameter $CO2SF_E$)

The use of fertilizer also can affect the carbon content of the soil, mainly if not exclusively by affecting the oxidation of carbon in the soil (Appendix C to this report; IPCC, 1996c; Fog, 1988). The limited data reviewed in Appendix C indicate that N fertilization can increase or decrease carbon oxidation and increase or decrease the carbon content of the soil, depending on the crop, fertilizer, soil type, environmental conditions, and other factors. On balance, the data support an assumption that N fertilization reduces the rate of carbon oxidation in the soil and increases the carbon content of soil. A reduction in the rate of oxidation of soil carbon or an increase in the carbon content of the soil is tantamount to a reduction in emissions of CO_2 from soil.

We distinguish between the effect of nitrogen on soil carbon at the site of fertilizer application (estimated in this section), and the effect of run-off nitrogen on soil carbon off site (included in the estimate of the next section). Formally, we estimate CO_2 emissions from soil due to N fertilization on site as.

$$CO2SF_E = CO2SF^*_E \cdot F_{N,SA,E} \cdot AF_E \qquad \text{eq. 78}$$

where:

$CO2SF_E$ = CO_2 emissions from soil, related to the use of synthetic nitrogen fertilizer or animal manure in energy-crop system E (g-CO_2/bu or ton).

$CO2SF^*_E$ = CO_2 emissions from soil, related to the use of synthetic nitrogen fertilizer or animal manure in energy-crop system E (g-CO_2/g-N-fertilizer; we assume – 1.0 [a negative emission rate] on the basis of data discussed in Appendix C).

All other terms are as defined above in this major section.

The effect of nitrogen fertilization on the storage of carbon in off-site biomass and soil (parameter $CO2NF_{EO}$)

The nitrogen fertilizer (synthetic or from animal manure) that leaches and runs off from agricultural fields eventually will eutrophy freshwater and marine ecosystems. The extra nutrient will stimulate plant growth and hence CO_2 uptake; the CO_2 uptake is equivalent to a reduction in CO_2 emissions. The leached nitrogen also may affect the rate of oxidation and hence storage of carbon in off-site soils (see discussion in previous section, and in Appendix C). The overall effect is analogous to that due to fertilization by nitrogen deposition from ambient NO_X (Appendix C), and can be estimated on the basis of some of the same data:

$$CO2NF_{EO} = \frac{MW_{CO2}}{MW_C} \cdot F_{N,SFA,E} \cdot NLF_E \cdot \left(1 - NO2F_E - N2OIF_E\right) \cdot \sum_{EO} R_{E,EO} \cdot C(CO2)_{AIR} / ND_{EO} \qquad \text{eq. 79}$$

where:

subscript EO = types of off-site ecosystems fertilized by nitrogen leaching (marine, freshwater, terrestrial [ground]).

MWCO2 = the molecular mass of CO2 (Table 5).

MWC = the molar mass of C (12.01 g / mole).

$CO2NF_{EO}$ = CO2 sequestered in plants and soils fertilized by nitrogen (from synthetic fertilizer or animal manure) that leaches off site of energy-crop system E (g-CO2-sequestered / bu or ton).

$R_{E,EO}$ = of nitrogen that runs off of energy-crop system E, the fraction that is deposited in ecosystem EO (see IPCC [1996c, 1997] and Appendix C).

$C(CO2)_{AIR} / ND_{EO}$ = net grams of carbon emitted or taken from the air as CO2, per year, per gram of nitrogen leached to ecosystem EO per year (see discussion of CEF for NOx in Appendix D; includes effects on soil carbon as well as effect on biomass carbon).

All other terms are as defined above in this major section.

Note that the amount of nitrogen available for fertilizing off-site ecosystems is equal to the amount runoff of field E less the amount lost as N_2O, NO_X, or NH_3 offsite.

The calculated value of CO2NF, which is tantamount to a negative CO_2 emission, turns out to be quite substantial: it cancels approximately 50% of the N_2O emissions from fertilizer. However, all of the key parameter values are uncertain, especially in the long run. The importance and uncertainty of this effect make it a critical area for additional research.

The nitrogen that leaves the field and fertilizes off-site ecosystems is not available to fertilize the field crops it was intended for. Thus, the greater the off-site loss of nitrogen, the less the on-site yields and the less standing biomass. Ideally, the on-site harvest yields, and the quantity of above-ground biomass (a parameter in the estimation of changes in carbon sequestration due to changes in land use), would be related formally to the fraction of nitrogen fertilizer that is retained on the site. If it is not possible to construct a formal relationship, then one at least should make check that the assumed nitrogen loss rate is consistent with the assumed on-site yields, and that the rate of change in the nitrogen loss rate is consistent with the rate of change of the yield. We have made crude consistency checks of this sort.

Changes in carbon in soil and biomass, due to cultivation and other changes in land use (independent of the use of fertilizer) (parameter $CO2C_E$)

The establishment and operation of energy-production systems changes land use. For example, surface coal mining destroys vegetation and disturbs soil, and energy crop systems (such as woody biomass) generally displace other kinds of biomass. The changes in the above-ground biomass and in the soils generally result in changes in the amount of carbon removed from the atmosphere and sequestered in the biomass and soil.

Soil. It is well established that cultivation and disturbance reduces the carbon content of soils (Appendix C to this report; IPCC, 1996c; Appendix K of DeLuchi, 1993; Mann, 1986). Generally, soils in natural forests contain more carbon per acre than do shrub land and grassland soils, which in turn contain more carbon than crop soils (IPCC, 1996c, 2000, 2001; Table K.12 of DeLuchi, 1993). The conversion of forest soils to permanent agriculture increases the oxidation of the organic matter in the soil, and, over the course of a few years, decreases its carbon content by about 40-50%. Even the conversion of range land to crop land can reduce the carbon content of the soil by 20-40% in a relatively brief period. If farming stops and the forest recovers, soil carbon will return to near its original level, but as long as the land is cultivated, the soil will contain 40-50% less carbon per acre than before.

The IPCC assessmen reports (1996c, 2000, 2001) report review of studies of the long-term loss of loss of carbon from soil as a result of cultivation. The loss is a function of the type of ecosystem displaced, local precipitation, temperature, biological activity, soil type, and other factors, and can span a range of two orders of magnitude, from 0.1 to 5 kg C per square meter[53]. The data presented in the IPCC (1996c) indicate a mean loss of about 3 kg-C/m^2 globally; the meta-analysis of Mann (1986) indicates 1 kg-C/m^2 for North America. Presumably, the loss globally is higher than the loss in North America because globally more forest is cleared for agriculture, and because conservation management practices, which can reduce carbon losses by 50% or more, are better in North American than elsewhere.

Energy crop systems, such as switch grass or SRIC poplar plantations, will reduce the carbon content of the soil if they replace forests, but increase soil carbon content if they replace traditional row crops such as corn. Fossil-fuel production systems, such as surface coal mining, that clear the land and thoroughly disturb soils presumably cause large losses of soil carbon.

Biomass. Plants growing remove carbon from the atmosphere, and plants decaying (oxidizing) release carbon back to the atmosphere or soil. As long as growth exceeds decay, as it does in a newly planted energy-crop system, the ecosystem on balance will transfer carbon from the atmosphere to the plants, and thereby increase the

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[53] Roberts and Chan (1990) distinguished carbon loss via oxidation due to increased microbial respiration stimulated by soil disturbance from carbon loss due to erosion and other means. They found that “the losses of organic matter owing to soil disturbance resulting from cultivation are small compared with other mechanisms of loss...[such as] losses due to wind and water erosion and apparent losses due to mixing of organic matter into deeper layers of the soil” (p. 150).

standing stock of carbon in the plant biomass. This increase in the plant carbon will continue until the ecosystem reaches equilibrium, at which point the release of plant carbon back to the atmosphere, as a result of oxidation (e.g., gradual decay, or fuel combustion), will balance the uptake of carbon by new growth. In the long-run or indefinite equilibrium, then, there will be a more-or-less constant amount of carbon in the plant biomass, with carbon oxidation and uptake in rough balance.

The constant amount of carbon in the plant biomass, built up during the growing phase, can be viewed as a one-time, short-term, negative emission of CO_2. The amount of carbon so sequestered varies from ecosystem to ecosystem: mature forests contain much more carbon per acre than do, say, soybean crops. The *change* in the carbon content of standing biomass due to the marginal production of switchgrass, trees, corn, or soybeans, or the complete destruction of the existing vegetation (as in surface coal mining), should be counted as a one-time change in CO_2 (a negative or positive emission) attributable to the energy-production system (coal, switchgrass, etc.).

It is important to note that neither the ultimate fate of the biomass -- whether gradual decay, immediate combustion, or conversion to a different fuel -- nor the frequency of harvest (or the time to equilibrium) materially affect the conceptual outline above. A system in which switchgrass is harvested once a year for conversion to ethanol is conceptually no different from a mature climax forest in which plants grow, die, and decay naturally, over decades: in both systems, there is some period of initial growth and net carbon fixation in the plant biomass, ending when the oxidation of the biomass roughly balances the regeneration or replanting[54]. In the case of switchgrass, the first planting grows for a year, and removes net carbon from the atmosphere, until the first harvest. At harvest, the mature grass is removed, converted to ethanol, and eventually burned. Thus, the harvest, like the natural decay in the forest, returns the fixed carbon to the atmosphere. At the same time, however, the next grass planting removes carbon from the atmosphere, roughly in pace with the oxidation of the first planting -- just as in the forest, regeneration and re-growth removes CO_2 from the atmosphere while decay is releasing CO_2.

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[54] Note that, at any time in the system, carbon is stored in the yet-to-be-burned biofuel as well as in the biomass feedstock. In fact, on average, of the total amount of CO_2 removed from the atmosphere by the system in the equilibrium, about half is in the biomass feedstock, and half is in the yet-to-be-burned biofuel. Recall that the total amount of CO_2 removed in the equilibrium is the amount sequestered (as carbon) in the initial mature growth of the biomass. Now, once we have begun the cycle of harvesting and replanting and re-harvesting the biomass, the density of the biomass system at any one time (in kg-C / m^2) will be about half the density of the harvested biomass, because some areas will be newly replanted (with close to 0 kg-C / m^2), some areas close to harvest (at close to the kg-C / m^2 level of harvested biomass), and most areas in between. But simultaneously, on average, only about half of the biofuel made from the harvested biomass has been burned to return its CO_2 to the atmosphere -- the other half retains its carbon, now as part of a motor fuel. Figure 6 illustrates this.

Figure 6 shows the basics of this process. See also Hakamata et al. (1997), Houghton et al. (1983), and especially (IPCC, 2000). For a general discussion of methodological issues in the estimation of the GHG impacts of biomass systems, see Schlamadinger et al. (1997).

What we have, then, in any system, is a one-time removal of CO_2 from the atmosphere, by the initial growth to equilibrium, followed by an indefinite period of balance between uptake and release of atmospheric carbon. In the switchgrass system, the first harvest ends the growth phase and begins the indefinite equilibrium phase; in the forest, natural processes govern. Thus, beginning with the equilibrium (or first harvest) and continuing indefinitely thereafter, there is (to a first approximation) no *further* change in net CO_2 flux, and the long-term net effect is the initial, short-term removal of CO_2.

Of course, the frequency of harvest or time to equilibrium, in concert with the growth rate, does determine the amount of carbon sequestered in the plant biomass, in equilibrium. In a switchgrass system, the carbon is built up for but a year before harvest and equilibrium; in a forest, the carbon is built up over decades.

<u>Method of analysis.</u> To estimate the change in the carbon content of the soil and plant biomass, due to cultivating corn, soybeans, switchgrass, or poplar, or removing vegetation for surface coal mining, one must know what biomass is being displaced by the energy-production system. To my knowledge, there is no model of land-use and energy systems sufficiently detailed, and properly specified with characteristics of energy-crop systems (switchgrass or trees), to project changes in land use, over the long run, due to the introduction or expansion of energy-crop or fossil-fuel systems. Consequently, one must rely on expert opinion, partial models, and other sources to estimate the changes in land use. This is unfortunate, because the change in land use is the key parameter in the estimation of the change in carbon sequestration in soil and biomass.

The change in the carbon content of the soil and biomass (the difference between the carbon content of the new system, and the carbon content of the displaced system) usually occurs over a few decades (IPCC, 1996c). This initial short-term change must be converted to an equivalent annual change over the life of the crop-to-energy program, for proper comparison with, and addition to, the other emissions streams in the analysis (such as emissions from the fuel-production facility). The best way to do this is to convert the short-term initial change to an equivalent instantaneous change at the beginning of the program, and then to annualize the equivalent instantaneous change over life of the crop-to-energy program. Also, on the assumption that the initial short-term change is reversed when the program is abandoned (i.e., assuming that the land reverts to its pre-program use, and gains or loses the amount of carbon originally lost or gained as a result of the initial change), the present value of the reversal should be deducted from the initial change.

We now can specify our formal model of CO_2 emissions from soil and biomass, due to land-use changes attributable to energy system E:

$$CO2C_E = \left(A\Delta CS_E + A\Delta CB_E\right)\cdot \frac{MW_{CO2}}{M_C}\cdot 1000 \cdot \frac{4047}{PY_E} \qquad \text{eq. 80}$$

where:

$CO2C_E$ = net CO_2 emission or sequestration in soil or biomass as a result of changing land uses to energy-crop system E (g-CO_2/bu or ton).
$A?CS_E$ = the change in carbon content of the soil due to energy system E, annualized over the life of the energy program (kg-C/m^2/yr).
$A?CB_E$ = the change in carbon content of the plant biomass due to energy system E, annualized over the life of the energy program (kg-C/m^2/yr).
MW_{CO2} = the molecular mass of CO_2 (Table 5).
MW_C = the molar mass of C (12.01 g/mole).
1000 = grams/kg
4047 = square meters per acre.
PY_E = the into-the-plant yield, for energy crop E (bu/acre/yr for soy and corn, net dry tons/acre/yr for wood and grass) (discussed above).

The actual initial short-term change in the carbon content of the soil ($?CS_E$) or plant biomass ($?CB_E$) occurs over some period, which is LS for soil and LB for biomass. I assume that this initial carbon change is linear over the period of actual carbon change, so that the estimated initial annual rate of change is given by $?CS_E$/LS and $?CB_E$/LB. Now, this estimated rate, over the period LS or LB, must be converted to an equivalent annualized change over the life of the energy program (LP), which could be indefinite. To do this, we take the present value of the actual annual carbon change (over the period LS or LB), and then annualize the present value over the life of the program LP. (This procedure is necessary because in general LS and LB will differ from LP. For example, the change in carbon content can occur over a relatively short period of time, as little as a year or so, whereas the energy program itself can last indefinitely. But before we annualize the present value of the initial annual carbon change, we must deduct the present value of the carbon change that we get back at the end of the energy program (after period LP), when the carbon change is reversed. Assuming that the reversal, upon reversion of the land to its original use, is just the negative of the original change, and noting that in this analysis the discount rate is a function of the time period (see Appendix D), we have:

$$A\Delta CS_E = \left(\frac{\Delta CS_E}{LS} \cdot PVS - \frac{\Delta CS_E}{LS} \cdot PVS \cdot (1 + r_{LP})^{-LP} \right) \cdot AMT$$

eq. 80a

$$= \frac{\Delta CS_E}{LS} \cdot PVS \cdot \left(1 - (1 + r_{LP})^{-LP}\right) \cdot AMT$$

$$A\Delta CB_E = \left(\frac{\Delta CB_E}{LB} \cdot PVB - \frac{\Delta CB_E}{LB} \cdot PVB \cdot (1 + r_{LP})^{-LP} \right) \cdot AMT$$

eq. 80b

$$= \frac{\Delta CB_E}{LB} \cdot PVB \cdot \left(1 - (1 + r_{LP})^{-LP}\right) \cdot AMT$$

where the present-value (PV) and amortization (AMT) terms are:

$$PVS = \frac{1 - (1 + r_{LS})^{-LS}}{r_{LS}}$$

$$PVB = \frac{1 - (1 + r_{LB})^{-LB}}{r_{LB}}$$

eq. 80 c - e

$$AMT = \frac{r_{LP}}{1 - (1 + r_{LP})^{-LP}}$$

where:

?CS_E = carbon change in soil due toenergy system E.

?CB_E = carbon change in plant biomass stock due to energy system E.

LS = the period over which the soil carbon changes.

LB_D = the period over which the carbon in the plant biomass changes.

PVS = the present-value factor for soil carbon change (converts the actual short-term, multi-year carbon change into an equivalent instantaneous year-zero change).

PVB = the present-value factor for biomass carbon change (converts the actual short-term, multi-year change into an equivalent instantaneous year-zero change).

AMT = the amortization term, to convert the year-zero change in carbon to an equivalent yearly change over the life of the energy crop program (note that the amortization term annualizes over the life of the energy crop program, LP, whereas the present value terms apply to the lifetime of the initial carbon changes, LB and LS).

LP = the life of the energy crop program..
r_{XX} = the discount rate for time period XX (LS, LB, or LP).
other terms defined for eq. 80.

Note that the denominator of AMT in eq. 80e cancels the $\left(1-(1+r)^{-LP}\right)$ term in eq 80a and 80b. Note too that if the discount rate *r* were constant (as it is in most analyses) rather than time-varying (as it is in this analysis), then the *r* in the numerator of AMT in eq. 80e would cancel the *r* in the denominator of PVS and PVB in eq. c and d, with the result that all terms with LP would disappear, and the life of the program (LP) would not have to be specified. However, a consequence of using a time-varying discount rate is that the life of the program, LP, must be specified, if only for the purpose of estimating r_{LP}.

Substituting eq. 80c-e into eq. 80a and 80b, we get:

$$A\Delta CS_E = \frac{\Delta CS_E}{LS} \cdot \left(1-(1+r_{LS})^{-LS}\right) \cdot \frac{r_{LP}}{r_{LS}} \qquad \text{eq. 80f}$$

$$A\Delta CB_E = \frac{\Delta CB_E}{LB} \cdot \left(1-(1+r_{LB})^{-LB}\right) \cdot \frac{r_{LP}}{r_{LB}} \qquad \text{eq. 80g}$$

Substituting eq. 80f and 80g into eq. 80, we get:

$$CO2C_E = \left(\frac{\Delta CS_E}{LS} \cdot \frac{\left(1-(1+r_{LS})^{-LS}\right)}{r_{LS}} + \frac{\Delta CB_E}{LB} \cdot \frac{\left(1-(1+r_{LB})^{-LB}\right)}{r_{LB}} \right) \cdot r_{LP} \cdot \frac{MW_{CO2}}{MW_C} \cdot \frac{4.047 \cdot 10^6}{PY_E}$$

Recognizing now that the carbon-change parameters CS and CB depend on the characteristics of the new energy-crop system E and of the displaced land use D, that the duration of the changes LS and LB depend mainly on the characteristics of the displaced land use D, and that the life of the energy crop program LP depends only the type of program E, we have::

$$CO2C_{E,D} = \left(\frac{\Delta CS_{E,D}}{LS_D} \cdot \frac{\left(1-(1+r_{LS_D})^{-LS_D}\right)}{r_{LS_D}} + \frac{\Delta CB_{E,D}}{LB_D} \cdot \frac{\left(1-(1+r_{LB_D})^{-LB_D}\right)}{r_{LB_D}} \right) \cdot r_{LP_E} \cdot \frac{MW_{CO2}}{MW_C} \cdot \frac{4.047 \cdot 10^6}{PY_E}$$

eq. 81

where the ?CS and ?CB terms are the difference between the carbon content of the displaced land use D and the carbon content of energy system E:

$$\Delta CS_{E,D} = CS_D - CS_E$$
$$\Delta CB_{E,D} = CB_D - CB_E$$
eq. 82a

The CO_2 emissions by displaced land-use type are aggregated over the displaced land uses to get the entire CO_2 effect for the energy crop system E:

$$CO2C_E = \sum_D FD_{E,D} \cdot CO2C_{E,D}$$
eq. 82b

where:

subscript D = land-uses displaced by energy-system E (tropical forests, temperate forests, boreal forests, tropical grasslnds, temperate grasslands, desert, tundra, wetland, generic agriculture, and low-intensity cultivation of the same type as energy system E).

$CO2C_{E,D}$ = net CO_2 emission or sequestration in soil or biomass as a result of replacing land-use type D with energy-crop system E (g-CO_2/bu or ton).

?$CS_{E,D}$ = carbon change in soil due to replacing land use type D with energy system E (kg-C / m^2; negative value means carbon gain).

?$CB_{E,D}$ = carbon change in plant biomass stock due to replacing land-use type D with energy system E (kg-C / m^2; negative value means carbon gain)

LS_D = the period over which the soil carbon in displaced land-use type D actually changes (years; discussed below).

LB_D = the period over which the carbon in the plant biomass in displaced land-us type D actually changes (years; discussed below).

LP_E = the life of the energy crop program E (years); we assume the following:

corn	grass	wood	soybeans
30	40	50	30

r_{LSD} = the discount rate at time = LS_D (recall that in this analysis the discount rate is a function of time; see Appendix D for details).

r_{LBD} = the discount rate at time = LB_D (recall that in this analysis the discount rate is a function of time; see Appendix D for details).

CS_D = carbon in soil on displaced land-use type D (kg-C / m^2) (Appendix C).

CS_E = carbon in soil of energy system E (kg-C / m^2) (Appendix C).

CB_D = carbon in plant biomass on displaced land-use type D (kg-C / m^2) (Appendix C).

CB_E = carbon in plant biomass of energy system E (kg-C / m^2) (discussed below).

$FD_{E,D}$ = acres of land-use type D ultimately displaced per acre of energy system E (discussed below).

All other terms are as defined above in this major section.

Thus, the life of the crop-to-energy program, LP is a parameter in the final model *only* because we use a time-varying discon rate; otherwise, it is not a factor. This can be understood intuitively: the longer the energy program, the greater the number of years over which the carbon change is annualized, but the less the reversion credit at the end of the period. It turns out that with a constant discount rate these two opposing factors cancel, so that the period LP does not matter.

Note the effect of the discount-rate parameter *r* in Eq. 81. If the discount rate is zero, then the parameter $CO2C_E$ (CO_2 lost from soils and biomass, due to energy system E, per unit of feedstock produced) in Eq. 81 is zero. This is because with a zero discount rate, the present value of the change in the in carbon sequestration at the beginning of the program is the same as the present value of the reversal at the end. Put another way, when the discount rate is zero, we don’t assign any value to a merely *temporary* change in emissions.

Conversely, if the discount rate is very large, then we don't care at all about the future reversion of land use and reversal of the initial change in emissions; we care only about initial change in land use and emissions. With a large discount rate, the parameter $CO2C_E$ in Eq. 81 reduces to the initial rate of change in emissions or sequestration.

In sum, the discount rate determines the value of the reversal of the initial change: a zero discount rate gives it a value equal to that of the initial change; a high discount rate gives it no value.

Adaptation for coal mining. In the case of land-use emissions due to coal mining, in order to express the results per energy-unit of coal produced, rather than per acre of land impacted, we have to add the following multiplicative term to Eq. 81:

$$\frac{AT_{coal,T}}{HHV_{coal}}$$

$$AT_{coal,T} = \frac{AT_{coal,S} \cdot TP_{coal,S,T} + AT_{coal,U} \cdot TP_{coal,U,T}}{TP_{coal,S,T} + TP_{coal,U,T}} \qquad \text{eq. 83}$$

where:

$AT_{coal,T}$ = acres of land disturbed per ton of coal produced, in year T.

HHV_{coal} = the higher heating value of coal (10^6 BTU/ton).

$AT_{coal,S}$ = acres of land disturbed per ton of coal produced from surface mines (see discussion below).

$AT_{coal,U}$ = acres of land disturbed per ton of coal produced from underground mines (assumed 1/4 of the value for surface mines).

$TP_{coal,S,T}$ = tons of coal produced from surface mines in year T (estimate based on projections in EIA's *AEO*).

$TP_{coal,S,U}$ = tons of coal produced from underground mines in year T (estimate based on projections in EIA's *AEO*).

The key parameter in the coal analysis is acres of land disturbed per ton of coal produced from surface mines. This statistic can be calculated from data on the acreage and production of Federal coal-mining leases, reported in the EIA's *Coal Industry Annual 1995* (1996). In 1995, 293,310 acres of Federal leases west of the Mississippi River produced 348 million short tons of coal. Assuming that all of the land leased was disturbed, and that all Federal leases west of the Mississippi were for surface mines (in 1995, surface mines west of the Mississippi produced 444 million tons -- more than 90% of the total production of 489 million tons [EIA, *Coal Industry Annual 1995*, 1996), we calculate 0.00084 acres of land disturbed per ton of coal produced from surface mines.

For the purpose of calculating r_{LP} in eq. 81, we assume that the coal mining lasts 45 years.

Carbon content of plant biomass. The kg-C / m^2 carbon content of the plant biomass in the "displaced" ecosystems is estimated in Appendix C. The carbon content of corn and soybean plants is estimated on the basis of the yield weight, residue and root weight, and carbon fraction:

$$CB_E = SY_E \cdot WB_E \cdot MCF \cdot RR_E \cdot CF_E \cdot K1 \qquad \text{eq. 84}$$

CB_E = carbon content of crop E (kg-C / m^2).
SY_E = standing annual-average yield of crop E (bu / acre; Table 19 and discussion elsewhere).
WB_E = weight per bushel of crop E (56 lbs / bu-corn, 60 lbs-bu-soybeans, at 15% moisture).
MCF = moisture correction factor, to get to dry weight (0.85)
RR_E = ratio of total plant biomass (crop, residue, roots) to harvested crop mass (see Eq.70).
CF_E = carbon weight fraction of dry crop E (see"sulfur content, carbon content, and heating value of biomass").
K1 = conversion factor from lbs / acre to kg / m^2 = 1 / 2.205 / 4047 = 0.000112.

These values nominally include the carbon in roots and plant litter, which carbon is not included in the estimates above of soil carbon.

The value for generic agriculture is calculated as a weighted average of the corn and soybean values, assuming 0.50 weights on each crop.

The carbon content of the standing biomass in poplar or switchgrass energy crop systems is estimated in a similar manner, as the product of the annual yield, the years of growth, and the carbon weight fraction:

$$CB_E = SY_E \cdot WB_E \cdot MCF \cdot RR_E \cdot CF_E \cdot K1 \qquad \text{eq. 85}$$

where:

CB_E = carbon content of energy crop E (kg-C / m^2).
SY_E = standing annual-average dry yield of crop type E (tons / acre / year) (based on the data of Table 20; see discussion elsewhere).[55]

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[55] The annual average yield is equal to the amount actually harvested divided by the years from initial planting to harvest. Thus, the amount actually harvested is equal to the annual average yield SY_e multiplied by the years from initial planting to harvest YH_e.

YH_E = years of growth, from initial planting to harvest (1 year for switchgrass, 6-10 years for poplar, depending on the region [Walsh, 1998a]; I estimate a production-weighted average of about 9.5 years).

CF_E = carbon weight fraction of dry energy crop E (see "sulfur content, carbon content, and heating value of biomass").

RR_E = scaling factor for unharvested leaf, litter, and roots from energy crop E, not included in the standing yield estimates (see Eq. 72).

K2 = conversion factor from tons / acre to kg / m^2 = 2000 / 2.205 / 4047 = 0.224.

Note that this method does not *explicitly* account for the stimulatory effect of nitrogen fertilization on carbon content. In principle, nitrogen fertilization affects the standing yield (SY), which in turn affects the carbon content of the biomass (see Eq.85), but this relationship between nitrogen and standing yield is not represented in Eq. 85. However, even though we have not modeled this N-yield function formally, we have tried to ensure that our assumptions about yields are consistent with our assumptions about fertilization.

The period over which the carbon content of the biomass and soil changes. The period over which the biomass or soil carbon content of an ecosystem changes can vary widely, from a less than a year to many decades, depending on what replaces what, and how. Generally, as shown in Figure 6, the loss of carbon that occurs as a result of conversion to agriculture occurs more quickly than does the build up of carbon that occurs after abandonment of agriculture and reversion to native ecosystems. For example, if a forest is slashed and burned to make way for agriculture, the carbon content of the plant biomass changes very quickly. Similarly, the carbon content of the soil changes quickly at first, and then more gradually after 20 years or so: Mann (1986) states that "authors in recent years have suggested that soils converted from native vegetation to permanent cropping lose organic matter rapidly in the first years of cultivation and continue to lose carbon at a slower rate, approaching a new equilibrium after 30 to 50 years (p. 279). His own meta-analysis of more than 50 studies indicates that most of the loss occurs with in the first 20 years (Mann, 1986). Agriculture and Agri-food Canada (1997) suggest that after a change from conventional to no-till agriculture, the carbon content of the soil will reach a new equilibrium after 10 to 25 years. Lal (2003) states that after cultivation most soils losee 1 / 2 to 2 / 3 of their soil organic carbon within 5 years in the tropics and 50 years in temperate regions (p. 440).

By contrast, it takes many decades after abandonmen of cultivation for soil carbon and vegetation carbon to return to their original (pre-cultivation) equilibrium levels. In their model of the terrestrial carbon cycle, Houghton et al. (1983) assume that 50 years after abandonment of agriculture recovered forests have 75% of the vegetation C and 90% of the soil C of undisturbed forests, and that recovered grasslands and shrublands have 100% of the vegetation C and 100% of the soil C of undisturbed ecosystems. Similarly, Robles and Burke (1996) state that active pools of soil organic matter (SOM) can recover to native levels for grasslands about 50 years after abandonment of agriculture. However, Burke et al. (1995) give a more nuanced view, stating that on cropland abandoned to grassland) 50 years is an adequate time for recovery of active SOM and nutrients, but that the recovery of *total* SOM is a much slower process (p. 793).j

With these considerations, and allowing that carbon contents change more rapidly in warm wet regions than in cool dry regions, I assume the following values for the period over which the carbon content of soil (LS_D) and biomass (LS_B) changes, by land-use type (years):

	Tropical forest	*Temperate forest*	*Boreal forest*	*Tropical grass*	*Temperate grass*	*Desert*	*Tundra*	*Wet-land*	*Generic agriculture*	*Low intensity*
LS_D	6	25	30	10	22	40	50	20	20	20
LB_D	4	15	18	6	10	15	20	5	3	3

The results are somewhat sensitive to this parameter: assuming for illustrative purposes a 2% discount rate (see Appendix D for a discussion of the actual time-dependent discount rate assumed in this analysis), a five-fold increase in the period, from 10 years to 50 years, results in a 30% decrease in the estimated annualized carbon change ($kg\text{-}C / m^2 / yr$). An increase from 15 years to 35 years results in a 17% decrease in annualized carbon change.

Changes in land use. Because of the large differences in the carbon sequestration of forest versus grass versus crop systems, the distribution of the land displaced by a new energy fuel program is perhaps the most important parameter in the analysis of GHG emissions due to changes in land use. Unfortunately, the interplay of economic, technological, political, regulatory, environmental, and historical forces is particularly difficult to model in this case. I do not attempt a formal model here.

It will, however, be useful at the outset to defend the proposition that a biofuel (produced from corn or soybeans) will bring new land (i.e., land that would not otherwise be cultivated) into production. I will do this by rebutting the two counterarguments that might be made as regards expanded production of corn and soybeans.

First, one might argue that the amount of land for agriculture simply is fixed, so that any increase in corn or soybean land will come at the expense of land for other crops, and not result in a net increase in cultivated land. However, it is clear that the amount of land is not fixed absolutely by nature or regulation. Also, economic forces can not actually fix the amount of land: although the higher crop prices (which result from the shift in demand induced by extra demand for ethanol or biodiesel) will suppress consumption of corn and soybeans for other uses, the suppression of other uses generally will be less than the increase due to the biofuel program. This is shown in Figure 4: a shift in demand from Q’ to Q results in a net increase in consumption of Q − Q*, which is less than the shift Q − Q’, but greater than zero. Moreover, one should also consider that at least some of the consumption squeezed out (Q* – Q’) by the higher price might have found substitutes in other sectors.

Second, one might argue that an increase in demand for corn or soybeans (due to increased demand for ethanol or biodiesel) will spur an increase in per-acre yields that would not have happened otherwise, with the result that at least some of the additional crops will be grown on existing acreage rather than new land[56]. That corn production has grown somewhat while harvested acreage has not over the past 20 years might be taken as evidence in favor of this proposition. However, there is much year-to-year variation: often, harvested acreage has increased with production, and in a few cases, harvested acreage has increased by a greater percentage than has production. Moreover, it is not necessarily the case that increases in yields are driven by increases in demand (outward shifts of the demand curve). The alternative proposition -- that increases in yield shift the supply curve out, reduce price, and spur additional consumption -- is at least as plausible. Indeed, the long-term decline in the real price of corn from 1951 to 1996 is evidence that supply-side improvements have reduced price and stimulated consumption. (If the market were driven primarily by shifts in demand, real prices would have risen.) Consistent with the proposition that increased output results from improved yields, the World Agricultural Outlook Board (WAOB, 1997) projects declining real prices and increasing harvested acreage for corn through the year 2005[57].

If we accept, then, that expanded production of corn, soybeans, wood, and grass will occur at least to some extent on “new” land, the question becomes: what new land? In the U. S., it seems reasonable to assume that the displaced land will be a mix mainly of CRP, pasture, fallow, and crops. For example, Perlack et al. (1992) assume the following distribution of land displaced by energy crops (SRIC wood, grasses, energy cane):

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[56] The argument would be that , in response to an expansion of demand, the marginal productivity of increasing yields is higher than the marginal productivity of new land.

[57] Note, too, the squeezing more crops out of an acre of land probably will slightly reduce the carbon content of the soil. I have accounted for this by assuming that any crops grown on land already in production, by increasing the yield, will reduce the carbon content of the soil by 0.1 kg-C / m^2.

Energy crop --> Land-use displaced:	Poplar in Oregon	Grasses in Nebraska	All energy crops
Corn, soybeans, other crop	6%	71%	51%
Closecrop	42%	11%	13%
Fallow, range, pasture, hay	44%	18%	31%
Forests	8%	0%	5%

It appears, however, that Perlack et al. (1992) did not actually model net displacement, in the final equilibrium, but rather first-order land uses. That is, apparently they assumed that 71% of the acreage planted in energy-crop grass in Nebraska would be land that now is used for corn or soybeans, but they did not worry about whether the initially displaced corn or soybeans might be grown somewhere else, on "new" (not-otherwise-cultivated) land. I believe that in the net equilibrium, less crop land and more range or pasture land, and perhaps even a bit more forest (somewhere) will be (or has been) displaced.

Finally, the distribution of displaced land undoubtedly depends greatly on the total extent of the displacement, and probably on whether one is analyzing a marginal increase or a marginal decrease in consumption and production.

My own assumptions distinguish ten categories of displaced land uses (including "low intensity," which refers to increasing the productivity on existing land grown already for the same crop), four crops for biofuels (corn, soybeans, grass, and SRIC wood), plus coal mining:

	Tr. forest	*Tem. forest*	*Bor. forest*	*Tr. grass*	*Tem. grass*	*Desert*	*Tundra*	*Wetland*	*Generic ag*	*Low intensity*
Corn	0.00	0.03	0.03	0.00	0.55	0.05	0.00	0.03	0.12	0.19
Grass crop	0.00	0.03	0.03	0.00	0.55	0.05	0.00	0.05	0.10	0.19
SRIC wood	0.00	0.03	0.03	0.00	0.65	0.05	0.00	0.05	0.15	0.04
Soybeans	0.00	0.03	0.03	0.00	0.60	0.05	0.00	0.05	0.10	0.14
Coal	0.00	0.10	0.10	0.00	0.60	0.05	0.00	0.05	0.10	0.00

Because most surface coal mining occur in the West, I assume that the bulk of the land disturbed by coal mining is range land, followed by desert.

The results. The change in carbon sequestration due to changes in land use can significantly affect fuel cycle CO_2-equivalent GHG emissions. The following shows the difference in CO_2-equivalent g/mi fuel cycle emissions (excluding materials manufacture and vehicle assembly) with and without emissions related to land use, for the four different biofuel feedstocks considered in this analysis. (All results are for a 7 mpg diesel vehicle in the U. S. in 2010.) Analysis of results (focusing on changes in soil carbon, which are an order of magnitude larger than are changes in biomass) reveals some interesting effects:

	corn/ ethanol	*soy/ biodiesel*	*grass/ ethanol*	*wood/ methanol*
with land-use changes	3,467	8,297	2,024	1,202
without land-use changes	2,565	3,079	1,282	673
difference	902	5,218	742	529
% change vs. w.o.	26%	63%	37%	44%

In all cases, changes in soil carbon due to changes in land use are a significant part of lifecycle GHG emissions for biofuels. Generally, the changes in soil carbon are large because all bio-feedstocks are assumed to displace mainly grasslands, which have higher soil carbon than do managed biocrop lands. Howver, the small amount of wetlands assumed to be displaced also has a significant impact because of the extremely large carbon content of wetlands.

The use of soybeans as a biofeedstock results in especially large emissions from land use changes, mainly because it takes almost 4 times as many acres of soybean to produce a BTU of biodiesel as it does acres of corn to produce a BTU of ethanol.

Finally, the CO_2-equivalent impact of changes in soil and biomass carbon sequestration due to coal mining is trivial: about 2% of the upstream emissions from the coal fuel cycle, and about 0.03% of total CO_2-equivalent fuel cycle emissions from electricity generation from coal. This is because coal mining disturbs relatively few acres per unit of energy produced.

Carbon content of on-site biomass as a function of nitrogen fertilization

In principle, nitrogen fertilization of energy crops affects the standing yield and hence the carbon content of biomass per hectare. However, rather than formally model this relationship; we merely try to ensure that our assumptions about yields are consistent with our assumptions about fertilization.

CO_2-equivalent GHG emissions from the burning of agricultural residues

Crop residues can be left on the field, used as product, or burned. The burning of residues produces most of the GHGs considered here. The IPCC (1997) and the EPA’s AP-42 (EPA, 1995) provide data on emissions of GHGs from the burning of agricultural residues.

We estimate grams of CO_2-equivalent GHG emissions from the combustion of corn and soybean residue per bushel of corn or soybeans produced:

$$GHGCB_E = WB_E \cdot MCF \cdot (RR_E - 1) \cdot CRFB_E \cdot GHGCB^*_E/2000 \qquad \text{eq. 85a}$$

$$GHGCB^*_E = \sum_G CEF_G \cdot ECB_{G,E}$$

where:

$GHGCB_E$ = CO_2-equivalent GHG emissions from the combustion of residue from energy crop E (g-CO_2-equivalent / bu-product)

WB_E, MCF, and RR are defined in eq. 85

weight per bushel of crop E (see eq. 72).

$CRFB_E$ = of the total crop residue, the fraction that is burned (mass basis) (Data on the fraction of crop residue that is burned rather than left on the field or used as product are not readily available. The IPCC [1997] recommends a value of 0.10 or less for developed countries if country- and crop-specific data are not available. I assume the following:

corn	grass	wood	soybeans
0.05	0.03	0.03	0.05

$GHGCB^*_E$ = CO_2-equivalent GHG emissions from the combustion of residue from energy crop E (g-CO_2-equivalent / ton-dry-residue burned)

2000 = lbs / ton

CEF_G = the CO_2-equivalency factor for gas G (Appendix D; see discussion below too)

$ECB_{G,E}$ = emissions of GHG G from the combustion of residue from energy crop E (grams-gas / ton-dry-residue-E; PM, CO, CH_4, and NMOC emission factors are EPA [1995] AP-42 factors for agricultural residue burning for corn, grasses, unspecified wood, and unspecified crops [used here for soybeans]; SO_2 emissions calculated assuming all sulfur in wood oxidizes to SO_2 [sulfur contents given in section "sulfur content, carbon content, and heating value of biomass;" NO_2 and N_2O emissions assumed to be 0.121 and 0.007 of biomass residue N, per IPCC [1997])

subscript G = GHGs (NMOCs, CH_4, CO, NO_X, N_2O, SO_2, PM)

A similar calculation is done for wood and grass crops, except that the parameters WB, MCF, and 2000 are not needed because in the case of wood and grass the GHGCB factor is in the units of g / dry-ton-product.

Note that the CEFs for CH_4 and CO are adjusted here to account for the fact that in the case of biomass burning, the carbon in the emitted CH_4 or CO molecule comes originally from atmospheric CO_2. The CO_2-equivalent impact of this is the *difference* between the CO_2-equivalent impact of the added CH_4 or CO and the impact of the removed CO_2. This difference is expressed as the CEF for CH_4 or CO (Appendix D) less the ratio of the molecular weight of CO_2 to the molecular weight of CH_4 or CO (2.75 or 1.57).

According to the IPCC (1997), about 10% of the carbon in the residue remains on the ground due to charcoal formation and other aspects of incomplete combustion. The sequestration of atmospheric C-CO_2 on the ground is negative emission of CO_2 for as long as the C remains on the ground. The CO_2 equivalent effect is calculated as a negative emissions of CO_2 today (when the sequestration occurs) less the present value of the emission of the sequestered C when it oxidizes in the future:

$$CO2CB_E = -WB_E \cdot MCF \cdot (RR_E - 1) \cdot CRFB_E \cdot CRFC_E \cdot CF_E \cdot \left(1 - \frac{1}{(1+r)^{LC_E}}\right) \cdot CO2/C \cdot 453.6$$

eq. 85b

where:

$CO2CB_E$ = CO_2-equivalent emissions due to sequestration of charcoal from biomass combustion (g-CO_2/bu-crop)

WB, MCF, RR, and CF are defined for eq. 85, and CRFB is defined for eq. 85

$CFCF_E$ = the fraction of carbon in the burned biomass that is not combusted, but instead is sequestered in the ground as charcoal; my assumptions are based on the IPCC (1997):

corn	grass	wood	soybeans
0.10	0.10	0.10	0.10

r = the discount rate (assumed to decline with time; see the discussion in Appendix D)

CO2/C = the ratio of the molecular weight of CO_2 to C

LC_E = the life of the charcoal formed from combustion of the biomass E; I assume the following (years):

corn	grass	wood	soybeans
20	20	20	20

Because I assume so little residue is burned, the CO_2-equivalent impact of burning turns out to be minor. Note that the effect is a slight negative warming (i.e., a cooling) because the cooling effect of SO_2, biomass aerosols (which unlike fossil-fuel aerosols have a high OM:BC ratio and hence a negative radiative forcing – see

Appendix D), and charcoal carbon sequestration exceeds the warming effect of N_2O and CH_4.

Summary of the contribution to fuel cycle CO_2-equivalent GHG emissions of the various types of land-use, fertilizer, and cultivation-related emissions

The foregoing parameter values for Eq. 69*et seq*. result in the following g-CO_2-equivalent emissions per bushel (corn, soybeans) or dry ton (wood, grass) in the year 2010:

	Corn	Grass	Wood	Soybeans
N_2O related to fertilizer input (synthetic plus manure)	4,783	58,115	4,511	340
N_2O related to biological N fixation, use of crop residue	1,292	1,226	645	20,273
credit for synthetic N displaced by excess biologically fixed N	0	0	0	(5,896)
N_2O from cultivation, independent of fertilizer use	67	2,084	1,930	223
NO_x emissions related to the use of synthetic fertilizer or animal manure	254	4,221	415	991
CH_4 and CO_2 soil emissions related to synthetic fertilizer and animal manure, and CH4 emissions independent of fertilizer use	(577)	(6,762)	(274)	(3,467)
CO_2 sequestered due to fertilization of off-site ecosystems by nitrogen fertilizer leached from field of application	(1,173)	(9,253)	(581)	(85)
CO_2 sequestration in on-site soil, due to cultivation	11,838	212,722	197,946	37,650
CO_2 sequestration in on-site biomass, due to cultivation	524	23,961	(135,207)	4,382
CO_2 equivalent GHG emissions from residue burning	241	131	615	597

Note that the dominant effect, by far, is changes in carbon content of soil due to cultivation. Next most important are N_2O emissions related to the use of fertilizer, manure, crop residue, or biological N fixation, and changes in the carbon content of biomass due to cultivation. In most cases, the effect of CO_2 sequestration from nitrogen fertilization of non-agricultural ecosystems , the effects of N_2O independent of fertilizer use, the effects of burning agricultural residue (assuming that only very small amounts of residue are burned), and all effects of CH_4 and NO_x, are relatively minor.

There are two reasons why changes in the carbon content of soil are the largest effect: 1) in general, soils store a great deal of carbon, and 2) cultivated lands generally have much less carbon than do undisturbed native lands. I have assumed that ultimately the alternative to any energy-crop system is the undisturbed, native vegetation. Other assumptions are possible, and could result in more or less of an impact on soil carbon than I have estimated here. For example, it is possible to assume that the alternative to an energy crop system are the maximum carbon-storing land uses. In the case, CO_2 emissions attributable to cultivation would be higher than estimated here.

Environmental impacts of corn farming

Pitstick (1992) reviews a study by the Economic Research Service (ERS) that estimates the shifts in agricultural production and changes in soil erosion as a result of increased production of ethanol from corn. The ERS finds that increased production of ethanol from corn will cause a decrease in the number of acres planted in soybeans, because the ethanol co-products (corn gluten feed and meal, distillers dried grains, and corn oil) will displace soybean products in the animal-feed and vegetable-oil markets. This finding suggests that the appropriate way to handle the ethanol co-products is to deduct from total ethanol-production emissions the emissions foregone from the production of soybean products (see discussion below). This is co-product method 1 in Appendix K of DeLuchi (1993).

The ERS also estimated that the net amount of soil erosion will increase in proportion to the net increase in planted acreage (acres planted in corn less acres that would have been planted in soybeans). This suggests that is appropriate to assume that removing corn stover from the field for use as a process fuel will increase erosion and deplete soil nutrients (Appendix K of DeLuchi, 1993).

Other environmental considerations

Harvesting practices can affect the nutrient content of the soil, which in turn can affect the use of fertilizer. For example, if corn stover is removed from the field and is used as an energy source in the corn-to-ethanol process, then fewer nutrients will be returned to the soil. Additional fertilizer will be required to balance this loss. The use of additional fertilizer will cause additional emissions of greenhouse gases from fertilizer manufacture, and additional emissions of NO and N_2O emissions from the field. DeLuchi (1991) calculates the affect on fertilizer-related greenhouse-gas emissions of using corn-stover as an energy source in the corn-to-ethanol process rather than leaving it in the field. There may be similar effects to harvesting whole trees in SRIC systems. Hendrickson et al. (1984) note that whole-tree harvesting “has consistently been found to reduce forest floor moisture content” (p. 118), and in their own study found that it “caused significant reductions in forest floor nutrients and mineralization rates” (p. 118). On the other hand, Freedman et al. (1984) did not find significant short-term nutrient depletion after whole-tree harvesting in forest stands in Nova Scotia, but noted that the effects of successive clear cuts in SRIC systems was “unclear.” Chatarpaul et al. (1984) conclude that the effects of whole tree harvesting will vary from site to site, but that “sufficient evidence is currently available regarding the detrimental effects of excess residue removal to urge a cautious, experimental approach in applying whole tree harvesting” (p. 124).

PRODUCTION OF OIL, GAS, AND COAL

Representation of international trade in crude oil, petroleum products, coal, and natural gas

The crude oil used to make petroleum products, such as gasoline, supplied to the U. S., comes from a variety of countries. In 1997, 44% of the crude oil input to U. S. refineries came from the U. S., 11% came from the OPEC countries of the Persian Gulf, 15% came from other OPEC countries (mainly Venezuela and Nigeria), 8% came from Canada, 9% came from Mexico, 3% came from Angola, and the rest came from other exporters (EIA, *PSA 1997*, 1998). On top of this, U. S. imports of finished petroleum products, made from crude oil from countries around the globe, were 7% of the total U. S. supply of finished petroleum products. The EIA projects that the share of petroleum imports will gradually increase over the next 20 years..

There also is significant international trade in coal and natural gas. The U. S. imports over 10% of the natural gas it consumes, mainly from Canada (EIA, *International Energy Annual 1996*, 1998). Some countries in Europe, such as Italy, Germany, and France, import well over half of their total consumption of natural gas, mainly from the countries of the former Soviet Union, and North Africa (EIA, *International Energy Annual 1996*, 1998). Countries in Europe and the Far East (especially Japan and Korea) import a significant fraction of their coal.

Emissions related to the production, transportation, and refining of crude oil, and the production of coal and natural gas, also vary from country to country. For example, oil producers in Africa vent and flare much more associated gas per ton of oil produced than do producers in the U. S. and Canada (EIA, *International Energy Annual 1996*, 1998). On the other hand, Canadian oil producers probably expend more energy to recover a ton of crude oil than do most other producers, on account of the high viscosity of much of the oil recovered. Emissions of methane from coal mining, and leaks of natural gas from transmission and distribution systems, also vary from country to country.

Because the energy used in the U. S. comes from many different countries, with different energy-use and emission factors, the model used in this analysis estimates energy-use and emission factors specific to major energy producing and oil refining countries, and then weights these factors according to the producing country's contribution to the particular energy supply in the U. S. (or in any one of the consuming countries that can be selected for analysis). The energy-use and emission factors are discussed in sections devoted to the type of emission or energy use (e.g., venting and flaring of associated gas, emissions of methane from coal mining). The estimation of the country-by-country contribution to the petroleum, coal, or natural-gas supply of the U. S. (or of any one of the consuming countries that can be targeted for analysis) is discussed next.

<u>Supply of petroleum.</u> The estimation of the ultimate source of crude oil embodied in petroleum products used in the U. S. proceeds in two steps: first, one

estimates the source of finished petroleum products supplied in the U. S.; then, one estimates the source of the crude oil used by each supplier of petroleum products. (A country that refines petroleum products for export to the U. S. might use its own crude oil, or crude oil from another country, or some mix of the two.) The EIA's *AEO* projects the total supply of petroleum products in the U. S., and imports of petroleum products from Canada, northern Europe, southern Europe, Venezuela, North Africa, Nigeria, Indonesia, the Persian Gulf, the Caribbean Basin, Asian exporters, and "other" areas. Given the EIA's projection, and assuming that the difference between products supplied in country C and products imported by country C is products made in country C, the source of the crude oil in petroleum products is:

products produced in:	*are assumed to be from crude oil from:*
U. S.	U. S. and countries exporting to U. S.
Canada	Canada and countries exporting to Canada
Northern Europe	Northern Europe (United Kingdom, Norway)
Southern Europe	North Africa (Algeria, Libya) and other countries exporting to Europe
Venezuela	Venezuela
North Africa (Algeria, Libya)	North Africa (Algeria, Libya)
Nigeria	Nigeria
Indonesia	Indonesia
Persian Gulf (Saudi Arabia, Kuwait, Iran, Iraq, UAE, Qatar)	Persian Gulf (Saudi Arabia, Kuwait, Iran, Iraq, UAE, Qatar)
Caribbean Basin (including Mexico, Colombia, Virgin Islands)	Venezuela
Asian Exporters (including Korea, Singapore)	Indonesia
Other (all areas)	Other Latin America (Colombia, Ecuador, Argentina)

Most of the petroleum products supplied in the U. S. are produced in the U. S. I assume that in the U. S., petroleum products are made from the "average" mix of domestic and imported crude oil. The EIA's *AEO* projects imports of crude oil from the U. S., Canada, Mexico, North Sea, Venezuela, North Africa, Nigeria, Indonesia, other Middle East, Ooher Latin America, other Africa, and other Asia.

With these data and assumptions, the ton weighted-average energy-use or emissions attributable to the use of crude oil for petroleum products in country C is estimated as:

$$WEF_{C,T} = \sum_{PPP} WEF_{PPP,T} \cdot CPP_{PPP,C,T}$$

$$WEF_{PPP,T} = \sum_{PCO} EF_{PCO,T} \cdot CCO_{PCO,C,T}$$

$$CPP_{PPP,C,T} = \frac{PP_{PPP,C,T}}{\sum_{PPP} PP_{PPP,C,T}}$$

$$CCO_{PCO,C,T} = \frac{CO_{PCO,C,T}}{\sum_{PCO} CO_{PCO,C,T}}$$

$$PP_{PPP,US,T} = PP(V)_{PPP,US,T} \cdot DPP_{PPP,US,97}$$

$$CO_{PCO,US,T} = CO(V)_{PCO,US,T} \cdot DCO_{PCO,T} \qquad \text{eq. 86a-f}$$

where:

subscript C = petroleum-consuming country selected for analysis (U. S. [US], in the base case).
subscript T = the target year of the analysis.
subscript PPP = countries that produce petroleum products (see above).
subscript PCO = countries that produce crude oil (see above).
$WEF_{C,T}$ = the weighted-average energy-use or emission factor attributable to use of petroleum in country C in year T (SCF of associated gas vented or flared, or BTUs of process energy, per ton of crude oil used directly or indirectly by country C).
$WEF_{PPP,T}$ = the weighted-average energy-use or emission factor attributable to petroleum-product-producing country PPP in year T.
$CPP_{PPP,C,T}$ = the contribution of petroleum-product-producing-country PPP to petroleum products supplied in country C in year T.
$EF_{PCO,T}$ = the emission or energy-use factor for crude-oil production in country PCO in year T (SCF/ton, or BTUs/ton, or miles of transport; discussed in separate sections below).
$CCO_{PCO,C,T}$ = the contribution of crude-oil-producing country PCO to crude oil supplied in country C in year T.
$PP_{PPP,C,T}$ = petroleum products supplied from producing country PPP to consuming country C in year T (tons; see Appendix B for countries C other than the U. S.) (note that the set of PPP includes C).
$CO_{PCO,C,T}$ = crude oil supplied from producing country PCO to consuming country C in year T (tons; see Appendix B for countries C other than the U. S.) (note that the set of PCO includes C).

$PP(V)_{PPP,US,T}$ = petroleum products supplied from producing country PPP to the U. S. in year T, volumetric basis (barrels; from the EIA's *AEO*).

$DPP_{PPP,US,97}$ = the average density of petroleum products supplied from producing country PPP to the U. S. in 1997 (tons / bbl; calculated from EIA's *PSA 1997* [1998]; I assume that the weighted-average density calculated in 1997 applies to all years of the analysis[58]).

$CO(V)_{PCO,US,T}$ = crude oil supplied from producing country PCO to the U. S. in year T, volumetric basis (barrels; from the EIA's *AEO*).

$DCO_{PCO,\ T}$ = the average density of crude oil produced in country PCO in year T (tons / bbl; see elsewhere in this report for estimate for crude produced in U. S.; for crude produced in other countries, I use the densities reported for 1996 in the EIA's *International Energy Annual 1996* [1998]).

The EIA's *AEO* projections of supply and imports for the U. S. distinguishes "light" from heavy products, and this is also done for the LEM:

light products	*heavy products*
finished motor gasoline	all other EIA petroleum products except still gas
distillate fuel	
jet fuel	
liquefied petroleum gases (does not include kerosene, gasoline blending components, or aviation gasoline)	

The method of eE. 86, or a close variant of it is used to estimate ton-weighted average venting and flaring emissions, energy intensity of oil production, ocean transport distance, and tons of petroleum shipped by international water per ton of petroleum produced. In the calculation of the weighted average refinery energy use, the crude-oil producing countries (as represented in Eq.86b,86d,86f) are not relevant, and refinery energy usage by refining country is substituted for the EF parameter in Eq. 86a.

Supply of coal and natural gas. The calculation of the weighted-average energy-use or emission factor attributable to the use of coal or natural gas in country C in year T is analogous to the calculation for petroleum, except that there is only one step, not two, because there is no distinction between producing countries and refining countries. For natural gas in the U. S., I use the EIA's *AEO* projections of the total

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[58] The overall average density of course will vary from year to year as the mix of individual products imported varies. However, this is a minor effect.

supply of gas to the U. S., and imports from Canada, Mexico, and Algeria (which the EIA reports as LNG). The U. S. is a net exporter of coal. The producing regions and countries for coal and natural gas are tabulated at the start of this report.

Conventions in pertaining to the disaggregation of consumption into "domestic" supply and "imported" supply. As described above, the model relates final consumption of oil, natural gas, and coal to domestic or foreign sources of production. It does this because emissions related to production, refining, and transport depend on where the oil, gas, or coal is produced.

I am unable to project the precise source of the oil, gas, or coal used in the transportation, electricity, and heating end uses represented in the LEM. Therefore, I assume that the end-use consumption is the same as total "average" consumption. Technically, this means that the likelihood that the modeled consumption of F (crude oil, petroleum products, natural gas, coal) comes ultimately from supply source i (domestic, imported from country 1, imported from country 2, etc.) is assumed to be equal to total national imports of F from source i divided by total national consumption of F. This ratio is calculated for each source i (including domestic production) that contributes to total domestic consumption. In the case of petroleum products, this ratio is the parameter CPP in equation 30c. *Note that this method assumes that all imports go to final total consumption; i.e., that no imports end up as exports, refinery feedstocks, or stock changes.* In the case of crude oil used in the U. S., the "final total consumption" in the denominator of the relevant ratio (the ratio represented by the parameter CCO, Eq. 86d) is interpreted specifically to be crude oil supply (or inputs) to refineries.

However, whereas for the U. S. the relevant quantity in the analysis of crude oil imports is crude oil *only,* in the case of all countries other than the U. S. the relevant quantity is crude oil+NGLs+refinery feedstocks (which include unfinished oils and "backflows" to refineries), which constitute all inputs to refineries. This is because the data source on imports in other countries (IEA, *Oil, Gas, & Electricity,* 2002) reports imports of crude+NGLs+refinery feedstocks, whereas the EIA's AEO reports only imports of crude oil in the U. S. In the case of crude oil+NGLs+refinery feedstocks, the denominator is not just refinery intake (as it is in the case of crude oil only, for the U. S.), but instead refinery intake + "direct use" as reported by the IEA (*Oil, Gas, & Electricity,* 2002). This is because imports of crude oil+NGLS+refinery feedstocks reasonably may be assumed to be just as likely to be used "directly" (say, for power generation) as is indigenous production.

Venting and flaring of associated gas

The calculation of venting and flaring of associated gas has been improved in several ways. First, the base-year data on venting and flaring and oil production, by country, have been updated from 1987 (Table M.7 of DeLuchi [1993]) to 1995 (EIA, *International Energy Annual 1993*, 1995). Table 23 shows the new data.

Second, because the EIA now reports crude-oil production, rather than oil production, by country, there is no need to estimate the former from the latter -- the actual crude-oil production data can be input to the model directly[59].

Third, the fraction of gas that is flared rather than vented now can be specified separately for each crude-oil production region. (Formerly, the same fraction applied everywhere.) On the basis of a re-examination of the data in Appendix M of DeLuchi (1993), and consideration of new data from other sources (e.g., Barns and Edmonds, 1990), I have assumed that 13-20% of all gas was vented rather than flared in the new base year of 1995 (Table 23; see Appendix E to this report), which is higher than The original vented fraction of 6% [p. M-25 of DeLuchi, 1993].

Fourth, the user now can specify the annual rate of change of venting and flaring (in SCF/bbl) and the fraction that is flared rather than vented, for every region. With this annual rate of change, and the base-year (1995) data mentioned above, the model calculates venting and flaring emissions (in SCF/bbl) for any year in the projection period. Table 23 shows the assumptions.

Fifth, as mentioned above, the model now explicitly assigns a source of crude oil to petroleum products. Previously, petroleum products from country X were assumed to be made from crude oil from country X, except products from the Caribbean were assumed to be made from Central American crude oil (footnote i, Table M.7). Now, the model allows the user to specify the source of the crude oil used to make petroleum products in each country. The model then calculates venting and flaring emissions on the basis of emissions in the country that is the source of the crude oil, (rather than in the country that actually refines the crude into products). The present assignments of sources of crude oil are based on international flows of crude oil, as reported by the EIA (*International Energy Annual 1992*, 1994).

Sixth, venting and flaring from Federal offshore oil wells has been added. (The EIA data on venting and flaring in the U.S. come from state agencies, which we do not report activity at Federal offshore oil wells.) See Appendix E to this report for further discussion.

Seventh, the model now has emission factors specifically for flared gas, and calculates CO_2-equivalent emissions in the same way that CO_2-equivalent emissions are calculated for other combustion sources: as the sum of CO_2 and CO_2-equivalent

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[59] "Oil production," as formerly reported, equaled crude oil + NGLs + other oils + refinery gain. Because vented and flared gas is associated with crude oil, I had to deduct NGLs, other oils, and refinery gain from the reported "oil production" in order to get to the crude oil production needed to estimated SCF/bbl. Now that the EIA reports crude production, that exercise no longer is necessary.

emissions of non-CO_2 greenhouse gases, where the CO_2-equivalent is equal to the mass of the non-CO_2 gas multiplied by its CEF, and CO_2 per se is calculated by carbon balance. (Formerly, it was assumed that flared gas was burned completely to CO_2, with no emissions of NO_x, PM, N_2O, or SO_2.) Generally, g/10^6-BTU emissions from flaring raw gas are higher than emissions from burning processed gas in boilers, because of the lower temperature, poorer fuel quality, and lack of controls in the case of flaring.

As a result of these changes to the structure and input data, estimated venting and flaring emissions have increased modestly, and total petroleum fuel cycle emissions have increased by 0.5% - 1.0%.

The use of vented or flared associated gas as a feedstock for F-T diesel or methanol

Vented or flared gas does not perform any useful work. If the energy in the gas could be put to useful work, the emissions resulting from the combustion of the gas would not count as a net “new” emission to the atmosphere because the gas would have been vented or flared anyway. From an environmental standpoint, if the gas is going to be burned, it is advantageous to put the energy to work.

Presently, associated gas is vented or flared when there is not enough demand for it economical to build natural-gas production and distribution infrastructure, and it is not, in fact, worthwhile even to re-inject the gas. Thus, in order to be able to put the gas to work economically, we must find less costly ways to bring the gas to market, or else convert the gas on site to other products that can be priced competitively in the world market. Some analysts believe that vented or flared gas can be converted to F-T diesel, methanol, or other liquid fuels and sold at close to the present price of world oil of around $20/bbl. A recent DOE-sponsored study says:

> DOE thinks that ultimately this..technology would produce high-grade liquids that could compete with refined products form crude oil at $20/bbl or less..These technological advances are making Fischer-Tropsch technology more attractive for the development of gas reserves currently deemed not viable, for sites where significant volumes of gas are flared, and for sites far from potential markets (Energy and Environmental Analysis, 1999, p. 3-30).

Given this, it is worthwhile to analyze the fuel cycle CO_2-equivalent emissions of natural-gas liquids derived from gas that otherwise would have been vented or flared. In the model, this is accomplished relatively easily by crediting against full fuel cycle emissions from the natural-gas-to liquids processes (F-T diesel, methanol) the emissions from the venting and flaring that would have occurred anyway, *and* by zeroing out emissions associated with the actual lifting of the gas, because associated gas by definition is already recovered along with the oil. The emissions that would have occurred anyway are estimated as follows:

$$CEGHG_{liquid,T} = \left(CEGHG_{FLgas} \cdot FLF_T + CEGHG_{Vgas} \cdot \left(1 - FLF_T\right)\right) \cdot \left[\frac{BTUI_{gas}}{BTUO_{liquid}}\right]_T \cdot \left(1 + LF_{liquid}\right)$$

$$CEGHG_{FLgas} = EMF_{CO2,FLgas} + \sum_g EMF_{g,FLgas} \cdot CEF_g$$

$$CEGHG_{Vgas} = HHV_{Vgas} \sum_g MF_{g,Vgas} \cdot CEF_g$$

$$FLF = \sum_C FLF_{C,T} \cdot FEED_C$$

eq. 87

where:

$CEGHG_{liquid,T}$ = CO_2 equivalent emissions from the venting and flaring of the amount of associated gas used to make the liquid fuel in year T (grams of CO_2-equivalent per 10^6-BTU of liquid fuel to consumers).

$CEGHG_{FLgas}$ = CO_2 equivalent emissions from the flaring of 10^6 BTU of associated gas (g / 10^6-BTU-gas).

FLF_T = the weighted average fraction of the gas that would have been flared (rather than vented) in year T

$CEGHG_{Vgas}$ = CO_2 equivalent emissions from the venting of 10^6 BTU of associated gas (g / 10^6-BTU-gas).

$\left[\frac{BTUI_{gas}}{BTUO_{liquid}}\right]_T$ = BTUs of gas input per BTU of liquid fuel output from the production plant, in year T (Table 17).

LF_{liquid} = the fraction of fuel lost to evaporation or spillage (assumed to be zero).

$EMF_{CO2,FLgas}$ = emission factor for CO_2 from flared gas (g-CO_2 / 10^6-BTU-gas) (estimated by carbon balance).

$EMF_{g,FLgas}$ = emission factor for gas g from flared gas (I assume that gas everywhere has the same emission factor).

$MF_{g,Vgas}$ = the mass fraction of gas g in the vented gas (Raw gas has been assumed to have the same composition everywhere).

HHV_{Vgas} = the higher heating value of vented gas (associated raw gas) (g / 10^6-BTU).

CEF_g = the CO_2-equivalency factor for gas g.

Because some vented or flared gas is vented, and vented gas is mainly methane, whereas essentially all of the constituents of the feedstock gas in the NGTL lifecycle ultimately are burned, it is possible that the “credit” for not venting and flaring the associated gas exceeds the entire CO_2-equivalent emission of the NGTL lifecycle. This occurs when the benefit of eliminating the venting of methane exceeds the emissions from all sources in the NGTL lifecycle other than the combustion of the feedstock. To a first approximation, this occurs when on the order of 10% of the associated gas is vented rather than flared.

Evaporative emissions of NMOCs and CH_4 from the crude oil cycle

The CO_2-equivalent of evaporative emissions of NMOCs and CH_4 from the production, transport, and storage of crude oil has been added. These emissions, in g-CO_2 equivalent / 10^6-BTU product, are estimated as:

$$CEGHG_T = \frac{MF_P \cdot gBTU_P}{DCO_T} \cdot \left(gGALCO_T \cdot \left(CEF_{NMOC} \cdot CFCO + CEF_{GNMOC-O3/CH4} \right) + CH4CO \cdot CEF_{CH4} \right)$$

eq. 88

where:

$CEGHG_T$ = grams of CO_2-equivalent emissions from evaporative loss of NMOCs and CH_4 from the crude-oil cycle (production, transport, and storage), per 10^6 BTU of petroleum product delivered (gasoline, diesel fuel, residual fuel, still gas, petroleum coke, LPG), in year T.

gGALCO = NMOCs lost from the crude-oil cycle (grams-NMOCs / gallon-crude oil; projected using Eq. 6, with parameter values below).

DCO_T = density of crude oil in year T (grams-crude / gallon-crude; this is a projected value, .

MF_P = the mass fraction of crude oil in petroleum product P (g-crude / g-product; this is 1.0 for every product except oxygenated gasoline, for which the value is about 0.88).

$gBTU_P$ = the HHV of product P (g / 10^6-BTU).

CEF_{NMOC} = the CO_2-equivalency factor for carbon in NMOCs (Appendix D).

$CEF_{GNMOC-O3/CH4}$ = the CO_2-equivalency factor for changes in O_3 and CH_4 due to emissions of NMOC from the combustion of gasoline.

CEF_{CH4} = the CO_2-equivalency factor for CH_4 (Appendix D).

CFCO = the carbon weight fraction of NMOCs lost from the crude-oil cycle (assumed to be 0.858).

CH4CO = CH_4 emissions from the production and transport (11 g/gal; based on EPA, 2000a).

The parameter values in Eq. 6 for the projection of $gGAL_C$ are:

V_L = the minimum value of g/gal-crude evaporative emissions from the crude-oil cycle, as an asymptote (0.10 g/gal; assumed on the basis of the analysis presented in DeLuchi et al., 1992).

V_U = the maximum value of g/gal-crude evaporative emissions from the crude-oil cycle, as an asymptote (3.0 g/gal; assumed on the basis of the analysis presented in DeLuchi et al., 1992, and emissions data in EPA [*National Air Pollutant Emission Trends,1900-1996,* 1997]).

V_{TB} = the g/gallon-crude emissions from the crude-oil cycle in the base year (1.0 g/gal; estimated as national NMOC emissions from oil and gas production in 1996 [EPA, *National Air Pollutant Emission Trends, 1900-1996,* 1997] divided by national refinery input of crude oil in 1996 [EIA, *PSA 1996*, 1997]).

k = shape exponent (the larger the absolute value of k, the more rapidly the limit is approached) (assumed to be -0.10).

T_B = the base year (1996).

These do not include evaporative emissions of gasoline from gasoline marketing, or venting and flaring emissions of associated gas, which are already in the model.

The addition of these emissions increases CO_2-equivalent GHG emissions by only about 0.1 g/mi in the gasoline fuel cycle. (The emissions are added to the “feedstock recovery” stage.)

Emissions of CO_2 removed from raw gas

The model calculates emissions of CO_2 removed from “raw” wet gas in the field and at natural-gas processing plants, per cubic foot of dry gas marketed, as follows:

$$CO2_{CF/\,CF-NG} = \frac{F_{NHC/\,GW} \cdot F_{CO2/\,NHC} \cdot F_{CO2-vented}}{1-\left(F_{NHC/\,GW} - F_{ET/\,GW} - F_{PR/\,GW} - F_{BU/\,GW} - F_{PE+/\,GW}\right)}$$

eq. 89

where:

$CO2_{CF/CF-NG}$ = cubic feet of CO_2 emitted per cubic foot of dry NG marketed.

$F_{NHC/GW}$ = cubic feet of non-hydrocarbon gases (CO_2, H_2S, He, and N_2) removed per cubic foot of gross gas withdrawal.
$F_{CO2/NHC}$ = cubic feet of CO_2 per cubic foot of non-hydrocarbon gases removed.
$F_{CO2\text{-vented}}$ = cubic feet of CO_2 vented per cubic foot of CO_2 removed.
$F_{ET/GW}$ = cubic feet of ethane removed per cubic foot of gross gas withdrawal.
$F_{PR/GW}$ = cubic feet of propane removed per cubic foot of gross gas withdrawal.
$F_{BU/GW}$ = cubic feet of butane removed per cubic foot of gross gas withdrawal.
$F_{PE+/GW}$ = cubic feet of pentanes and higher alkanes removed per cubic foot of gross gas withdrawal.

The parameter values in DeLuchi (1993; p. G-13) resulted in $CO2_{CF/CF\text{-}NG}$ = 0.022. I now have revised some of the parameter values, as follows.

$\underline{F_{NHC/GW}}$. The original parameter value was calculated by dividing total non-hydrocarbon gases removed by gross withdrawals, for the states that reported both. The difficulty here is that this ratio (non-hydrocarbon gases removed per unit of gross gas withdrawal) probably is different for the states that did not report non-hydrocarbon gases removed, because the ratio estimated for the states that did report probably is skewed by the unusually large amount of non-hydrocarbon gases removed from gas produced in Wyoming (about 15% of its gross withdrawals in 1994 and 1995; EIA, *Natural Gas Monthly April 1996*, 1996). For all reporting states, including Wyoming, the ratio in 1994 was 0.036; for all reporting states except Wyoming, the ratio was 0.026 in 1994 (EIA, *Natural Gas Annual 1994*, 1995). In a similar analysis, the EIA (*Emissions of Greenhouse Gases in the United States 1987-1994*, 1995) assumes a ratio of 0.02, on the basis of the data for Texas. However, I think that it is more accurate to calculate the national-average ratio $F_{NHC/GW}$ with the assumption that the ratio for the non-reporting states is the same as the ratio for all reporting states except Wyoming (0.026). With this assumption, the parameter $F_{NHC/GW}$, for all states (reporting and not reporting, including Wyoming) is equal to 0.032.

$\underline{F_{CO2/NHC}}$. Okken and Kram (1989) report that worldwide, raw gas contains about 2% CO_2. The EIA (*Natural Gas 1998, Issues and Trends,* 1999) estimates that in 1997, non-associated gas in the U. S. contained 2.5% CO_2, and associated gas 0.2%, with a weighted average (85% non-associated gas) of about 2.2%. If pipeline gas contains 0.8% CO_2 (Table 5), then the removed CO_2 must be about 1.4% of the raw gas. (This is not an exact calculation of course, but it is sufficient for our purposes here.) Given that the all non-hydrocarbon gases removed are about 3% of the raw gas (parameter $F_{NHC/GW}$, above), CO_2 removed must be about half of all non-hydrocarbon gases removed.

In its similar analysis of CO_2 emissions from natural gas plants, the EIA (*Emissions of Greenhouse Gases in the United States 1987-1994*, 1995) cites data from Texas that indicate that CO_2 is 90% of non-hydrocarbon gases. However, this figure seems too high as a national average. If total non-hydrocarbon gases are about 6% of raw gas, then by the EIA estimate, raw gas contains over 5% CO_2. In order to end up with only 1% CO_2 in pipeline gas, the CO_2 removed would have to be about 4% of the raw gas -- more than the percentage of total non-hydrocarbon gases removed.

I therefore assume that CO_2 is 50% of the total non-hydrocarbon gases removed, H_2S is 30%, and N_2 is 20%.

$\underline{F_{CO2\text{-vented}}}$. In DeLuchi (1993), I assumed that this parameter equals 0.85, which means that I assumed that only 15% of the removed CO_2 is recovered and not emitted. In its own analysis of non-combustion CO_2 emissions from natural-gas processing, the EIA states that “of the 500 billion cubic feet of carbon dioxide produced along with U. S. natural gas, most is emitted to the atmosphere” (EIA, (*Natural Gas 1998, Issues and Trends*, 1999, p. 68). According to the EIA, only a small amount of CO_2 is recovered and used to re-pressurize wells, mainly in Texas and Wyoming. (An earlier EIA report states that virtually all of the removed CO_2 in Texas is recovered [EIA, *Emissions of Greenhouse Gases in the United States 1987-1994*, 1995].) In light of this, I assume that $F_{CO2\text{-vented}} = 0.85$.

$\underline{F_{ET/GW}, F_{PR/GW}, F_{BU/GW}, F_{PE+/GW}}$. These have been re-estimated with 1994 data (EIA, *Natural Gas Annual 1994*, 1995) instead of the 1989 data in DeLuchi (1993).

Note that this is an interim calculation, which allocates the CO_2 emissions to dry natural gas. In the final calculation of CO_2-equivalent GHG emissions, the CO_2 emissions are allocated to NGLs as well as to dry NG, in proportion to the energy content of the total output of each.

The foregoing assumptions result in the raw gas composition shown in Table 5. The calculated CO_2 content of the raw gas is consistent with the estimates reported above by Okken and Kram (1989) and the EIA (*Natural Gas 1998, Issues and Trends*, 1999), and the calculated H_2S content is consistent with EPA (AP-42, 1995) data that indicate that raw gas contains about 1% H_2S.

The measure SCF-CO_2/SCF-dry gas is converted to g-CO_2/10^6-BTU-dry-gas by multiplying SCF of dry gas by the calculated heating value of the gas (10^6-BTU/SCF), and SCF of CO_2 by g-CO_2/SCF-CO_2. The latter is calculated using a modification of the ideal gas law. Formally:

$$CO2_{g/\ mmBTU-NG} = \frac{CO2_{CF/\ CF-NG} \cdot MW_{CO2} \cdot D_{CO2} \cdot 28.32}{\dfrac{\underline{VHHV\ *_{NG}}}{1000000}} \qquad \text{eq. 90}$$

$$VHHV\ *_{NG} = \frac{VHHV_{NG} \cdot 28.32}{1.0548}$$

where:

$CO2_{g/mmBTU\text{-}NG}$ = grams of CO_2 emitted per 10^6 BTU of dry natural gas produced.

$CO2_{CF/CF\text{-}NG}$ = cubic feet of CO_2 emitted per cubic foot of dry NG marketed (Eq. 89).

MW_{CO2} = the molecular mass of CO_2 (Table 5).

D_{CO2} = the molar concentration of CO_2 (moles / liter; Table 5) .

28.32 = L / ft^3

$VHHV^*_{NG}$ = the volumetric higher heating value of dry natural gas (BTU / SCF).

$VHHV_{NG}$ = the volumetric higher heating value of dry natural gas (kJ / L; Eq. 32, calculated based on parameters in Table 5).

Emissions of SO_2 from incineration of H_2S removed from raw gas

Most raw natural gas contains hydrogen sulfide (H_2S), a corrosive compound that must be removed before the gas can be shipped in pipelines. According to the EPA (AP-42), most of the H_2S waste gas is used as a feedstock in nearby sulfur recovery or sulfuric acid plants. However, some of the H_2S is incinerated, and so burns to H_2O and SO_2.

Most raw gas contains on the order of 1% H_2S by volume (EPA, AP-42). If all of this were incinerated, total SO_2 emissions from natural gas processing would be on the order of 14 million short tons. However, the EPA (*National Air Pollutant Emission Trends, 1900-1996,* 1997) estimates that in the 1990s, natural gas production has resulted in on the order of 100 thousand short tons of SO_2 per year. This implies that over 99% of the sulfur in the raw natural gas is removed and recovered or used.

The model calculates grams of SO_2 emitted per 10^6 BTU of dry gas produced with Eq. 89 and 90, with parameters for SO_2 or H_2S in place of parameters for CO_2:

$F_{H2S/NHC}$ = cubic feet of H_2S per cubic foot of non-hydrocarbon gases removed (0.3; as explained above, this results in 1% H_2S in raw gas).

$F_{SO2\text{-}vented}$ = cubic feet of SO_2 emitted (from incinerators) per cubic foot of potential SO_2 (as H_2S) removed from raw gas (0.0065 in 1996, declining 0.4% / year; based on estimates and projections in EPA's *National Air Pollutant Emission Trends,1900-1996,* 1997).

MW_{SO2} = the molecular mass of SO_2 (64.06 g / mole).

D_{H2S} = the molar concentration of H_2S (moles / liter; Table 5).

These assumptions result in about 5 g-SO_2 / 10^6-BTU, which in turn results in the order of 0.02 g / mi SO_2 emissions. This is small, but not utterly trivial; in fact it is roughly the same as the actual tailpipe emissions from vehicle using low sulfur fuel.

The calculated g-SO_2 / 10^6-BTU gas is apportioned between natural gas and natural gas liquids according to the energy produced of each.

Note that any emissions of SO_2 from fuel combustion are accounted for as fuel-combustion emission factors.

Emissions of SO_2 from production and storage of crude oil

Hydrogen sulfide (H_2S) is released to the atmosphere by the production and storage of crude oil. However, the EPA does not estimate these emissions, and in fact until recently little was known about them. Recently, Tarver and Dasgupta (1997) analyzed the emissions and fate of H_2S from the production and storage of crude oil at fields in west Texas. They measured H_2S concentrations in the air and the soil at oil producing sites and at similar non-producing areas. They found that crude oil storage tanks were the major sources of sulfur gas emissions, and that sulfate levels in the soil downwind of oil storage tanks were about two orders of magnitude higher than sulfate levels in soils upwind, or in soils in areas with similar geology but no oil production. However, ambient SO_2 levels around the storage tanks were not elevated. The authors concluded that most of the H_2S emitted from oil storage tanks is absorbed onto dust particles, oxidized to particulate sulfate, and then deposited into the soil.

Tarver and Dasgupta (1997) estimated that the production of 69,858 bbl of west-Texas crude resulted in the emission of about $3 \cdot 10^6$ g S in the year of their study. According to the EIA (*PSA 1996*, 1997; *PSA 1990*, 1991), oil produced in the inland areas of Texas has a sulfur content of about 0.7%, and an API gravity of 38.4, which corresponds to a density of 3153 g / gal or 132,426 g / bbl. Thus, about 4.6% of the sulfur in the west Texas crude was emitted as sulfur in H_2S.

I assume that this 4.6% figure applies to all crude oil production and storage and that 10% of the emitted H_2S is oxidized to SO_2, and the remaining 90% forms particulate sulfate. (Although Tarver and Dasgupta did not find evidence of oxidation to SO_2, in less dusty areas, it is possible that some of the H_2S will oxidize directly to SO_2, rather than form particulate sulfate). Of the 90% that forms particulate sulfate, I assume that half remains in the air long enough to be worth counting as an ambient pollutant. I assume that this particulate sulfate has a formula mass of 200 g / mole.

Thus, g-CO_2-equivalent emissions per gram of oil produced are calculated as follows:

$$CEH2S_T = FH2S \cdot SF_{oil,T} \cdot \left(FSO2 \cdot \frac{MW_{SO2}}{MW_S} \cdot CEF_{SO2} + (1 - FSO2) \cdot FPM \cdot \frac{PM}{S} \cdot CEF_{PM} \right)$$

eq. 91

where:

$CEH2S_T$ = grams of CO_2-equivalent emissions of sulfur compounds derived from H_2S from sulfur in crude oil, per gram of crude oil produced in year T.
FH2S = grams of sulfur emitted as H_2S per gram of sulfur in crude oil (0.046, as discussed above).
$SF_{oil,T}$ = the sulfur weight fraction of crude oil input to refineries in the U.S. in year T (discussed elsewhere in this report).
FSO2 = the fraction of sulfur, in emitted H_2S, oxidized to SO_2 (assumed to be 0.10, as discussed above).
MW_{SO2} = the molecular mass of SO_2 (64.06 g / mole).
MW_S = the molar mass of S (32.06 g / mole).
CEF_{SO2} = the CO_2-equivalency factor for SO_2 (discussed elsewhere in this report).
FPM = of the sulfur in H_2S not oxidized directly to SO_2, the fraction converted to ambient particulate sulfate (assumed to be 0.50, as discussed above; the remainder is assumed to be deposited rapidly in the soil).
PM = the formula mass of particulate sulfate formed from H_2S from oil tanks (assumed to be 200 g / mole, as explained above).
CEF_{PM} = the CO_2-equivalency factor for PM (discussed elsewhere in this report).

From this, it is a simple step to calculate g-CO_2-equivalent per million BTU of fuel consumed: the measure $CEH2S_Y$ is multiplied by any fuel-loss factor, and by g-fuel / 10^6-BTU fuel.

Emissions from the use of concrete to plug oil and gas wells

Appendix H of DeLuchi (1993) reports a figure of 1.1 lbs of concrete / bbl-oil produced, and implies about 300 lbs / 10^6-SCF NG. I have assumed these values in the model. They have a negligible impact on fuel cycle emissions -- about 0.1 g / mi CO_2-equivalent emissions.

Emissions of methane from coal mining.

In the LEM, methane emissions from coal mines in each of the major producing regions of the world are calculated as function of the type of mine (underground or surface), the amount of methane vented, the amount of methane flared, the amount of methane used as a fuel, and, of the amount used as a fuel, the fraction that displaces other consumption of natural gas, the remainder being assumed to satisfy new demand.

In Appendix M of DeLuchi (1993), the generation of coal bed gas was estimated to be 380 SCF / ton, with 5% recovered and used as a fuel, and another 5% flared rather than vented. Since that estimate was made, several comprehensive studies of methane emissions from coal mining have been completed. On the basis of those studies (e.g., Thakur et al., 1996; EIA, *Emissions of Greenhouse Gases in the United States 1987-1994,* 1995; EIA, *Emissions of Greenhouse Gases in the United States 1996* 1997). I have re-estimated the baseline emission rates in the U. S., and projected changes through the year 2050. I assume that the SCF/ton emission rate from underground mines increases slightly, on account of mines getting deeper, but that the amount of gas recovered and used as a fuel also increases.

The new parameter values, for the U. S. and other major producing and exporting countries, are shown in Table 24. See Appendix E of this report for further discussion of U. S. parameter values.

The new calculated overall leakage rates for the U. S. in the year 2015 are substantially lower than the rate assumed in Table 5 of DeLuchi (1991), and as a result, CO_2-equivalent emissions from coal mining have declined by almost 30%, and from the coal-to-electricity fuel cycle by about 2%.

ENERGY USED IN MINING (FEEDSTOCK RECOVERY)

Overview

The Bureau of the Census' *1992 Census of Mineral Industries* reports data on fuel and electric energy consumed at establishments that recover coal, oil and gas, uranium in the U. S. (These data are not available in hard copy; they are available only as a spreadsheet file, from the Census' web site www.census.gov.) In Tables F.1, F.2, and F.3 (coal), G.1 and G.2 (natural gas and natural-gas liquids), H.1 and H.2 (petroleum), and I.1 and I.2 (uranium) of DeLuchi (1993), the Census data from the 1982 and 1987 Censuses of Mineral Industries was used to estimate the energy used to recover coal, gas, oil, and uranium. The same has been done with the 1992 Census data, following the methods presented in DeLuchi (1993).

The previous model called for two kinds of inputs: the total amount of process energy used to recover a BTU of feedstock, such as coal, and the percentage distribution of that recovery energy among the different kinds of process energy, such as diesel fuel and electricity. This has been changed: the model now calls for two different sets of input data for the U. S.:

i) BTUs of each kind of process energy (diesel fuel, gasoline, electricity, gas, etc.) per ton of feedstock (coal, crude oil, uranium, or natural gas) produced in a base year

ii) the percentage change in the energy intensity by fuel type from a base year to the target year.

This new method has three advantages over the old. First, it makes direct use of base-year Census data. Second, because the amount of process energy required for recovery is related directly to the mass of the feedstock, but not necessarily to the energy content of the feedstock, it is better to project recovery energy per ton or cubic foot of feedstock than per BTU. Third, the new method calls for projections of the amount of each kind of process energy used per ton of primary feedstock produced in the U. S. (e.g., BTUs-electricity / ton-coal), rather than for distribution of the total process energy among the different kinds. This is superior because one can project the BTU / ton amounts on the basis of the EIA's *AEO* projections or other considerations (such as historical data for 1982, 1987, and 1992).

Using the data from the 1982, 1987, and 1992 *Census of Mineral Industries*, I have estimated the actual amounts of BTUs of each kind of process fuel used per ton of coal, crude oil (from conventional onshore recovery), uranium, or raw natural gas in the U. S. On the basis of these estimates, I assume BTU / ton energy requirements for each kind of process fuel and feedstock in the U. S. for a particular base year. Generally, 1992 is the base year, and values from the 1992 *Census of Mineral Industries* as base-year values. Percentage changes per year were determined on the basis of the EIA's *AEO* projections and other considerations.

The energy intensity of the recovery stage, in BTUs of process energy per BTU of feedstock produced (as shown in Table 3 of DeLuchi [1991]), now is calculated by dividing the projections of BTUs of process energy per ton of feedstock by the projected energy content of the feedstock in BTUs per ton. Because the BTU / BTU energy intensity now is the product of BTU-process-energy / ton-feedstock and ton-feedstock / BTU-feedstock, it properly reflects projected changes in the energy content of the feedstock, due perhaps to declining quality. (Recall that in the previous model, BTU / BTU was input directly.)

In most cases, the changes discussed above to the structure and input data of the estimation of GHG emissions from mining have only a minor effect on overall fuel cycle emissions. However, fuel cycle emissions from the oil recovery stage have increased by 20%, although this results in less than a 1% increase in fuel cycle g / mi emissions, because emissions from recovery are a minor fraction of the total. GHG emissions from the natural-gas recovery stage have declined slightly. In the case of methanol made from natural gas, the overall effect is a 1% reduction in total lifecycle GHG g / mi emissions.

Documentation of miscellaneous U. S. parameter values

1). As mentioned above, the data on fuels and electric energy consumed at U. S. mining establishments in 1992 is provided in a spreadsheet available from the Bureau of the Census website. The spreadsheet shows the physical quantity of coal, distillate fuel, residual fuel, natural gas, gasoline, and electricity consumed, and the dollar expenditure on "other" and "undistributed" fuels. ("Other" fuels are coke, LPG, wood, and other minor fuels. Expenditures on "undistributed" fuels are those by establishments that did not report the quantity of fuels consumed, or were not mailed a survey.) Thus, in order to have a complete accounting of energy use by mining establishments, one must estimate the energy content of "other" and "undistributed" fuels, on the basis of the dollar expenditures on these fuels. For "other" fuels, the total expenditures is multiplied by the Census' estimate of the average 10^6-BTU / $ energy value of "other" fuels -- 0.210 in 1992, according to Roehl (1997). I assume that "undistributed" fuels should be distributed to all of the specific fuel categories (except electricity)[60] in proportion to reported expenditures; that is, I assume that the distribution of undistributed fuels is the same as the distribution of reported distributed fuels, where the distribution is with respect to expenditure. (The Census actually makes the same assumption, except at the level of all expenditures in all mining industries [Roehl, 1997], whereas I make this assumption for each industry in the mining sector.)

2). Many fuel data are not disclosed by the Census, so as not to reveal information about individual companies. Some of these data can be back-calculated on the basis of higher-level totals, but most cannot. I have estimated the ones that cannot.

3). I have revised historical data on the production of uranium concentrates in 1987, on the basis of new EIA data (*Uranium Industry Annual 1996*, 1997). Also, I now use total production from mines, rather than total product shipped. I have assumed that in 1982, 1987, and 1992 censuses, uranium mining alone consumed 95% of the fuels and electricity reported for the uranium / radium / vanadium industry as a whole. Finally, uranium's share of energy use in metal-mining service industries is assumed to be equal to the ratio of uranium-mining energy to all metal-mining-energy. (All of these assumptions are relevant to the estimate of BTUs-process-fuel / ton-uranium historically, which estimates serve as the basis of my projection.)

4). The Census reports fuels and electric energy consumed at oil-producing and gas-producing establishments combined; it does not report data for oil-producing establishments or gas producing establishment alone. Hence, the reported total must be apportioned to oil and to gas separately. the apportioning factors for energy use in the oil and gas field-service industry has been changed on the basis of three metrics: the ratio of the value of natural gas production to the value of natural gas + crude oil production; the ratio of the number of gas wells to the number of gas + oil wells; and the ratio of the cost of drilling gas wells to the cost of drilling gas + oil wells (all data from EIA's *AER 1996*, 1997):

	1982	*1987*	*1992*
Value of domestic production	0.37	0.42	0.48
Number of exploratory and development wells	0.33	0.32	0.47

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[60] Expenditures on "undistributed" fuels do not include any expenditures on electricity. The reported cost and quantity of electricity includes the Census' estimates of the cost and quantity of electricity consumed at establishments that did not report data or were not mailed a survey.

Cost of drilling exploratory and development wells	0.52	0.43	0.51

5). I have distinguished three kinds of oil recovery: conventional onshore recovery, conventional offshore recovery, and heavy oil recovery. I distinguish these three because they have quite different energy requirements (offshore recovery is much more energy intensive than is onshore recovery), and because their shares of total oil recovery can vary considerably from country to country.

In the model, the user inputs fuel use, per ton of oil recovered, for conventional onshore oil recovery in the U. S., and then estimates the BTU / ton energy requirement of heavy oil recovery and offshore oil recovery relative to that input for onshore conventional oil recovery. Now, the Census of Mineral Industries reports total inputs for all U. S. oil recovery, offshore as well as onshore, and heavy (enhanced oil recovery) as well as conventional, and as a result the inputs for onshore conventional recovery alone -- which is what the model now calls for -- must be back calculated from data on the onshore / offshore / heavy oil split, and the energy intensity of offshore and heavy-oil recovery relative to that of onshore. In 1992, the year of the most recent Census of Mineral Industries, production from offshore oil wells was 17% of production from all wells (EIA, *AER 1997*, 1998), and production from enhanced oil recovery probably was about 6% of all production (it was about 9% in 1997 [EIA, *AEO 1999*, 1998]). (Kadam et al. [1999] use a data base from the *Oil & Gas Journal* to estimate that in 1994 offshore production was 20%. They further state that enhanced / advanced oil recovery was 11%, of total domestic production.) DeLuchi (1993) cites estimates that offshore oil recovery is several times more energy intensive than is onshore, and McCann and Magee (1999) provide estimates that indicate that the extraction of heavy crude is at least twice as energy intensive as is the extraction of light crude. Assuming then a factor of 3.0 for offshore relative to onshore production, and factor of 2.0 for heavy or enhanced oil production relative to onshore production of light oil, we can back-calculate that BTU / ton energy intensity of conventional onshore recovery is about 70% of the overall average BTU / ton intensity for all oil recovery in 1992. This was used as the basis for estimating the input / ton requirements for onshore conventional oil recovery in the U. S[61].

6). The energy intensity of natural gas recovery is represented as BTUs per ton of marketed production. Marketed production is equal to gross withdrawals from wells (excluding lease condensate) minus: non-hydrocarbon gases removed, gas used for re-pressuring, and gas vented and flared. Put another way, marketed production is equal to dry natural gas plus the natural gas liquids originally contained in the total gas stream. Because marketed production is the output of the field production stage, and

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[61] Oil producers may use a small amount of CO_2, to enhance oil recovery that is produced from fuels "outside" of the oil industry itself. If so -- if ultimately the source of this CO_2 is not accounted in the *Census of Mineral Industry* data I use, then I underestimate inputs to and emissions from oil production.

the input to the natural-gas processing stage, it is appropriately related to the process energy used in field production.

The energy intensity of natural-gas processing is represented as BTUs per ton of wet gas processed. Because all natural gas liquids must first be recovered with the gas stream, and then extracted from the wet gas at a processing plant, the final energy ratio of interest for NGLS, BTUs-process-energy / BTU-NGL-delivered, is equal to BTUs-process-energy / ton-gas-marketed, or BTUs-process-energy / ton-wet-gas processed, multiplied by the NGL heat content in tons-NGL / BTU-NGL. However, because some marketed gas production is dry enough to bypass the processing plants and go directly to consumers, the ratio of interest for NG, BTUs-process-energy / BTU-NG-delivered, is equal to BTUs / ton-processed multiplied by the ratio of the gas output of processing plants to total dry gas production, and then by the heat content of dry gas (tons-NG / BTU-NG).

In order to calculate these ratios, the reported volumetric production data (EIA's *Natural Gas Annual;* Bureau of the Census' *1992 Census of Mineral Industries*) must be converted to tons. The conversion is documented in Table 26, which shows EIA and Census production data for the years for which the Census reports energy used in mining (1982, 1987, and 1992).

7) The g / 10^6-BTU emissions calculated here for NG production, on the basis of Census data on fuel use for NG recovery and EPA emission factors for different fuels, can be compared with emission factors for offshore NG production, calculated from independent data in the EIA's *Natural Gas 1998, Issues and Trends* (1999) (g / 10^6-BTU):

	NMOGs	NO_X	SO_X	CO	TSP
EIA, offshore NG production	3.0	27	1.5	7.9	1.6
calculated here for 1990	0.8 / 6.6*	30	1.3	13	0.8

*6.6 includes VOCs from leaks and flares.

The agreement with the EIA estimates is quite good.

8). As mentioned above, I base my projections of the percentage change per year in energy intensity by fuel partly on the EIA's *AEO* projections of mining energy intensity. The EIA projects BTUs of energy per 1992 dollar of output for mining (table 32 of the supplemental data). According to the documentation of the industrial module of the National Energy Modeling System, the EIA assumes that 1992 dollars per ton of output is constant (EIA, *Model Documentation Report: Industrial Sector Demand Module of the National Energy Modeling System*, 2000). This means that projections based on BTUs / 1992$ are the same as projections based on BTUs / ton (which is what I want) would be. The most recent EIA projections of the percentage change per year are:

	1999-2020	2010-2020
Residual Oil	-0.9%	-0.6%
Distillate Oil	-0.9%	-0.6%
Motor Gasoline	-0.6%	-0.3%
Other Petroleum	-0.3%	0.1%
Natural Gas	-0.3%	0.1%
Lease and Plant Fuel	1.6%	0.8%
Steam Coal	-0.9%	-0.8%
Renewables	-0.2%	0.1%
Purchased Electricity	-0.7%	-0.4%
Total	*0.7%*	*0.4%*

Energy intensity of feedstock recovery in other countries

As discussed elsewhere, the model now accounts for international flows of coal, oil, gas, and uranium It estimates emission and energy-intensity factors specific to major energy producing and oil-refining countries, and then weights these factors according to the producing country's contribution to the particular energy supply in the U. S. (or in any one of the consuming countries that can be selected for analysis).

The energy intensity of feedstock recovery (oil production, coal mining, natural-gas production, uranium mining, and production of natural gas liquids) in energy-producing regions outside of the U. S. is entered *relative* to the estimated overall BTU/ton intensity in the U. S. (This method assumes that the distribution of individual process fuels is the same as in the U. S.) In the case of oil production, BTU/ton energy intensity of oil recovery, in each country, is calculated relative to the BTU/ton energy intensity of conventional onshore oil recovery in the U.S., on the basis of the amount of conventional oil produced from onshore wells, conventional oil produced from offshore wells, and heavy or enhanced oil production, and the assumed relative energy intensity of each type of production:

$$BTONR_{C,T} = \sum_{P} OPF_{P,C,T} \cdot BTONR_{P,C} \qquad \text{eq. 92}$$

where:

subscript P = types of oil production (conventional onshore oil recovery; conventional offshore oil recovery; heavy or enhanced oil recovery).

subscript C = major oil-producing countries or regions (Table 25).

$BTONR_{C,T}$ = the BTU/ton energy intensity of oil recovery in country C in year T, relative to the energy intensity of recovery conventional onshore oil in the U. S.

$OPF_{P,C,T}$ = of total oil production in country C in year T, the fraction that is of type P (Table 25).

$BTONR_{C,T}$ = the BTU/ton energy intensity of oil recovery of type P in country C in year T, relative to the energy intensity of recovery conventional onshore oil in the U. S.; assumed to be as follows, for all years and all countries:

onshore conventional	offshore conventional	heavy or enhanced
1.00	3.00	2.00

In the case of coal mining, gas recovery, and uranium mining, I assumed a relative energy intensity of 1.0 for all countries (i.e., the same energy intensity as in the U. S.), except:

- gas recovery in Northern Europe is assumed to be 50% more energy intensive than recovery in the U. S. – Most of the production from Northern Europe (Norway, the Netherlands, and the U. K.) is from the North Sea (EIA, *North Sea*, 1998)[62], and offshore gas recovery presumably is more energy intensive than is on-shore recovery. (In the U. S., most gas is onshore).
- coal mining in South America, Asia, South Africa, Eastern Europe, and Russia is assumed to be 10-20% more energy intensive than coal mining in the U. S., on account of presumably less energy-efficient recovery methods.

PIPELINE TRANSMISSION AND DISTRIBUTION OF NATURAL GAS AND HYDROGEN

Energy intensity of natural gas transmission

The energy required transporting natural gas by pipeline, and the total amount of gas leaked from compressors, joints, and other parts of the system, are related to the distance of transmission. Because of this, and because the average gas transmission distance varies from country to country and from end user to end user (e.g., the average transmission distance of gas from Russia to Italy is much greater than the distance from the North Sea to England; the average transport distance to a methanol production plant probably will be less than the average distance to a CNG station), the model now estimates the energy intensity of natural gas transmission as a function of transmission distance. Given the transmission distance for each natural-gas end use (electric utilities, industry, etc.; see Table 27) relative to the distance to the transportation sector (relative distances specified by user), the overall transmission distance from producer to consumer for country C relative to that in the U. S., the energy efficiency of compressors

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[62] Virtually all of the Norwegian gas and about 1/3 of the Dutch gas is offshore (EIA, *North Sea,* 1998).

in producer countries relative to that in the U. S, the overall transmission energy intensity for all end uses, and other parameters, the model calculates the energy intensity of transmission to each end-use sector U (EIT_U), in BTUs-pipeline fuel / BTU-gas-consumed. This is done for the consuming country selected for analysis, on the basis of the contribution of gas-producing countries to the gas supply in the selected country. Values for other countries are referenced to average values in the U. S.

Formally, for the energy intensity of transmission *except* from producing wells to natural-gas-to-liquids plants in producing countries:

$$EIT_{U,T,C} = \frac{RD_U}{RD_{Avg}} \cdot EIT_{Avg,C,T}$$

$$RD_{Avg} = \frac{\sum_U CS_{U,C,T} \cdot RD_U}{\sum_U CS_{U,C,T}}$$

$$EIT_{Avg,C,T} = EIT_{Avg,US,T} \cdot \sum_{GP} REITM_{GP} \cdot RTD_{GP,C} \cdot NGC_{GP,C}$$

$$CS_{U,C,T} = \frac{C_{U,US,T}}{C*_{US,T}} \cdot \frac{1-EIT_{Avg,C,T}}{1-EIT_{Avg,US,T}}$$

$$C*_{US,T} = \sum_U C_{U,US,T} - C_{LP,US,T} - 0.5 \cdot I_{US,T} \qquad \text{eq. 93}$$

where:

subscript GP = gas producing countries (see parameter NGC)
subscript U = natural gas end uses (see Table 27)
$EIT_{U,T}$ = the energy intensity of transmission to end-use sector U in year T (BTU-pipeline-fuel / BTU-gas-to-sector)
RD_U = the transmission distance to end-use sector U, relative to the distance to the LNG or CNG transportation sector sector (Table 27; assumed to be the same in all countries)
RD_{Avg} = the consumption-weighted average distance to all end-use sectors, relative to the distance to the commercial sector (Table 27)
$EIT_{Avg,C,T}$ = the overall average energy intensity of pipeline transmission in country C in year T (BTU-pipeline-fuel / BTU-NG-consumed)
$CS_{U,C,T}$ = the consumption share of end-use sector U in country C in year T (Table 27 shows values for the U. S.)
$REITM_{GP}$ = the energy requirements per mile of pipeline transmission (BTU-pipeline-fuel / BTU-NG-consumed / mi-transported) in gas-producing country GP *relative* to that in the U. S. (assumed to be 1.0 for developed

countries, 1.10 for the Former Soviet Union, and 1.05 for all other countries)

$EIT_{Avg,US,T}$ = the overall average energy intensity of pipeline transmission in the U. S. in year T (BTU-pipeline-fuel / BTU-NG-consumed) (0.032 for the U. S.; equal to EIA *AEO*-reported pipeline gas ÷ (total gas consumption - field use - 0.5 · imports)[63]

$RTD_{GP,C}$ = the average transmission distance from wells in gas producer GP to all consumers in country C, relative to the average domestic transport distance to all end uses in the U. S. (assumed to be 1.10 for Canada to the U. S.; see Appendix B for countries other than the U. S.)

$NGC_{GP,C}$ = the contribution of gas-producing country GP to the total gas supply of consuming country C selected for analysis (gas from country GP divided by total gas supply in country C), based on EIA's *AEO* projections for the U. S. as follows (see Appendix B for other countries):

Producer:	U. S.
U.S.	0.87
Canada	0.12
Mexico	0.00
Northern Europe	0.00
Southern Europe	0.00
Algeria	0.01
Indonesia	0.00
Persian Gulf	0.00
Russia, Asia	0.00

$C_{U,US,T}$ = the consumption of natural gas in end-use sector U in the U. S. in year T (EIA's *AEO* projections)

$C^*_{T,US,T}$ = the total amount of gas moved by U. S. pipelines in year T

$C_{LP,US,T}$ = the amount of gas used for lease and plant fuel in the U. S. in year T (EIA's *AEO* projections)

$I_{US,T}$ = net imports to the U. S. in year T (EIA's *AEO* projections)

Note that the U. S. pipeline energy intensity is estimated with respect to the total amount of gas actually moved by U. S. pipeline, which is not the same as the total end-

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[63] I subtract field use, which is called "lease and plant fuel" in the EIA statistics, and half of imports, because no lease and plant fuel is shipped by pipeline, and a portion of the energy required to transmit imported gas is consumed in the exporting country (i.e., Canada).

use consumption, because lease and plant fuel is not shipped by pipeline, and some imports are moved by foreign rather than domestic pipeline compressors.

The case of transmission from producing wells to natural-gas-to-liquids (NGTLs) plants in gas producing countries is handled analagously, but more simply, as the product of the relative transmission distance, the relative per-mile energy intensity (parameter REITM), and the contribution of the gas-producing country to total NGTL supply (parameter NGC), for each gas-producing country. The relative transmission distance is the distance from producing wells to NGTL plants in the producing country, relative to the average distance to all end uses in the U. S. This is assumed to be 20-25% for most producing countries, on the grounds that NGTL plants are likely to be located relatively close to major gas-producing fields.

Leaks of natural gas

Partly on the basis of the results of an EPA / Gas Research Institute study [GRI, 1996] updated by the EPA and EIA (see Appendix E to this report), the calculation of CO_2-equivalent emissions of gas leaks from natural-gas systems has been changed. (See Appendix E to this report for a review of studies of leakage from natural gas systems.)

Although at a general conceptual level the estimation of CO_2-equivalent emissions of gas leaks is straightforward, there are, as always, niggling details to get straight in the calculation. In general, the CO_2-equivalent emission of gas leaks is equal to the CO_2-equivalency factor for natural gas multiplied by the amount of natural gas leaked. The CO_2-equivalency of natural gas is a function of the composition of the gas, and the CO_2-equivalency factors for the components of the gas. The amount of natural gas leaked depends on unit leakage rates for each stage, gas input and output for each stage, the allocation of leaks to multiple products, and other factors. For the U. S., the unit leakage rates are derived, with some adjustments, from the EPA / GRI (1996) detailed study of methane leaks from the natural gas industry. For other major gas-producing countries, leakage rates from recovery and processing are estimated directly; leakage rates from transmission and storage are calculated relative to the rate for the U. S. as a function of the length of the system in the producing country relative to the length in the U. S. and the leakage rate per mile in the producing country relative to the leakage rate per mile in the U. S. These input or calculated leakage rates for producing countries are then weighted by the contribution of each producing country to the gas supply of the country selected for analysis.

Formally:

$$GLGHG_{C,T} = \left(\sum_{G} MF_G \cdot CEF_G\right) \cdot gBTU_{NG} \cdot \left(\sum_{i} GL_{i,C,T} \cdot UA_{i,U} \cdot K_{i,T} \cdot IO_i \cdot MP_i\right)$$

$$GL_{i^*,C,T} = \left(\sum_{GP} GL_{i^*,GP,92} \cdot NGC_{GP,C}\right) \cdot \left(1 + \frac{\Delta GL_{i^*}}{100}\right)^{T-1992}$$

$$GL_{i=d,C,T} = GL_{i=d,GP=C,92} \cdot \left(1 + \frac{\Delta GL_{i=d}}{100}\right)^{T-1992}$$

$$GL_{i,US,92} = \frac{\dfrac{CH4L_{i,US,92}}{CH4VF}}{TP_{i,US,92}}$$

eq. 94 a-f

$$UA_{i=transmission,U} = \frac{RD_U}{RD_{Ave}};\ UA_{i=other,U} = 1$$

$$K_{i,T} = \left(1 + IO_{i+1} \cdot GL_{i+1,T}\right) \cdot K_{i+1,T}$$

where:

subscript G = gas constituents of natural gas (methane, ethane, propane, carbon dioxide, nitrogen, and so on)

subscript i = stages of the natural-gas fuel cycle (production, processing, transmission, distribution, dispensing)

subscript i* = stages of the natural-gas fuel cycle *except* distribution

subscript “i=d” refers to the distribution stage of the fuelcycle

subscript “GP = C” means that the gas-producing country is the target country selected for analysis (in the case of the distribution stage)

subscript C = country selected for analysis

subscript GP = gas producing countries (see eq. 26)

$GLGHG_{C,T}$ = CO_2-equivalent GHG emissions from fuelcycle leaks of natural gas, per energy unit of gas delivered, in country C in target year T (g / 10^6-BTU)

MF_G = the mass fraction of gas G in natural gas (grams of G per gram of natural gas)

CEF_G = the CO_2-equivalency factor for gas G (Appendix D)

$gBTU_{NG}$ = the gram / 10^6-BTU mass heating value of natural gas (calculated from the heating value of the constituent gases)

$GL_{i,C,T}$ = the system-average rate of loss of from gas stage i in target year T, attributable to country C (ratio of gas lost to gas output from stage)

$UA_{i,U}$ = end-use specific adjustment factor; equal to the gas loss rate for end-use U divided by the system-average gas loss rate, for stage i

$K_{i,T}$ = the cumulative loss factor for stage i in year T (see also the discussion of K factor in "own-use" section elsewhere)

IO_i = input/output factor for stage i; the ratio of the output of stage i to the output of stage i-1 (similar to the K factor; Table 28)

MP_i = allocation of emissions from stage i to multiple products of stage i; equal to the HHV of NG output from stage i divided by the HHV of all products output from stage i (Table 28)

$GL_{i,GP,92}$ = the system-average rate of gas loss from stage i in gas producing country GP in 1992 (ratio of gas lost to gas output from stage i; discussed below and in Appendix B)

$NGC_{GP,C}$ = the contribution of gas-producing country GP to the total gas supply of consuming country C selected for analysis (see eq. 26)

$?GL_i$ = the annual percentage change in the rate of gas loss from stage i (Table 28)

T = target year

$CH4L_{i,US,92}$ = the volume of methane vented or leaked from stage i in the U. S. in 1992 (10^9 cubic feet; see updated EPA/GRI [1996] report, Appendix E to this report, and Table 28)

CH4VF = the volume fraction of methane in natural gas (90-96%)

$TP_{i,US,92}$ = gas output from stage i in the U. S. in 1992 (Table 28)

$UA_{i=transmission,U}$ = end-use specific adjustment factor for gas loss from transmission stage (this accounts for different transmission lengths, and hence different emission rates, for different end uses)

RD_U = the relative transmission distance for end-use sector U (Table 27)

RD_{ave} = the average relative transmission distance for all end-use sectors (Table 27)

I distinguish between gas leaks during production, processing, and transmission (stages i*), and gas leaks during distribution (stage i=d), because the former can occur in the gas-producing countries that export to the country of interest C, whereas distribution-stage emissions occur only in the country of interest C. Also, I assume that only the baseline gas loss per stage varies from country to country.

Gas leakage rates in major gas producing countries (parameter $GL_{i,GP,92}$). Gas leakage rates in countries other than the U. S. in the base year of 1992 are estimated as follows:

- *Gas recovery:* For developed countries (except Canada), the leakage rate for gas recovery is assumed to be the same as in the U. S. For developing countries, the rate is assumed to be 50% higher (in relative terms).

- *Gas processing:* For developed countries (except Canada), the leakage rate for gas recovery is assumed to be the same as in the U. S. For developing countries, the rate is assumed to be 50% higher (in relative terms).
- *Gas transmission and storage:* As mentioned above (and in except in the case of Canada), leakage rates from transmission and storage are calculated relative to the rate for the U. S., as a function of the length of the transmission system in the producing country relative to the length in the U. S. and the leakage rate per mile in the producing country relative to the leakage rate per mile in the U. S. Formally,

$$GL_{TM,GP,92} = GL_{TM,US,92} \cdot RTD_{GP,C} \cdot RLTM_{GP}$$

where the subscript TM refers to transmission, the terms GL and RTD are defined in the sections above, and RLTM is the leakage rate per mile in country GP relative to that in the U. S. Regarding the relative leakage rate: I assume 1.0 for Canada, Europe, Australia, and generic developed countries; 1.10 for the Persian Gulf, Asian exporters, the Caribbean basin, and "other;" 1.15 for Mexico, Indonesia, Malaysia, and generic less developed countries; 1.25 for North Afria and Nigeria; and 2.0 for the FSU. In the case of the FSU, the assumption is based on the estimates of Reshetnikov et al. (2000), which show very high leakage rates from piplines in Russia (see Appendix B for more discussion).

- *Gas distribution:* Leakage rates are estimated on the basis of a variety of sources; see Appendix B for details.

In the case of Canada, leakage rates are calculated from a Canadian version of the EPA / GRI study done for the U. S. See Appendix B for details.

Miscellaenous notes of method and data regarding leaks of natural gas. The calculation for the U. S. involves more than just applying the EPA / GRI (1996) summary finding that gas leaks amounted to 1.4% of gross production in 1992. In fact, there are at least seven reasons why this overall 1.4% differs from the correct leakage rate estimated here.

First, the overall 1.4% emission rate depends on the emission rate and throughput weight (input / output factor) for each stage, and hence will change if the throughput weights change. For example, the 1.4% emission rate is the result of a certain amount of gas being processed at NGL plants. In the future, a greater or lesser fraction of marketed production might go to NGL plants, depending on whether the raw produced gas requires more or less processing, with the result that the overall emission rate will change, all else equal. (Note that in the LEM, the processing segment now is treated explicitly, as a separate segment.) This is addressed by estimating emission rates and input / output factors separately for each stage.

Second, EPA / GRI estimate emissions of methane only; they do not estimate emissions of the other minor constituents of natural gas, including carbon dioxide and ethane (an indirect greenhouse gas). This is corrected by scaling the estimated volume emissions of methane by the ratio of total gas volume to methane volume.

Third, the total overall 1.4% emission rate includes methane emissions from combustion, which I have estimated separately. Hence, I count here only venting and fugitive emissions.

Fourth, the EPA / GRI study inappropriately excludes emissions from foreign (primarily Canadian) transmission systems shipping gas to the U.S., and inappropriately includes emissions from U. S. systems exporting gas. Because pipeline imports are an order of magnitude greater than exports (EIA, *Natural Gas Annual 1995*, 1996), the net effect is to understate emissions from transmission systems that deliver gas to U. S. consumers. The magnitude of the underestimation depends on the length and quality of the foreign transmission systems, among other factors.

Fifth, the 1.4% emission rate incorporates the average leakage rate for transmission systems. However, any particular NG fuel cycle being analyzed will involve transmission distances greater or less than the average, and consequently will have more or less than the average leakage from transmission systems, because losses from the transmission system are related to the length of the transmission system. I estimate leakage from the transmission system as a function of distance.

Sixth, as old leaky equipment is replaced by new equipment, and as the industry otherwise seeks specifically to reduce methane emissions, leakage rates will decline (EPA / GRI, 1996). In 1993, the U. S. natural gas industry joined with the U. S. EPA to establish the "Natural Gas Star Program" to reduce methane emissions. Companies participating in the program adopt technologies and management practices to prevent gas leaks and improve system efficiency (EIA, *Natural Gas 1998, Issues and Trends*, 1999, p. 37). In 1993, the industry reduced methane emissions by about 5 BCF; in 1997, by about 15 BCF (about 2% and 5% of the total loss estimated for 1992 [Table 28]). The goal for the year 2000 is about 45 BCF (EIA, *Natural Gas 1998, Issues and Trends,* 1999), or probably on the order of a 15% reduction from "no-STAR-program" emissions in the year 2000. Given this, my assumptions are shown in Table 28.

Seventh, the production and processing stages produce natural gas liquids (NGLs) as well as natural gas. The correct way to handle this is to attribute all of the incremental production and processing emissions to the incremental NG production, but then estimate what emissions are displaced as a result of the marketing of the NGL co-product. However, for simplicity, and because NGL production is small compared to NG production, I assume that NGLs and NG are interchangeable in energy terms. Thus, total emissions are allocated to NG and NGLs according to the total energy content of the production of each[64].

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[64] Darrow (1994) allocates field emissions on the basis of energy content, but assigns 75% of NGL plant emissions to NGLs, and 25% to NG, on the grounds that NGL plants are the "primary production process" for NGLs. I note though, that energy content of the residue gas from NGL plants far exceeds the energy content of the NGLs (by about an order of magnitude).

Note that I assume that the loss rates for the transmission and distribution stages are proportional to the length of the pipelines (EPA / GRI, 1996), with the result that the leakage rate from a NG-to-methanol transmission system is less than the rate from a NG-to-CNG system, because of the shorter assumed transmission distance in the methanol system.

Although the EPA / GRI (1996) study (as updated by EPA and EIA), upon which my analysis is based, clearly is the best ever done for the U. S., it is likely that a good deal of uncertainty in estimates of NG leakage rates remains. For example, Shorter et al. (1997) used a tracer gas, SF_6, to estimate the leakage from gas plants, separator stations, wells, storage fields, compressor stations, metering stations, high-pressures stations, and vaults, and found that for many sources the leakage rate (L / min) varies over 2 or 3 orders of magnitude. (In the EPA / GRI study, tracer gas measurements were used to characterize emissions from meters and pressure regulating stations [Harrison et al., 1996].)

Work and energy use of gas-turbine and gas-engine compressors

Table G.5 of Vol. 2 of DeLuchi (1993) presents data from a survey of transmission companies, and from a review of the literature, which show that the installed horsepower capacity of compressor engines is about 4 times higher than the installed capacity of compressor turbines. Consistent with this, the recent EPA / GRI (1996) detailed analysis of methane emissions from the U. S. gas system estimates that in 1992 engines provided 80% of compressor horsepower-hour work, and turbines 20% (not counting work provided by electric compressors)[65]. Allowing that electric compressors provided 5% of total compressor work (Table G.5 of DeLuchi [1993]), I assume that in 1992, engines provided 76% of the total work, turbines 19%, and electric motors 5%. I then assume that the share of turbines increases slightly.

Because emissions are estimated per BTU input to the compressor, rather than per unit of work provided, these work-output shares must be converted to energy-input shares. To do this conversion, I assume that turbines use 1.33 times as much energy per horsepower-hour as engines, and that electric motors use 0.25 times as much energy per horsepower-hour as engines (DeLuchi, 1993).

In the current version of the model, the installed hp-hour capacity of turbines is slightly less than the installed capacity of engines. Because engines use energy more efficiently, the total energy used by turbines equals the total energy use by engines. Specifically, I assume that 49.4% of pipeline energy is used in turbines, 49.4% in engines, and 1.3% in electricity-driven compressors (Table 4 of DeLuchi [1991]). The EPA states “for reciprocating engines, two stroke designs contribute approximately two-thirds of installed capacity” (p. 3.2-1). I assume that 2 / 3 of the energy used by reciprocating engines is used in 2-cycle lean burn engines, that 1 / 6 is used in 4-cycle

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[65] Against this, the fifth edition of AP-42 (EPA, 1995) states that “population statistics show a nearly equal installed capacity of turbines and reciprocating engines” (p. 3.2-1).

lean-burn engines, and that 1/6 is used in 4-cycle rich-burn engines. This is consistent with the emission estimates in the EPA/GRI (1996) report.

Note on natural gas storage

Natural gas is stored in depleted oil and gas reservoirs, salt caverns, and aquifers in order to buffer seasonal or weekly variations in gas demand. Gas is added to storage during periods of low demand, and withdrawn during periods of high demand. In 1995, 2.6 TCF of natural gas were added to storage facilities and 3.0 TCF were withdrawn (EIA, *Natural Gas Annual 1995*, 1996). Total storage capacity is expected to increase by about 10% by the year 2000 (EIA, *The Value of Underground Storage in Today's Natural Gas Industry*, 1995).

It takes energy to move natural gas in and out of storage facilities. This energy -- or at least the portion that is provided by using natural gas a fuel -- is counted as pipeline fuel in the EIA's statistics. Form EIA-176, "Annual Report of Natural and Supplemental Gas Supply and Disposition," asks respondents to report the amount of gas "used in pipeline, storage, and/or distribution operations" (EIA, *Natural Gas Annual 1995*, 1996, p. 217, survey p. 4). (Virtually all of the storage sites are operated by pipeline and distribution companies, [EIA, The *Value of Underground Storage in Today's Natural Gas Industry*, 1995].) Hence, the energy requirements of storage operations should be included in the EIA's projections of pipeline fuel in its *AEO*.

Transmission of natural gas as LNG

Previous versions of the LEM have had LNG as an end-use transportation fuel, where the LNG is made in the *consuming* country from domestic pipeline gas. However, these earlier versions did not include LNG made from foreign ("remote") natural gas and transported to the consuming country via LNG tankers. Because several analysts recently have projected increasing production of LNG worldwide (Flower and King, 2003; *Inside Fuels and Vehicles*, 2003; EIA, "U. S. LNG Markets and Uses, 2003; Valais et al., 2001), the LEM now includes LNG as a possible component of the natural-gas production-and-use-system. With this addition, the LEM represents the natural-gas system in two ways:

With international LNG transport	Without international LNG transport
• foreign gas recovery	• domestic gas recovery
• foreign gas processing	• domestic gas processing
• foreign gas liquefaction	
• international LNG transport	
• domestic LNG offloading and regasification	
• domestic gas transmission & storage via pipeline	• domestic gas transmission & storage via pipeline

• domestic gas distribution via pipeline to end users (or to central liquefaction plants)	• domestic gas distribution via pipeline to end users (or to central liquefaction plants)
• domestic gas compression or liquefaction for vehicles	• domestic gas compression or liquefaction for vehicles
• domestic distribution of LNG via truck to end users (if applicable)	• domestic distribution of LNG via truck to end users (if applicable)

Thus, LNG links gas production in one country with pipeline transport in another. Note that in this representation the LNG subsystem is part of the NG system; it is *not* a separate, complete alternative-fuels pathway. Note too that the LEM still has LNG as an end-use fuel made from the pipeline gas. This means that if the user specifies LNG as an end-use transportation fuel, and the country in question imports some gas as LNG, the LNG is offloaded at the marine terminal, regasified, shipped via pipeline to a liquefier, re-liquefied, then delivered as LNG to the end user. It is not possible in the LEM to specify that imported LNG be used directly as a transportation fuel, at the marine terminal, rather than be regasified and shipped via pipeline.

Because LNG imports are a minor fraction of gas consumption in all but a handful countries in the world (most notably Japan and Korea), I have kept the representation of the LNG system relatively simple. Most importantly, I have assumed that all of the process energy throughout the entire LNG chain – gas liquefaction, LNG transport, and regasification – is provided by natural-gas turbines. This general assumption, and my specific assumptions regarding BTUs-process-energy/BTU-LNG-delivered, are based on the following:

• GM et al. (2002b) present data and analysis that indicate the following for an LNG system:

-- liquefaction and loading: 0.09 to 0.14 BTU-NG/BTU-LNG, depending mainly on the efficiency of the gas turbine system;

-- LNG transport: 0.319 MJ [LHV]/tonne-km or about 475 BTU [HHV]/ton-mile (where the energy figure includes fuel used for the return trip)

-- LNG transport: 0.07 BTU-fuel/BTU-LNG-delivered for 10,200 km;

-- regasification: 0.01 BTU-LNG/BTU-gas.

• URS Australia (2002) provides a complete input-output analysis for a large modern LNG facility proposed for Australia. Their analysis indicates 0.154 BTUs of natural gas are consumed in gas turbines for each BTU of LNG produced. This however includes processing of raw gas to remove sulfur, carbon dioxide, and other impurities, a step which is accounted separately in the LEM. This step probably consumes around 0.020 BTUs/BTU gas, which means that the consumption of the LNG facility itself is about 0.13 BTUs-NG/BTU-LNG.

- URS Australia (2002) also provides a complete analysis of emissions of all pollutants from all sources of LNG operations. Their analysis indicates that vast majority of the emissions -- including emissions of methane from all sources -- come from gas turbines. The gas-turbine emission factors by URS Australia (2002) are similar to the ones used in the LEM.
- Valais et al. (2001) report that in recent years LNG plants have gotten more efficient and much larger. They also report that "major advances anticipated for the new generation of LNG tankers," including improved insulation, less evaporation, reliquefaction of evaporated gas, and more efficient propulsion modes including diesel/electric drives that can use evaporation gas as well as diesel oil. Finally, they report that regasification terminals have become more efficeint as well.
- Summing up progress in energy efficiency, Valais et al. (2001) report that the energy "self-consumption" of the overall LNG chain has declined from 15 to 20% in the 1960s and 1970s to 12 to 15% in the 1980s and "apparently" as low as 8 to 10% for the latest projects. However, it is not clear what energy requirements are met by this "self consumption".
- Various papers presented at the recent GasTech (2002) conference in Qatar, and information on LNG consulting website (www.users.qwest.net/~kryopak/LNGships.html) indicate that there are at least several kinds of propulsion systems for LNG tankers: diesel engine systems (with reliquefaction of boil off gas), natural-gas turbines (which use boil-off gas and LNG as needed), ships with electric drives, and dual-fuel gas/diesel systems (which use available boil-off gas and then diesel fuel as needed). The choice of propulsion systems depends mainly on cost factors.
- Flower and King (2003) state that "typcally 2 to 3 % of gas is used or lost in the regasification process" (p. 2).

Given these data, I make the following assumptions:

- Liquefaction and loading requires 0.12 BTUs-NG/BTU-LNG
- LNG transport requires 475 BTUs [HHV]-LNG/ton-mile-LNG (where the numerator includes energy required for the empty backhaul, and the denominator refers to the one-way transport distance)
- regasification requires 0.01 BTUs-LNG/BTU-LNG delivered

As mentioned above, I assume that all of this fuel is used in large natural-gas turbines. To calculate the total energy requirement of LNG transport, I mulitply the energy consumption per ton-mile by the one-way transport distance; the transport distance is calculated on the basis of the distance from producing countries to the consuming countries weighted by the contribution of each producing country. My assumptions regarding LNG transport result in about 0.66 BTU/BTU-LNG for 10,200 km transport, very close to the figure calculated by GM et al. (2002b).

Pipeline transmission of hydrogen

There are four general hydogen production and transmission pathways in the LEM:

- Hydrogen produced from water via electrolysis, then transmitted as a gas via pipeline to refueling stations with compressors or small-scale liquefiers;
- Hydrogen produced from water via electrolysis, then transmitted as a gas via pipeline to large centralized liquefaction facilities, from which liquid hydrogen is transmitted mainly via truck to refueling stations;
- Hydrogen produced from steam reforming of natural gas at the site of refueling;
- Hydrogen produced from steam reforming of natural gas at large centralized facilities, then liquefied and shipped mainly via truck to refueling stations.

Parameters regarding hydrogen-production are discussed in the major section on production of alternative fuels. Parameters regarding compression and liquefaction and leakage from fueling stations and truck transfers are discussed in the major section on fuel marketing and dispensing. Parameters regarding truck transport of LH_2 are discussed in the section “Distribution of LNG and LH_2.” Parameters regarding energy use and leakage of hydrogen pipelines are discussed in this section.

Energy intensity of pipeline transmission. The energy intensity of pipeline transmission of hydrogen from water-electrolysis sites to end users or centralized liquefaction facilities is estimated as a function of the transmission distance and compression energy relative to that for natural gas:

$$EITH2 = EIT \cdot RTD_{H2?NG} \cdot RPCE_{H2?NG} \qquad \text{eq. 94 g}$$

where:

EITH2 = the energy intensity of hydrogen transmission by pipeline (BTU-pipeline-energy / BTU-hydrogen-delivered)
EIT = the energy intensity of natural gas transmission (estimated above)
$RTD_{H2?NG}$ = the transmission distance for hydrogen relative to that for natural gas (discussed below)
$RPCE_{H2?NG}$ = the compression energy for hydrogen relative to that for natural gas (in terms of BTU / BTU / mi) (discussed below)

The transmission distance for hydrogen relative to that for natural gas depends on the distribution of the sources of electricity for water electrolysis (which produces hydrogen) compared with the distribution of natural gas fields, relative to end-use markets. These distributions will vary from country to country and region to region, and in the case of hydogen will depend on the source of electricity (hydropower, nuclear, or solar power). I assume that nuclear power and hydropower sites are more decentralized (and closer to end users) than are natural gas-production fields, and hence that the parameter RTD is equal to 0.60, in the case of hydrogen delivered to end users and compressed, and 0.50 in the case of hydrogen delivered to centralized liquefaction plants.

Models of the work requirements of compression, such as are disussed in this report and in DeLuchi (1992), and information cited in Appendix L of DeLuchi (1993), indicate that it takes about 2.5 time more energy to compress a BTU of hydrogen to a given pressure than it does to compress a BTU of NG. I assume this value here for the parameter RPCE.

Leakage from hydrogen pipelines. Hydrogen leakage from pipelines is estimated separately for transmission (from production fields to city gates or centralized liquefaction facilities) and distribution (from city gates to end users, in the case of compressed hydrogen or small-scale liquefaction).

In the case of transmission, leakage is estimated as a function of the transmission distance relative to that for natural gas and the leakage rate per mile relative to that for natural gas, using a leakage analog of equation Eq. 94 g. The relative transmission distance is assumed to be as estimated for the parameter RTD in equation 94 g. To estimate the relative leakage rate per mile, one has to consider that on the one hand a hydrogen molecule is smaller and lighter than a methane molecule and hence more prone to escape, but that on the other hand for reasons of safety and economics it may be worthwhile to build and operate hydrogen systems so that the leakage rate is equal to or less than that for natural gas. I assume that the ratio of the hydrogen leakage rate to the natural-gas leakage rate per mile is 1.50 – less than what would be expected purely on the basis of the relative mobility of the two gases, but probably higher than what could be achieved with best practice. In support of this, Zittel (1996) reports a relatively low leakage rate of only 0.1% from an industrial hydrogen distribution system in Germany.

In the case of pipeline distribution from the city gate to end users, leakage is estimated as a function of the relative leakage rate per mile, discussed above.

SHIPMENT OF FEEDSTOCKS, FUELS AND VEHICLES

Distribution of coal, crude oil, and petroleum products: general method

In DeLuchi (1993), the energy used to distribute coal, crude oil, and petroleum products was calculated and input to the model on the basis of historical data on tons and ton-miles of shipments of coal, oil, and products, by mode, in 1987. This was different from the method used to calculate the energy used to distribute methanol, ethanol, and LPG (Table E.1b of DeLuchi [1993]). For those fuels, distribution energy for mode M was calculated per ton of fuel shipped by mode M, as the product of: an assumed average length of haul by mode M (miles), energy intensity (BTU/ton-mile-shipped by mode M), and a modal usage factor (tons of fuel shipped by mode M per ton of fuel produced). Now, I have changed the basis of the calculation for coal, crude oil, and petroleum products to be the same as the basis of the calculation for ethanol, methanol, and LPG. Thus, for all fuels, distribution energy is calculated per ton of fuel produced, as the product of miles, BTU / ton-mile, and tons shipped by M per ton produced:

$$E/TP_{F,M,T} = TS/TP_{F,M,T} \cdot LH1W_{F,M,T} \cdot EI_{F,M,T_B} \cdot \left(1+\frac{\Delta EI_{F,M}}{100}\right)^{T-T_B} \qquad \text{eq. 95}$$

where:

F = fuel being distributed (coal, crude oil, light petroleum products, heavy petroleum products, methanol, ethanol, LPG).
M = distribution mode (domestic ship, foreign ship, rail, pipeline, or truck).
T_B = base year for energy intensity data.
T = target year of the analysis.
$E/TP_{F,M,T}$ = the energy consumed in target year T by distribution mode M per ton of fuel F produced (production in this context includes field production + factory or refinery production + imports + stock changes) (BTU / ton).
$TS/TP_{F,M,T}$ = tons of fuel F shipped by mode M per ton of fuel F produced in target year T.
$LH1W_{F,M,T}$ = the one-way length of haul per average ton[66] of fuel F by mode M in target year T (miles).
$EI_{F,M,T}$ = the energy intensity of mode M hauling fuel F in base year T_B (BTU / ton-mile) (see below).
?$EI_{F,M}$ = the annual percentage change in the energy intensity of mode M hauling fuel F.

This unifies the input, presentation, and interpretation of data. Most of the primary data sources are the same as those used in DeLuchi (1993).

Note that the mileage, LH1W, is the one-way distance, not the round-trip distance. This, of course, is because the fuel in question (say, coal), is shipped only one way; hence, to calculate ton-miles, one multiplies tons by the one-way shipping

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[66] The length of haul per average ton is not the same as the length of haul per average shipment. This distinction matters because the Bureau of the Census *Commodity Flow Survey* (1996, 1999), which serves as the basis for some of my estimates of TS / TP, provides estimates of tons, ton-miles, and average miles per shipment. Ton-miles is calculated by multiplying the tonnage of each shipment by the mileage of the shipment. Given this, the length of average haul per ton is equal to ton-miles divided by tons. This is not necessarily the same as the average miles per shipment. To see this, suppose that there are two shipments, one of 1 ton for 1 mile, the other of 100 tons for 100 miles. The average miles per shipment is 50.5 miles, while the average haul per ton is equal to the total ton-miles (10,001) divided by total tons (101), or 99 miles.

distance. Now, if the ship returns empty, then the energy used in the empty backhaul must be counted in the total energy E in the calculation of energy intensity EI:

$$EI_{F,M,T_B} = \frac{E_{F,M,T_B}}{TS_{F,M,T_B} \cdot LH1W_{F,M,T_B}} \qquad \text{eq. 96}$$

where:

$E_{F,M,Tb}$ = energy used by mode M to ship fuel F in base year $T_{B,}$ including energy used for empty back-hauls (BTUs).
$TS_{F,M,Tb}$ =fuel F shipped by mode M per in base year T_B (tons).
$LH1W_{F,M,Tb}$, $EI_{F,M,Tb}$ are as defined above.

So, if the carrier returns empty, then E includes the energy used on the empty backhaul; if the carrier returns with another product, E includes only the energy used to haul fuel F one way. As noted in Appendix E of DeLuchi (1993), virtually all ships return empty.

International waterborne shipment of crude oil, petroleum products, and coal: estimated tons-shipped/ton-produced, and average length of haul

The estimates of tons shipped per ton produced (parameter TS/TP in Eq. 95) and average length of haul (parameter LH1W in Eq. 95) for international waterborne shipment of crude oil, petroleum products, and coal have been refined and updated. TS/TP now is calculated on the basis of estimated petroleum or coal supply from producing countries (PCO, PPP, or PCL) to the country C of interest, and the fraction of the supply shipped by international water:

For crude oil (CO):

$$TS/TPCO_{C,T} = \sum_{PCO} CCO_{PCO,C,T} \cdot FCOW_{PCO,C} \qquad \text{eq. 97a}$$

where:

subscript C = petroleum or coal-consuming country selected for analysis (U. S. [US], in the base case).
subscript T = the target year of the analysis.
subscript PCO = countries that produce crude oil.
$TS/TPCO_{C,T}$ = tons of crude oil shipped by international waterborne commerce per ton of crude oil supplied (imported plus produced domestically) in country C in year T.

$CCO_{PCO,C,T}$ = the contribution of crude-oil-producing country PCO to crude oil supplied in country C in year T (a fraction of total crude supply in country C).

$FCOW_{PCO,C}$ = of the total crude oil produced by country PCO for country C, the fraction that is shipped to C via international water (assumed to be 0.0 for PCO = C [domestic production] and for any two countries that have extensive pipeline, rail, or road transport between them; 1.0 for all other PCO-to-C supply).

For light or heavy products or coal, substitute PP (products) or CL (coal) for CO (crude oil) in TPCO, CCO, PCO, and FCOW in Eq. 97a

The average length of haul, LH1W, now is calculated on the basis of the distance from each exporting country to the country of interest C, and the amount of crude oil, petroleum product, or coal shipped from the exporting country to C. For crude oil:

$$LH1WCO_{C,T} = \frac{\sum_{PCO} CCO_{PCO,C,T} \cdot LH1W_{PCO,C} \cdot FCOW_{PCO,C}}{TS\ /\ TPCO_{C,T}}$$

eq. 97b-c

$$LH1W_{PCO,US} = \frac{\sum_{PADD} CO(V)_{PCO,PADD,94} \cdot LH1W_{PCO,PADD}}{\sum_{PADD} CO(V)_{PCO,PADD,94}}$$

where:

subscript PADD = major port in each Petroleum Administration Defense District (PADD) of the U. S., designated as follows:

	PADD I	PADD III	PADD V
region	East Coast	Gulf Coast	West Coast
major port	New York	Houston	Los Angeles

$LH1WCO_{C,T}$ = the ton-weighted average length of haul, by international ocean transport, of crude oil used by country C in year T (miles).

$LH1W_{PCO,C}$ = the average length of haul, by international ocean transport, of crude oil from producing country PCO to consuming country C (miles; for distances to countries C other than the U. S., see Appendix B).

$PP(V)_{PPP,PADD,94}$ = petroleum products supplied from producing country PPP to U. S. port PADD in 1994 (EIA, *PSA 1994,* 1995).

$CO(V)_{PCO,PADD,94}$ = crude oil supplied from producing country PCO to U. S. port PADD in 1994 (EIA, *PSA 1994*, 1995).

$LH1W_{PCO,PAD}$ = shipping distance from producing country PCO to U. S. port PADD (miles; based on port-to-port distances provided by the Defense Mapping Agency [1985], plus my estimates as necessary[67]); assumptions regarding ports as follows:

Crude oil -- region (country [port])	*Products -- region (country [port])*
Mexico [Tampico]	Northern Europe (United Kingdom, Belgium, Netherlands [Rotterdam], and others)
North Sea (United Kingdom, Norway [Bergen])	Southern Europe (Spain, France, Italy [Naples])
OPEC	OPEC
Venezuela [Maracaibo]	Latin America (Venezuela [Maracaibo])
North Africa (Algeria [Algiers])	North Africa (Algeria [Algiers], Libya)
West Africa (Nigeria [Lagos], Gabon)	West Africa (Nigeria [Lagos], Gabon)
Indonesia [Jakarta]	Indonesia [Jakarta]
Persian Gulf (Saudi Arabia [Ad Damman], Kuwait)	Persian Gulf (Saudi Arabia [Ad Damman], Kuwait)
Other Middle East (Oman [Matrah], Yemen)	
Other Latin America (Colombia [Cartegena], Trinidad and Tobago)	Caribbean Basin (Virgin Islands [Charlotte Amalie], Netherlands Antilles, Mexico, and others)
Other Africa (Angola [Luanda])	
Other Asia (China [Shang Hai])	Asian Exporters (Korea [Pusan], Singapore, and others)

I selected producing regions and countries to capture most of world output for export.

For light or heavy products, substitute PP (products) for CO (crude oil) in all parameters in Eq.97b-c. For coal, substitute CL (coal) for CO in Eq. 97b.

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[67] For example, I assume that the distance to Houston is 50 miles more than the distance to Galveston; that the distance to Rotterdam is 120 miles less than the distance to Bergen; and that the distance to Matrah, Oman is 700 miles less than the distance to Ad Damman, Saudi Arabia.

Note that in the case of the U. S., the overall average shipping distance is the barrel-weighted average distance to the 3 PADD ports, where the barrel weights are based on 1994 receipts at each PADD. To the extent that the proportions going to different ports vary from year to year, the average distance to the U. S. will vary at least slightly from year to year. However, this undoubtedly is a quite minor effect.

Domestic waterborne shipment of crude oil and petroleum products: estimated tons-shipped/ton-produced, and average length of haul

The estimates of tons shipped per ton produced (parameter TS/TP) and average length of haul (parameter LH1W) for domestic waterborne shipment of crude oil and petroleum products have been updated on the basis of 1994 and 1995 data from the Army Corps of Engineers' *Waterborne Commerce* (1995) and the EIA's *PSA 1994* (1995). Table 29 shows the 1994 data for tonnage shipped, ton-miles shipped, and tons produced, by commodity, and the calculated TS/TP and LH1W. On the basis of the calculated values shown in Table 29, new values have been input for TS/TP and LH1W.

In consideration of these updated values for petroleum products, some of the assumptions for TS/TP and LH1W for methanol transport have been changed as well.

Domestic waterborne shipment of coal, crude oil, and petroleum products: energy intensity

In Table E.1.a of DeLuchi (1993), I assumed that vessels that carry coal domestically have energy intensity (parameter EI) of 500 BTU/ton-mile, and that vessels that carry petroleum domestically have an energy intensity of around 200 BTU/ton-mile. My assumption for coal vessels was based on estimates in Table E.2, and my assumption for petroleum was based on estimates of the energy intensity of tankers of different sizes.

I have revisited these assumptions, and looked more closely at the types of vessels that carry each type of commodity. Data from the Army Corps of Engineers' *Waterborne Commerce* (1995) indicate that barges haul a high fraction of the coal moved by water (87.5% of all coal ton-miles by water), and a significant fraction of domestic petroleum products, but essentially no crude oil (Table 29). Moreover, the average domestic petroleum-product tanker apparently is smaller than previously assumed: according to the EIA (*The Energy Information Administration's Assessment of Reformulated Gasoline*, 1994), a typical U. S.-flag petroleum-product tanker operating in U. S. waters is less than 50,000 dwt, whereas in Table E.5 of DeLuchi (1993), I assumed that the bulk of petroleum-product tankers in domestic service are in the 60,000 or 90,000 dead-weight-ton (dwt) size class.

Barges have an energy intensity of on the order of 280 to 480 BTU/ton-mile (Rose, 1979; Booze-Allen Hamilton, 1977); here, I assume 350 BTU/ton-mile. Smaller tankers are more energy intensive than larger tankers (Table E.5 of DeLuchi [1993]). Shifting the distribution of tankers carrying petroleum products towards the lighter dwt classes increases the weighted-average BTU/ton-mile by about 10%, to 213

BTU / ton-mile. The size distribution and hence average EI of crude oil tankers remains unchanged.

Given this, I now calculate a weighted average energy intensity (EI) for domestic waterborne commerce, equal to the EI for barges multiplied by the fraction of ton-miles by barge, plus the EI for tankers multiplied by the fraction of ton-miles shipped by tankers. The EI factors are given above, and the ton-mile fractions are based on the data in *Waterborne Commerce* (Army Corps of Engineers, 1995)

Pipeline shipment of crude oil and petroleum products: estimated tons-shipped/ton-produced, and average length of haul

In DeLuchi (1993), shipping parameters for crude oil and petroleum products on the basis of ton-mile data were estimated from the Association of Oil Pipelines (AOP). I have estimated new ton-shipped / ton-produced and average-length-of-haul parameters on the basis of data reported by the Bureau of Transportation Statistics (1996), the *CFS* (Bureau of the Census, 1996, 1999), and EIA's *PSA 1996* (1997). My estimates are shown in Table 29, and discussed in the notes to that table and in the text here.

Tons-shipped per ton produced. To estimate tons-shipped / ton-produced, we need to know tons shipped by pipeline, and total tonnage produced. Table 29 shows our estimate of total tonnage produced, which is based on total petroleum products supplied[68]. To estimate tons shipped, we refer to several data sources.

The Eno Foundation's *Transportation in America* estimates the following for 1994 (as reported by the Bureau of Transportation Statistics, *National Transportation Statistics 1997*, 1996):

	10^6 ton-miles shipped	10^6 tons shipped	average length (miles)
crude oil	322,600	not reported	756
petroleum products	268,800	not reported	400
total	*591,400*	*1057.9*	*not reported*

However, if one calculates the missing tonnage data for crude oil and petroleum products as ton-miles divided by miles, the resultant tonnages do not add to the reported total of 1057.9 (Eno's reported ton-miles are similar to the ton-miles reported by the AOP, as cited in DeLuchi (1993), most likely because the Eno Foundation, the source of the Table 29 data, uses the AOP statistics.) Thus, the reported figures are

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[68] Product supplied is equal to field production + refinery production + imports - stock change - exports. It is appropriate to include imports here because the *CFS* data "include imported products at the point that they left the importers domestic location for shipment to another location" (Bureau of the Census, 1996, p. vi). In other words, the *CFS* includes all domestic movement of imported petroleum after it has landed in the U. S. It is appropriate to exclude exports because exported petroleum is shipped from the West Coast or the Gulf Coast (EIA, *PSA 1996*, 1997), where the refineries generally are located on the coast, and presumably load exported petroleum directly onto international tankers.

mutually inconsistent. They appear, though to indicate about 400 million tons of crude oil and 600 million tons of products shipped by pipeline. By comparison, the *1993 CFS* reports that in 1993, about 430 million tons of "the products of petroleum refining" were shipped via pipeline[69] – considerably less than the 600 million indicated by the Eno data for 1994. Turning to crude oil, the EIA's *PSA 1996* (1997) reports that in 1996, refineries received 1.85 billion bbls (278 million tons) of domestic crude and 0.92 billion bbls (138 million tons) of foreign crude via pipeline, for a total of 417 million tons. (The *CFS* does not cover crude oil.) Given these data, my assumptions are shown in Table 29.

Average length of haul. I assume that the data from the Eno Foundation shown above are roughly reasonable. (Note that in the 1980s, the average length of haul of crude oil was at least 800 miles.) The *CFS* does not report the length of shipment on the main trunks of pipelines.

Truck shipment of petroleum products: tons-shipped/ton-produced, and average length of haul

A minor part of emissions of greenhouse gases from the oil fuel cycle is emissions from trucks that transport petroleum products. When the 1993 ANL report (DeLuchi, 1993) was written, there were no reliable, complete data on tons, ton-miles, and miles of shipment of petroleum products by trucks. DeLuchi (1993) used five different data sources and methods to estimate ton-miles of travel by trucks carrying petroleum products (pp. H-38 to H-40; Table E.1a). Final estimates were based on data from the 1982 *Truck Inventory and Use Survey* (*TIUS*, Bureau of the Census, 1985), which reported miles, but not ton-miles, by trucks carrying petroleum products, and noted in Table E.1a (DeLuchi, 1993) that the estimate was "rough"[70].

Recently, somewhat better data on tons-shipped / ton-produced, and average length of haul, have become available. The 1993 *CFS* (Bureau of the Census, 1996) reports tons and ton-miles of shipments of "products of petroleum refining" (STCC 291) in the U. S. in 1993. According to the 1993 CFS, about 860 million tons of petroleum products were shipped an average 50 miles (about 43 billion ton miles) in 1993. The 860 million tons shipped by truck is 95% of the 880 million tons supplied (see pipeline section above); thus, we can conclude that virtually all petroleum products were shipped by truck at some point.

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[69] (The *1997 CFS* reports that in 1997 565 million tons of gasoline, aviation fuel, fuel oil, and other petroleum products were shipped via pipeline [Bureau of the Census, 1999].)

[70] Notice that the discussion on pages H-38 and H-40 of DeLuchi (1993) does not treat empty backhaul correctly. On pages E-4 and E-16 I present the correct method, which incorporates into the "BTU / ton-mile" energy intensity figure the energy consumed during the empty backhaul. With the backhaul energy incorporated into BTU / ton-mile, the correct "ton-mile" figure with which to multiply BTU / ton-mile is one that *excludes* empty back-haul miles. However, on pages H-38 and H-40, I incorrectly include empty back-haul miles in my analysis of the ton-mile data.

Data from the *1997 CFS (*Bureau of the Census, 1999) indicate that in 1997 gasoline and aviation fuel were shipped an average of 56 miles per ton, and that fuel oils were shipped an average of 54 miles per ton.

The average shipping length (50 miles) and the estimated ton-miles (43 billion) from the CFS can be confirmed independently. The EIA's *Alternatives to Traditional Transportation Fuels* (1994) cites an estimate by the National Petroleum Council of a one-way haul of 40 miles. The ton-mileage can be checked using data from the most recent *TIUS*, as detailed below.

The 1992 *TIUS* (Bureau of the Census, 1995) reports truck miles of travel by trucks carrying petroleum products[71], in 14 weight categories (Table 31). The weight categories refer to average total weight *when loaded.* With these data and some assumptions, one can calculate ton-miles of product and average weight of product for 1992.

To do this calculation, one must make four sets of estimates or assumptions. First, one must estimate the actual average loaded weight *within* each of the 14 weight classes. All of the classes except the first and last are small enough that the midpoint of the class must be a reasonable approximation. However, the first class is "less than 6,001 lbs), and the last is "130,001 or more". I assume that trucks less than 6,001 lbs weigh 5,000 lbs, and that trucks more than 130,001 lbs weigh 150,000 lbs. (Table 31). I expect that the resulting estimated average loaded weights are very close to the true average weights.

Second, one must deduct the weight of the empty vehicle from the reported assumed average weight, to get the weight of the product carried. I have made a separate assumption for each weight class. The assumed empty vehicle weight ranges from 3,700 lbs to 33,000 lbs, and is a progressively smaller fraction of the loaded weight (Table 31). There probably is a 10% to 30% error in my assumptions.

Third, one must estimate the fraction of total miles with the average load. I assume that half of the total truck miles are empty, and that half are with the average load.

With the data of Table 31, I calculate that trucks that carried petroleum products in 1992 had an average product load of 13.8 tons, and transported the products 42.2 billion ton-miles, excluding empty back-haul mileage which I presume to be half of total truck mileage. This is quite close to the CFS estimate of 43 billion ton miles for 1993[72].

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[71] The "petroleum" category in the *TIUS* includes only products; crude oil is included under "mining products"

[72] This implies about 35 billion ton-miles in 1987 (2.55 billion truck miles of loaded travel in 1987 multiplied by my assumption of 13.8 tons per truck [same as estimated for 1992]), the base year for the analysis in DeLuchi (1993) -- much less than the 125 billion assumed in Table E.1a of DeLuchi (1993).

Train, water, truck, and pipeline transport of coal

The EIA, the Bureau of the Census, the Army Corps of Engineers (ACE), and the Interstate Commerce Commission (ICC) independently collect data on the shipment of coal by rail, water, truck, and pipeline. Table 30 summarizes ton and ton-mile data from these surveys, for 1993 and 1995. These data were used to estimate tons-shipped per ton produced (TS / TP), and the average length of haul one way (LH1W).

The Census' *CFS* reports tons and ton-miles by all modes; the EIA's Coal Distribution survey reports tons by all modes; the ACE's *Waterborne Commerce* reports tons and ton-miles by water; and the ICC's *Waybill* reports tons and ton-miles by rail. The estimates of TS / TP were based on the EIA's tonnage data for three reasons: the EIA's is a comprehensive survey of coal producers and distributors; the EIA data agree with the ACE data on water transport, and the ICC data on rail transport[73]; and the EIA transport data can be used with the EIA's production data to produce a measure of tons-shipped / ton-produced (TS / TP). These TS / TP estimates are summarized in Table 30, part B.

The EIA's study of coal transportation patterns (*Energy Policy Act Transportation Rate Study*, 1995) indicates that for rail shipment of coal to power plants, TS / TP has remained relatively steady, but for barges it has increased, and probably will continue to increase. I assume an increase of 0.4% / year (coal to power only).

Table 30, part B, also shows estimates of the average length of haul one way (LH1W), based mainly on the CFS data. The CFS-based estimates are consistent with average distance shipped per ton of coal contracted for shipment by electric utilities from 1979 to 1993: about 600 miles by train, 300 miles by barge, 30 miles by truck, and 60 miles by other (EIA, *Energy Policy Act Transportation Rate Study*, 1995). The EIA also reports that shipment distances by train have increased, mainly because power plants are using more and more low-sulfur coal from the West. I assume that shipment distance to power plants by train will increase by 0.4% per year.

Pipelines. Previously, I estimated tons shipped and average length for the Black Mesa slurry pipeline, and then made an ad-hoc adjustment to shipping distance to account for all distribution by tramway and conveyor belt. It turns out, however, that much more coal tonnage is shipped by tramway and conveyor than by slurry pipeline, albeit for a much shorter distance[74]. Table 30 presents the estimated TS / TP and average shipment length for the pipeline, tram, and conveyor combined. On the basis of these

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[73] It appears that, on account of its breadth, the Census' *CFS* is the least accurate for any particular commodity and mode: the other three sources agree with one-another but not with the CFS.

[74] The slurry pipeline from the Black Mesa mine in Arizona to the Mohave Generating Station in Nevada carries 4.8 million tons of slurry over 273 miles, for a total of 1.3 billion ton-miles (EIA, *Energy Policy Act Transportation Rate Study*, 1995). The data of Table 30 indicate that about 100 million tons are carried an average of about 50 miles by conveyor or tram, a total of about 5 billion ton-miles.

data, new assumptions for pipeline / tramway / conveyor transport of coal[75] have been used.

In Table E.1a I assumed 600 BTU-electric / ton-mile for the Black Mesa coal slurry pipeline. As discussed in note e to Table E.2, only about half of this energy is required to actually move the coal; the rest is used to prepare the coal and to de-slurry it. It is doubtful that tramways and conveyor belts, which as just noted carry much more coal than do long-distance coal-slurry pipelines, consume nearly as much energy per ton-mile as do coal-slurry pipelines. I assume an average of 250 BTU-electric / ton-mile (in the base year) for all pipeline, tramways, and conveyor belts.

Disposal of byproducts of coal combustion

The calculation of the energy used to dispose of the byproducts of coal combustion has been refined. Diesel fuel used to transport byproducts, per ton of coal produced, is estimated as:

$$DCW = AF \cdot AW \cdot (1 + FGD) \cdot (1 - WM) \cdot \big(OFF \cdot OFFLH + (1 - OFF) \cdot ONLH\big) \cdot BTU / TM$$

eq. 98

where:

DCW = diesel fuel consumed to transport the byproducts of coal combustion (BTUs / ton-coal-produced).
AF = ash content of coal (weight fraction; Table 4).
AW = of the total ash content of the coal, the fraction that ends up as byproduct (weight basis; 0.87, according to EIA data [*Coal Data, A Reference*, 1995] on the production and use of byproducts from utility coal combustion in 1992).
FGD = weight of sludge from flue-gas desulphurization (fraction of the weight of the ash byproduct; 0.24, according to EIA data [*Coal Data, A Reference,* 1995] on the production and use of byproducts from utility coal combustion in 1992).

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[75] It is conceivable that some of the energy used to power tramways and conveyor belts is reported as mining energy in the Bureau of the Census survey of fuels and electric energy consumed by mineral industries. However, if the companies follow the Census' instructions, there should be virtually no overlap. In the survey of fuels and electric energy consumed by mineral industries, the Census specifically excludes "hauling and other transportation beyond the mine property (except out of open pits in conjunction with mining)" (Bureau of the Census, *1987 Census of Mineral Industries*, 1991, p. v). This nicely complements the EIA's survey of coal distribution, which covers coal distributed from mines other consumers (EIA, *Coal Industry Annual 1993*, 1994). I presume, then, that none of the energy that I calculate for coal distribution is counted already as energy used for coal mining.

WM = of the total byproduct ash and FGD sludge produced, the fraction that is marketed, rather than simply disposed of in a landfill (0.31, according to EIA data [*Coal Data, A Reference,* 1995] on the production and use of byproducts from utility coal combustion in 1992; Spath et al. [1999] assume 0.25 for FGD sludge and 0.28 for ash).

OFF = of coal byproduct that is sent to landfills, the fraction that is sent to land fills off site from the plant (0.15; my assumption).

OFFLH = the average length of haul from the plant to the off-site disposal (20 miles; my assumption [Spath et al., 1999, assume 5 km for all disposal, on-site or off]).

ONLH = the average length of haul from the plant to the on-site disposal (1 mile; data cited by Mann and Spath [1997]).

BTU / TM = energy consumed by trucks transporting byproducts (4,000 BTU / ton-mile; based on data cited by Mann and Spath [1997]).

The portion of the byproduct that is marketed may displace the production of other materials, and hence reduce emissions of greenhouse-gas emissions in other sectors. This is accounted for in the model:

$$BDCW = \frac{AF \cdot AW \cdot (1+FGD) \cdot WM \cdot FDC_{CWB} \cdot GHG_{CWB} \cdot 2000}{HHV_{coal}} \quad \text{eq. 99}$$

where:

AF, AW, FGD, and WM are as defined in Eq. 98

BDCW = emissions (credit) from products displaced by the byproducts of coal use (g-CO_2-equivalent-displaced / 10^6-BTU-coal).

FDC_{CWB} = of coal byproducts actually marketed (WM), the fraction that displaces existing production (the remainder, 1-FDC, is assumed to satisfy net new demand) (I assume 0.75).

GHG_{CWB} = fuel cycle CO_2 equivalent GHG emissions from the products displaced by the byproducts of coal use (g-CO_2-equivalent / lb-product displaced) (discussed below).

2000 = lb / ton

HHV_{coal} = the higher heating value of coal (10^6-BTU / ton; e.g. Table 4).

Products displaced by coal byproducts. The EIA data (*Coal Data, A Reference,* 1995), and the discussion in Spath et al. (1999) on the production and use of the byproducts of coal combustion show that ash and FGD sludge byproduct is used mainly in concrete and cement products, as structural fill, and as a base or sub-base for roads. All products displaced by coal byproducts are assigned the fuel-cycle CO_2

equivalent emissions attributable to 50% concrete (about 150 g/lb) and 50% cement (about 600 g/lb). This results in an emissions-displacement credit equal to about 15-20% of total fuel cycle emissions from the production and transport (but not end use) of coal.

Transport of biomass to biofuel production facility.

Turhollow and Perlack (1991) and Mann et al. (1995) assume a 40 km haul from field to the fuel production or power generation facility. Mann and Spath (1997) assume that 70% of the wood is delivered (to a biomass power plant) by truck, and 30% by train, an average of only 27.6 km. Perlack et al. (1992) assume that biomass is hauled by truck 25 to 50 miles from the field to the biomass-to-ethanol conversion plant, depending on the site; for their analysis, I calculate a tonnage-weighted average of 34 miles for all sites. They also assume that a small fraction of biomass travels 90 miles by barge, and 140.5 miles by railroad. Wooley et al. (1999) assumed a collection radius of 40 miles for switch grass, and 23 miles for corn stover. Walsh (1998a) recommends assuming a 50-mile haul by truck.

My assumptions for wood and grass are similar to those of Perlack et al. (1992): all biomass travels 50 miles by truck, 10% travels 100 miles by barge, and 20% travels 150 miles by rail.

Note that in the calculation of biomass ton-miles, I use the actual weight of the biomass as transported, rather than the dry-weight basis used in the calculation of yields and carbon inventory. (Biomass is partially but not completely dried in the field before transport.) Perlack et al. (1992) estimated that actual transport weight is 125% of the dry weight. Walsh (1998a) recommended assuming that wood is transported with 50% moisture content, and grass with 13-15%. and these were used.

Transport of corn from farm to corn-to-ethanol facility

In Appendix K of DeLuchi (1993), the transport of corn from the farm to the ethanol facility was estimated to consume 5,600 BTU/bushel, mainly as diesel fuel used by trucks. More recent data allow a more detailed estimate.

First, I note that Shapouri et al. (2002) and Conway et al. (1994) point out that some transport energy actually is included already in the primary estimates of energy used in the corn-farming stage. Specifically, the *FCRS*, my primary source of data on energy use in corn farming, includes the fuel cost for transporting corn from the farm to the first point of sale or storage (including the return trip), typically a local grain elevator[76]. Therefore, what remains to be represented here is transport from this first point of storage or sale to the ethanol processor.

Eq. 95 was used to estimate the energy intensity of corn transport from local collection to the ethanol facility. This equation requires estimates of tons of corn

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[76] Conway et al. (1994) estimate that 25% of the total energy to transport corn to the ethanol plant is included already in the estimates of energy use of corn farming.

shipped by each mode per ton of corn produced for shipment (tons / ton), length of haul by each mode for the average ton shipped (miles), and the energy intensity of each mode (BTU / ton-mile). The last parameter is discussed elsewhere in this report. Here are discussed the tons-shipped / ton-produced and the average length of haul.

Miles per average ton. Grabowski (2002) cites estimates that corn is hauled about 50 miles by truck from the farm to the ethanol processor. However, Shapouri et al. (2002) assume much longer distances, by three modes:

- by truck from farm to collector (included in farm energy data)
- 40 miles by truck from collectors to terminals
- 350 miles by barges from terminals to ethanol plants
- 400 miles by rail from terminals to ethanol plants.

Finally, the Bureau of the Census (1999) *1997 CFS* reports the following transport data for "corn, except sweet" for the U. S. for 1997 (miles / ton were calculated by dividing their reported ton-miles by reported tons):

	10^3 tons shipped	calculated miles / ton
Truck	126,000	70
Rail	84,000	862
Domestic water (mostly barge)	53,000	842
Other and unknown (assumed mostly trucks)	41,000	55

The distances calculated from the Census data are much longer than the distances assumed by Shapouri et al. (2002). It is possible that ethanol facilities are located closer to corn producing regions than are other kinds of corn producers; however, Shapouri et al. (2002) do not offer any reasons why this should be so. I assume 50 miles by truck, 600 miles by rail, and 650 miles by barge.

Tons-shipped / tons-produced. This parameter can be estimated by dividing actual tons shipped, as reported by the Census and shown immediately above, by an estimate of total tons of corn produced for shipment. (This presumes that corn is shipped to ethanol facilities using the same modal mix as the average corn shipment to any processor.) The *1997 Census of Agriculture* (National Agricultural Statistics Service, 1999) reports that in 1997 8.58 billion bushels of corn were produced for grain or seed. Assuming that 95% of this production was shipped off the farm, and given 56 lbs per bushel, 228 million tons of corn was produced for shipment to processors in 1997. Using this in conjunction with the Census (1999) data on tons shipped by mode (shown above), the tons-shipped / ton-produced-for-ethanol are 0.70 for truck, 0.40 for rail, and 0.25 for domestic water.

Bulk distribution of ethanol from corn, ethanol from wood or grass, biodiesel from soy, and methanol from wood

EA Energy Technologies Group (1991; summarized in Bechtold and Wilcox [1993]) has done a detailed analysis of the distribution infrastructure required for expanded use of ethanol for transportation. They analyzed the modes that would carry neat ethanol from production sites to bulk terminals, on the basis of the location of production and consumption centers, total ethanol shipments, transportation distances, terrain, and the existing gasoline infrastructure (EA Energy Technology Group, 1991). They consider cellulosic as well as corn ethanol.

EA Energy Technologies Group (1991) estimated the straight line shipping distance between production and consumption centroids, and the amount of ethanol shipped by each mode between the centroids, for cellulosic and corn ethanol combined. Separating cellulosic from corn ethanol, the volume shares by mode are:

	Pipeline	Barge	Rail
corn ethanol	0.61	0.09	0.30
cellulosic ethanol	0.40	0.14	0.46
both	0.48	0.12	0.40

The gallon-weighted average *straight-line* shipping distances by mode are:

	Pipeline	Barge	Rail
corn ethanol	617	418	320
cellulosic ethanol	552	564	383
both	593	522	365

The actual shipping distance probably will be at least 10% longer than the straight-line distance. They assume that trucks carry ethanol 50 miles from terminals to bulk plants, and 50 miles from bulk plants to final consumers (EA Energy Technologies Group, 1991).

Note the relatively high share of rail in the transport of ethanol from cellulosic material. This broadly consistent with the a detailed analysis of the ethanol fuel cycle by the U. S. DOE (1994), in which ethanol from biomass is assumed to be distributed by rail and truck to consumers within a 200-mile radius of the production plant. (The fuel-production facilities will be relatively small and decentralized.)

Wang et al. (1998) assumed that bulk distribution of biodeisel from soy is similar to bulk distribution of ethanol from corn. This seems reasonable.

The tons-shipped / ton-produced, and average shipping distance, area based in part on the figures derived from EA Energy Technologies Group (1991). For methanol and ethanol from wood, I have decreased the length of haul by rail, barge, and pipeline, decreased the share of tonnage shipped by water, and increased the share shipped by rail (Table E.1b of DeLuchi [1993]). These changes decreased fuel-cycle CO_2-equivalent emissions by about 7 g / mi.

Bulk distribution of LPG

EA Energy Technologies Group (1992, as summarized in Wilcox and Bechtold, [1993]) did a detailed analysis of the distribution infrastructure required for expanded use of LPG for transportation. They analyzed the modes that would carry LPG from production sites to bulk terminals, on the basis of the location of excess LPG production capacity, total LPG shipments, transportation distances, terrain, and the existing LPG infrastructure (EA Energy Technology Group, 1992, p. 5-5). They note that in general pipelines are more economical than trains or barges, and hence estimate that in the year 2010, pipelines will handle 60% of all LPG shipments from production facilities to bulk terminals, trains will handle 34%, and barges 6% (EA Energy Technologies Group, 1992; Wilcox and Bechtold, 1993).

EA Energy Technologies Group (1992) estimated the straight line shipping distance between production and consumption centroids, and the amount of LPG shipped by each mode between the centroids. With these data, one can estimate the gallon-weighted average *straight-line* shipping distances by mode:

Pipeline	Barge	Rail
604	982	178

They assumed that trucks carry LPG from the terminals to the final consumers, a distance of 50 miles (EA Energy Technologies Group, 1992). I base my estimates of tons-shipped / ton-produced, and average shipping distance, in part on the figures derived from EA Energy Technologies Group (1992).

I assume that the energy requirements of transporting LPG are 10% higher than the requirements of transporting ambient liquid fuel, because LPG must be maintained at a few atmospheres pressure.

Truck distribution of LPG is discussed in a separate section.

Truck distribution of methanol, ethanol, LPG, biodiesel, and F-T diesel

DeLuchi (1993) based the length of haul (parameter LH1W in the equation above) and tons-shipped / ton-produced (parameter TS / TP) for methanol, ethanol, and LPG on a qualitative consideration of plant siting with respect to end users, and on estimated ton-miles by trucks carrying petroleum products. However, as explained immediately above, my revised estimate of ton-miles by trucks carrying petroleum products is about one-quarter of the value originally assumed in DeLuchi (1993). This suggests that I implicitly overestimated the average haul by a factor of about four, assuming that virtually all petroleum products are transported by truck at some point. Indeed, I now calculate that the average haul in 1987 was on the order of 40 miles one-way (on the basis of 843 million tons of products supplied in 1987). Therefore, I have greatly reduced all of the assumed one-way haul lengths by truck for methanol, ethanol, and LPG. Consistent with this, EA Energy Technologies Group (1991, 1992) assumed that the average transportation distance from bulk LPG or ethanol plants to service stations will be 50 miles in the year 2010.

Because I assume that the plants that produce F-T diesel from natural gas would be located in the same places as would the plants that produce methanol from natural gas, I assume the same ton/ton and average-length-of-haul parameters for the feedstock transport and fuel distribution phases of the F-T diesel fuelcycle as I assume for the NG-to-methanol fuelcycle. For biodiesel, I assume the same distances as for ethanol from corn.

Distribution of LNG and LH_2

In the model, LNG or LH_2 can be made at the refueling site, at the end of the pipeline, or at a centralized facility and then shipped by truck, ship, or rail to refueling stations. In the latter case, I assume that a minor amount of the liquefied fuel is transported by rail and ship, and that all of it moves by truck. (Data from the Army Corps of Engineers [1995] indicate that 2% of all LPG and LNG was shipped an average 333 miles by domestic water.) The truck shipment distance for LH_2 is longer than the distance for LNG, on the grounds that LH_2 facilities likely will be larger and more centralized than LNG facilities; that is, regional rather than metropolitan. Powars et al. (1994) assumed a 50-mile one distance from a regional LNG plant to service stations.

I assume that cryogenic transport ships, rail cars, and tankers have a slightly higher BTU/ton-mile energy consumption than do their conventional liquid-fuel counterparts (15% higher in the case of LNG, and 20% higher in the case of LH_2) because of the greater mass of the storage container.

Energy consumption of rail, ship, and truck transport

Base-year energy intensity. In Appendix E of DeLuchi (1993), the BTU/ton-mile energy intensity of rail and truck transport were estimated, by commodity (coal, crude oil, or petroleum products) in the year 2000, by multiplying Rose's (1979) estimate of the energy intensity (by commodity) in 1976 by my estimate of the ratio of the year 2000 intensity to the year 1976 intensity. Recently, Vanek and Morlok (1998) have published a similar exercise, by updating another author's estimate of the energy intensity, by commodity, in 1972, on the basis of the change in the overall modal energy intensity (i.e., the change over all commodities) between 1972 and 1993. The following compares their BTU/ton-mile estimates with mine:

	coal		petroleum products	
Year (source)	truck	rail	truck	rail
1972 (in Vanek and Morlok, 1998)	2146	366	1830	630

1976 (Rose, 1979)	2590	450	2270	860
1993 (Vanek and Morlok, 1998)	2691	263	2294	453
2000 (DeLuchi, 1993)	2072	270	1816	516

All of the 1972 estimates cited in Vanek and Morlok (1998) are significantly *lower* than the 1976 estimates in Rose (1979). Furthermore, Vanek and Morlok (1998) estimate about a 30% decrease in the energy intensity of rail transport, and a 25% increase in the energy intensity of truck transport, over the 21-year period 1972 to 1993, whereas I estimated a 40% decrease in rail energy intensity and a 20% decrease in truck energy intensity over the 24-year period 1976 to 2000. It appears that Vanek and Morlok (1998) and DeLuchi (1993) agree only that the energy intensity of rail transport has declined significantly.

Giving partial weight to the work of Vanek and Morlok (1998), I have slightly reduced my estimates of the energy intensity of rail transport in the year 2000, and slightly increased my estimates of the energy intensity of truck transport.

Change in the energy intensity. The EIA's *AEO* (supplemental table 55) projects the following annual changes in the BTU / ton-mile energy consumption of different modes of freight transport (1999 to 2020):

trucks	-0.8% / year
trains	-1.0% / year
ships	-1.2% / year

According to the EIA's *Model Documentation Report* (1994), the EIA projects the fuel economy of trucks as a function of the price of fuel, and of technological improvement over time independent of the price. The BTU / ton-mile energy use of ships is projected on the basis of an analysis of historical trends. The BTU / ton-mile energy use of trains is projected with an exponential decay function (Decision Analysis Corporation, 1994).

An earlier version of the *AEO* projected lower annual percentage changes:

trucks	-0.4% / year
trains	-0.5% / year
ships	-0.5% / year

I assume values closer to the EIA's lower projections of the yearly percentage changes, because the more recent but higher values seem to me to be unsustainable out to 2050. My assumptions for all modes are:

	Feedstocks	Fuels	Materials
Trains	-0.5	-0.5	-0.5
Ships, domestic water	-0.6	-0.2	-0.2
Ships, international water	-0.6	-0.2	-0.2
Pipelines	-0.3	-0.3	-0.3
Trucks	-0.6	-0.6	-0.6

International transport of LNG

The LEM now represents international trade in LNG explicitly. As part of this new representation, parameters for shipment by LNG tanker – BTU/ton-mile, tons/ton, and average miles – have been added to model. I assume thatLNG tankers require 475 BTU/ton-mile, on the basis of a data in GM et al. (2002b) (discussed elsewhere in this report). I assume that the tankers use the LNG fuel itself, including any fuel that otherwise would boil off. I assume that the LNG is used in a gas trubine.

FUEL MARKETING AND DISPENSING

Electricity use at liquid bulk-storage facilities and service stations

The previous version of the model did not include emissions from fuel and electricity use at bulk liquid storage facilities (bulk plants and bulk terminals) or from electricity use at service stations to pump liquid fuels. I have now estimated these emissions and incorporated them into the model. I assume that gasoline, diesel fuel, LPG, methanol, and ethanol will be stored at bulk storage facilities, as gasoline is now. (Compressed and liquefied gaseous fuel is assumed to be delivered directly to stations via pipeline.) I estimate the present emission rate per gram (not gallon; work is related to the mass of the fuel pumped) of gasoline throughput at bulk terminals and plants, and assume that it will apply to all of liquid fuels just mentioned. I also estimate electricity consumption for pumping at gasoline service stations, per gram of gasoline dispensed, and again assume that the rate will apply to all of the liquid fuels just mentioned. (Emissions from compression and liquefaction of gaseous fuels of course already are included in the model.) I do not include emissions associated with energy use for all other functions at service stations (such as heating and lighting), because presumably this energy use will be more or less independent of the type of fuel delivered (although one could argue that the different storage requirements of different fuels will result in different numbers of buildings and different amounts of energy for heating and lighting).

Emissions from storage facilities and service stations are calculated as the product of energy usage per unit of output and emissions per unit of energy usage. Energy usage is calculated from data on expenditures for energy, which are shown in the Table 32 below. Calculated energy usage per unit of output is shown in Table 33, and the emission factors are shown in Table 34. (I assume that the 1987 energy intensities apply to future years.) The table on energy use per unit of activity (Table 33) includes all energy used at service stations -- not just pumping energy -- even though the model includes for service stations only pumping energy, because those are the original data, and because it is interesting to see total energy use in any event. (Also, at a later date I might incorporate total energy use into the model.) In the next paragraph I discuss how I estimate the portion of total service-station electricity use that is for pumping fuel.

As mentioned above, the estimate shown in Table 33, 0.10 kWh / gallon, includes power for lighting and other building functions as well as power to pump gasoline. Presumably, alternative-fuel stations will use the same amount of electricity for lighting and heating and other functions besides pumping as gasoline stations do, so in order to estimate energy use at alternative-fuel stations, I need to separate pumping power use from other power use at gasoline stations. Data from EIA surveys (EIA, *Commercial Buildings Characteristics 1992*, 1994; *Energy End-Use Intensities in Commercial Buildings*, 1994) show that in 1989, mercantile and service buildings, which include gasoline stations, consumed 34,500 BTUs of electricity per square foot of floor space, and $27.5 \cdot 10^6$ BTUs of electricity per employee, for cooling, ventilation, lighting, cooking, office equipment, and refrigeration -- everything *except* things like pumping gasoline. Multiplying these figures by the total square footage (assuming 1500 ft^2 per establishment multiplied by the number of establishments) or the total number of employees in SIC 554 in 1987 (Bureau of the Census, *1987 Census of Retail Trade,* 1991) results in an estimate of 2 to 7 billion kWh of electricity, for everything other than pumping at gasoline stations. SIC 554 actually consumed 10 billion kWh in 1987 for all purposes including pumping fuel. Therefore, if this calculation is valid, 3 to 8 billion kWh of electricity was used to pump gasoline in 1987. On the basis of this, I assume that 0.065 kWh / gallon is used to pump gasoline, out of the total electricity consumption of 0.10 kWh / gallon. (This then is converted to kWh / gram, which as noted above is the basis of the calculations in the model.) This results in an energy efficiency of 99.9% for the pumping stage, which seems reasonable.

Calculated emissions from the use of energy at bulk storage facilities have been added to the stage formerly called “Fuel distribution,” now renamed “Fuel distribution and storage”. Calculated emissions from the use of energy at service stations (for pumping) have been added to the stage formerly called “Compression and liquefaction,” now renamed “Fuel dispensing”. The input energy usage data (corresponding to Tables 3 and 4 of DeLuchi [1991]) are in new rows similarly renamed.

I assume that the energy requirements of dispensing and storing LPG are 10% higher than the requirements for ambient liquid fuels, because the LPG must be maintained at a few atmospheres of pressure.

Upstream evaporative NMOC emissions from gasoline marketing and fuel dispensing

Two changes have been made here. First, “upstream” NMOC emissions from gasoline marketing (excluding emissions from vehicle refueling, but including emissions from refilling storage tanks at service stations) have been reclassified as “fuel distribution” emissions rather than as “vehicular” emissions. Second, the previous emission factor of 4 grams-NMOC / gallon for all years has been replaced with a projection of emissions as a function of the target year, with upper and lower bounds. The double-sided logistic function of Eq. 6 is used, with the following parameter values:

V_L = the minimum value of g / gal-gasoline evaporative emissions from marketing of conventional gasoline, as an asymptote (2.3 g / gal; assumed on the basis of the analysis presented in DeLuchi et al., 1992).

V_U = the maximum value of g / gal-gasoline evaporative emissions from marketing of conventional gasoline, as an asymptote (22 g / gal; assumed on the basis of the analysis presented in DeLuchi et al., 1992, and emissions data in EPA [*National Air Pollutant Emission Trends, 1900-1996, 1997*])

V_{TB} = the g / gal-gasoline emissions in the base year of 1988 (12.34; DeLuchi et al., 1992).

k = shape exponent (the larger the absolute value of k, the more rapidly the limit is approached) (assumed to be -0.10).

T_B = the base year (1988).

Emissions from refueling vehicles, formerly classified as vehicular emissions (because the MOBILE emissions model counts them as vehicular emissions), now are classified as emissions from “fuel dispensing,” and are estimated in g / gal rather than g / mi. (It is more accurate to express these emissions per gallon, because the emissions vary with the quantity of fuel dispensed and the number of refueling times, rather than with miles driven.) The emissions are estimated with double-sided logistic function of Eq. 6 with the following parameter values:

V_L = the minimum value of g / gal-gasoline evaporative emissions from refueling with conventional gasoline, as an asymptote (0.3 g / gal spillage plus 0.4 g / gal from refueling itself, assuming 100% use of onboard refueling controls at 93% efficiency [DeLuchi et al., 1992]).

V_U = the maximum value of g / gal-gasoline evaporative emissions from refueling with conventional gasoline, as an asymptote (4 g / gal; assumed on the basis of the analysis presented in DeLuchi et al., 1992]).

V_{TB} = the g/gal-gasoline emissions for conventional gasoline in the base year of 2000 (2.3; on the basis of the analysis in DeLuchi et al., 1992).
k = shape exponent (the larger the absolute value of k, the more rapidly the limit is approached) (assumed to be -0.10).
T_B = the base year (2000).

These refueling and upstream evaporative losses are relevant to the analysis of GHG emissions in two ways. First, the NMOC emissions themselves contribute to tropospheric ozone formation and hence to global warming. Second, the fuel lost must be made up by increasing throughput (at all stages except refueling), and this entails increased use of process energy and hence increased GHG emissions. However, both of these -- the effect of NMOC emissions on ozone, and the effect of fuel loss on process-energy use -- are relatively minor.

Diesel fuel has very low evaporative emissions: 0.03 g/gal for gasoline marketing, and 0.01 g/gal fuel dispensing.

Upstream evaporative NMOC emissions from marketing and dispensing of reformulated gasoline, methanol, ethanol, LPG, F-T diesel, and biodiesel

These are estimated *relative* to the g/gal upstream evaporative emissions for gasoline or diesel fuel. My estimates are based partly on the relative volatility of the fuel, and partly on emissions data, and are as follows:

reformulated gasoline, relative to conventional gasoline	0.85
LPG, relative to conventional gasoline	0.60
methanol, relative to conventional gasoline	0.60
ethanol, relative to conventional gasoline	0.40
biodiesel, relative to petroleum diesel	0.50
F-T diesel, relative to petroleum diesel	1.00

To obtain g/gal emissions estimates, I multiply these factors by g/gal emissions for gasoline or diesel fuel marketing or dispensing, as documented above.

Note that evaporative emissions of LPG from marketing and fuel dispensing are estimated relative to g/gal evaporative emissions from gasoline, whereas LPG leaks on vehicles are estimated as a percentage of fuel throughput, as are leaks of CNG. In other words, upstream LPG is treated analytically like a liquid; on board the vehicle, LPG is treated analytically like a gas. In regards to the estimate of marketing emissions, Unnasch and Browning (2000) report that LPG marketing produces emissions on the order of 1-2 g/gal, which is slightly less than what I estimate for gasoline. They also report that “current vehicle hose coupling” losses are 7.6 ml for a 12 gallon transfer, which is less than a gram of LPG per gallon. (They state that “dry break” couplings could essentially eliminate this loss.)

Finally, note that the fuel loss rate (expressed as grams / gram or gallon / gallon) is estimated with respect to fuel *output* from the distribution stage, net of losses, not with respect to fuel input to the distribution stage, so that the K factor, discussed elsewhere in this report, is equal to 1+L, where L is the loss rate with respect to output. Thus, if 100 grams of fuel are input to the distribution stage, and 10 grams are then lost, the loss rate is 10 / (100-10) or 10 / 90, not 10 / 100. Of course, in practice, the loss rate is so small that there is no appreciable difference between the output and the input.

Energy required to compress or liquefy gases

The calculation of the energy requirements of compression or liquefaction has been refined. I have estimated new energy-consumption figures (BTUs of compression or liquefaction energy per BTU of fuel compressed or liquefied), have accounted explicitly for gas leakage or gas boil-off and any re-liquefaction of boil-off, and for the number of transfers of liquefied fuel. Formally:

$$BTU_{PE} / \; BTU_{FM} = BTU_{PE} / \; BTU_{FO} \cdot \frac{1 + FLR \cdot FLT_{T_B} \cdot TR \cdot \left(1 + \frac{\Delta FL}{100}\right)^{T-T_B}}{1 - (1 - FLR) \cdot FLT_{T_B} \cdot TR \cdot \left(1 + \frac{\Delta FL}{100}\right)^{T-T_B}}$$

eq. 100

where:

BTU_{PE} / BTU_{FM} = BTUs of compression or liquefaction energy per BTU of fuel delivered to the motorist (Table 35).
BTU_{PE} / BTU_{FO} = BTUs of compression or liquefaction energy per BTU of gas compressed or liquefied (Table 35, and discussed below).
FLR = of fuel boiled off, the fraction that is re-liquefied (Table 35).
FLT_{T_B} = fuel leakage or boil-off, per fuel transfer, in a base year T_B (percentage of the net output delivered to consumers) (Table 35; discussed in the next section).
TR = the number of fuel transfers (e.g., liquefaction plant to truck, truck to refueling station, station to vehicle) (Table 35).
?FL = the annual percentage change in the leakage or boil-off rate (Table 35; discussed in the next section).
T = the target year.
T_B = the base year (assumed here to be 1992).

Note that the term to account for gas lost to the atmosphere is in the denominator, whereas the term to account for gas returned for re-liquefaction is in the numerator. Gas lost to the atmosphere reduces the amount of BTUs actually delivered to the motorist per unit of BTU used by compression or liquefaction, whereas boil-off gases that are re-liquefied increase the energy requirements per unit of gas that finally makes it to consumers.

BTU_{PE}/BTU_{FO}: hydrogen compression. The energy required to compress hydrogen is estimated as a function of the storage pressure on board the vehicle. The user now specifies the hydrogen storage pressure, and the model then calculates the electricity use of the hydrogen compressor with the following simple expression:

$$BTU_{PE-e} / \ BTU_{FO-H_2} = 0.0170505 + 0.0006769 \cdot PSI^{0.5} \qquad \text{eq. 101}$$

where:

BTU_{PE-e} / BTU_{FO-H2} = BTUs of electrical energy consumed by the compressor per BTU of hydrogen produced.

PSI = the storage pressure of hydrogen on board the vehicle (Table 35, and discussed below).

This simple expression is a regression fit to the output of a detailed engineering model of a high-pressure hydrogen refueling station. The expression used here reproduces the output of the detailed hydrogen station model almost perfectly (99.99% accuracy). The calculated BTU_e / BTU_{H2} replaces the assumed value in Table 3 of DeLuchi (1991). (Note that the value shown in Table 3 of DeLuchi [1991], 0.300, is a misprint; it should have been 0.030).

The higher the storage pressure, the greater the energy requirements of and emissions from compression, but the more compact the storage onboard the vehicle. In order to analyze these tradeoffs and find an “optimal” solution, one would need a theoretically complete cost-benefit analysis of hydrogen storage pressure, which would consider the cost of hydrogen storage and hydrogen fuel as a function of pressure, the cost of redesigning the vehicle to accomodate the bulk of the storage system as a function of its bulk (which in turn is a function of pressure), and consumer valuation of storage space, vehicle redesigns, and driving range.

Until recently, most studies of costs and efficiency of hydrogen vehicles (including my own) have assumed, without formal analysis, that the “optimal” storage pressure is between 5,000 to 6,000 psi. Recently, however, Mitlisky et al. (2000) recommend investigating storage pressures as high as 10,000 psi, and already, vessels for this pressure are being designed and tested. Under the name “Hydrogen 700 Project,” leading car manufactuers intend to further promote the technology of storing hydrogen gas in vehicles under pressures up to 700 bar (about 10,000 psi). Weisberg et al. (2002) apparently are developing cost-benefit models of hydrogen storage, and have stated that “recent theoretical results suggest that the best hydrogen containment solutions must store gas at pressures as high as 15,000 psi” (p. 206).

My own current analysis of the lifecycle cost of fuel-cell vehicles as a function of hydrogen storage pressure indicates that 10,000 psi reasonably balances considerations of cost and bulk. Therefore, in this analysis here, I assume a pressure of 10,000 psi. This increases lifecycle emissions by about 3% compared with my previous assumption of storage at 6,000 psi.

Leakage or boil-off of gas related to fuel dispensing.

A small amount of gas leaks from compressors or liquid-fuel storage tanks, seals, refueling lines, and refueling couplings at gaseous fuel stations. These emissions are relevant because they contribute to tropospheric ozone formation and global warming, and must be made up by increased fuel production and throughput upstream.

CNG and CH_2. I assume that there are two sources of gas leakage from gaseous-fuel compression stations: the refueling system and the compressor. (Leakages from the pipeline supplying the station are counted elsewhere in this analysis.)

One can estimate a loss rate from a CNG refueling system on the basis of a crude calculation of the percentage of gas lost when the high-pressure refueling nozzle is disconnected from the vehicle. The interior line and nozzle volume from which the gas can escape probably is on the order of 10 cubic centimeters (see Powars et al., 1994, in regards to LNG). If the density of the gas in the line is 180 g/L (corresponding to 3,600 psi), then at most a few grams “escape” from the line and nozzle volume. The CNG vehicles in my model are estimated to hold about 20 kg of fuel; hence, a typical refueling probably transfers at least 10 kg of gas. Thus, the loss probably is not more than 0.02%.

Estimating losses from the high-pressure compressor is more problematic. The Center for Transportation Research (1998, p. 2) reports “minor leaks” from CNG refueling stations. GM et al. (2002b) report a statement from a manufacturer that a *hydrogen* compressor with worn packings can have losses of 3-5%, but that new compressors have losses “far below this”. They also report a measured value of 0.57% hydrogen loss from a diaphragm compressor. GM et al. (2002b) assume a loss value of 2%, again for hydrogen compressors.

Data on leakage rates from CNG compressor stations on pipelines also are relevant. These suggest a *maximum* leakage rate of about 0.5% for a high-pressure refueling-station compressor.

Given these data and estimates, I assume a total leakage rate (refueling system + compressor) of 0.4% for a CNG station. I assume that leakage rate for CH_2 is related to that for CNG, according to eq. 47, which assumes that at the same pressure, hydrogen leaks 50% more than does CNG (because that it is lighter), and that the leakage rate varies with the square root of the storage pressure.

LNG. Although leakage from a properly functioning LNG station should be relatively small, because LNG dispensers are fully automatic and self-sealing, and have a vapor return line that sends vaporized fuel back to the liquefier or gas pipeline, significant leaks can occur, either on account of bad equipment or bad practices. The U. S. Department of Transportation (U. S. DOT, 1995) visited LNG bus refueling sites, and witnessed significant leakage from all equipment and activities: bulk transfer from truck to bulk storage, bulk storage, bus fueling, and vapor recovery. In one bulk transfer to a hot temporary storage vessel, more than 15% of the LNG was lost. (This 15% presumably is with respect to the initial amount of LNG; if so, the figure with respect to the final amount of LNG, which is what we are interested in here, would be 18%.) At one refueling site, LNG vapors were released when all vehicles where refueled, most released droplets, and several released streams. However, at another site there were no fuel leaks at any point in the operation. Powars et al. (1994) mention an LNG coupling design that has no leakage during operation, and 10 cc of loss upon disconnection.

It is likely that there will be fewer instances of serious leakage as refueling procedures and equipment improve. Indeed, given that zero-loss LH_2 refueling stations have been designed and operated (see discussion below), it seems reasonable to assume that standard practice at LNG stations should be essentially zero loss. I assume an average loss of 2.0% (of the output net of losses) in 1992, declining 5.0% per year in relative terms. At this rate, station losses contribute about 2% to fuel cycle GHG emissions in the year 2015.

$\underline{LH_2}$. Wetzel (1998) provides an excellent description of recent progress in the design and operation of LH_2 refueling stations. Since 1991, researchers at the Solar-Wasserstoff-Bayern facility in Germany have worked to minimize the refueling time and refueling losses for BMW’s liquid-hydrogen car. In the second half of 1996, they tested a system in which the gasified hydrogen in the system is displaced into the vehicle tank and then condensed by spraying super-cooled LH_2 into the tank. With this system, the boil-off LH_2 is re-liquefied; there is no gas return line, and no cryovalve. The fuel-line is disconnected in a “clean break” with no gas leakage. Refueling takes 2.6 minutes, with no loss of LH_2.

I cannot evaluate whether the loss of LH_2 in this system really is zero, or just a small amount, say less than 1%. Also, it is not clear whether there can be more boil-off than can be re-liquefied. I assume 4% boil-off (of the output net of losses) in 1992, declining 7%/year in relative terms.

Table 35 shows all of the foregoing assumptions. These rates do not include any fuel leakage or boil-off from the vehicle itself (that leakage or boil-off is treated as a vehicular loss). Finally, I assume that the composition of gas leaks are the same as the composition of the fuel. This means, for example, that leaks from CNG stations have the composition shown in Table 5 and are not 100% CH_4.. It also means that if hydrogen fuel has trace carbon compounds, the CO_2-equivalent effect of these compounds is estimated.

Boil off of liquefied gases as a result of fuel transfers

Thus far, we have accounted for the following sources of evaporative, leakage, or boil-off emissions:

- all vehicular losses (ordinary "fugitive" evaporative or leakage emissions; losses due to tank failure; boil-off losses; see the section on vehicular emissions).
- all losses due to refueling vehicles (previous subsection)
- evaporative emissions of liquid fuels from fuel marketing (elsewhere in this major section).
- leaks of gas from pipelines (in section on natural gas transmission).

It remains to estimate boil-off losses from the transfer of liquefied fuels from plant to truck, and from truck to refueling station.

Appendix L of DeLuchi (1993) assumes that in the various transfers of liquid hydrogen (plant to truck, truck to station, and station to vehicle), a total of 16% of the fuel is lost or has to be re-liquefied (DeLuchi 1993). Excluding the transfer from station to vehicle, which now is accounted separately, the total loss would be about 10%, or 5% per transfer. Sherif et al. (1997) report that boil-off losses from the storage, transfer, transport, and handling of LH_2 can consume up to 40% of its combustion energy. Zittel (1996) says that total losses from a liquid-hydrogen production and transport system have been reported to be 1-10%.

However, it seems likely that in any extensive use of LH_2 as a vehicle fuel the losses due to fuel transfers (from plant to truck, truck to station, and station to vehicle) will be minimized. As discussed above, Wetzel (1998) describes a recently developed LH_2 refueling station which actually has no LH_2 loss, mainly because super-cooled LH_2 is used to condense any hydrogen that has evaporated in the lines and vehicle tank. (Of course, there still may be some energy cost to keeping the lines and tanks cool, and a limit to the amount of gaseous hydrogen that can be condensed.) I assume that losses from truck transfers are the same as losses from refueling (Table 35) and that half of the "lost" gas is vented, and half is re-liquefied.

In the case of LNG, two scenarios are considered: one in which LNG is liquefied at a central plant, and then trucked to refueling sites; and a second in which LNG is liquefied at the refueling site. In the first scenario, there are three transfers, just as in the in LH_2 scenario. Losses from truck transfers are assumed to be the same as losses from refueling (Table 35). (Of course, in the first scenario, emissions from the trucks themselves are added.)

EMISSION FACTORS FOR INDUSTRIAL BOILERS, OTHER STATONARY SOURCES, AND NON-ROAD ENGINES

Organic compounds

Formerly, organic compounds were referred to as mostly “hydrocarbons” (HCs), and organic compounds excluding methane as “non-methane hydrocarbons” (NMHCs). Now, in keeping with the terminology adopted by the EPA, I refer to total organic compounds (TOCs) and non-methane organic compounds (NMOCs), except of course when I cite data identified specifically as NMHCs.

Organic compounds include aldehyde emissions, except as noted. NMOCs exclude only methane; i.e., they include ethane. They therefore are not the same as volatile organic compounds (VOCs), which exclude ethane as well as methane. I do not use VOCs anywhere in this report.

In the basic emission factors, the following are tracked separately:

- TOCs excluding aldehydes, from the exhaust.
- Total evaporative or leakage emissions of organic compounds.
- Aldehyde emissions, from the exhaust.
- NMOC emissions from exhaust and evaporation or leakage.
- total carbon emissions from exhaust and evaporation or leakage.
- NMOC emissions weighted by their relative ozone-creation potential.

I assume that emissions reported as “TOC” in AP-42 include aldehydes, and so subtract aldehydes from the measure “TOCs excluding aldehydes, from the exhaust”. I assume that emissions reported as “HCs” or “THCs” do not include aldehydes. The EPA’s discussion (Lindhjem, 1997) of the relationship between total hydrocarbons and total organic gases (which I assume are the same as total organic compounds) indicates that these assumptions are reasonable.

PM and SO_2 emissions; black carbon and organic matter component of PM for all sources in the LEM

PM and SO_2 emissions from all combustion sources (vehicles, boilers, trains, ships, etc.), and from some non-combustion sources (e.g., catalytic crackers in petroleum refineries; sulfur removal and recovery units at crude oil and natural-gas processing plants) have been added to model. Most of the PM emissions factors are from EPA’s AP-42; SO_2 emissions are calculated on the basis of the sulfur content. However, several sources of fugitive dust (e.g., coal mining, agricultural operations, and roads) are not yet included.

For the purpose of calculating CO_2-equivalent emissions, the LEM has CEFs for black carbon (BC) aerosols from combustion, organic-matter (OM) aerosol from combustion, and dust (which generally comprises earth-crustal material) (Appendix D). Thus, in order to be able to calculate CO_2-equivalent emissions from combustion sources, we must know the BC and OM fraction of PM emissions.

We estimate BC and OM as a fraction of post-control $PM_{2.5}$ emissions, because BC aerosols larger than 2.5 microns apparently have relatively little effect on climate.

This requires that we know three quantities for each emissions source: $PM_{2.5}$ / PM; BC / $PM_{2.5}$, and OM / $PM_{2.5}$. Table 41 presents our estimates of these quantities for *all* sources in the LEM. For BC and OM content we draw primarily on the recent reviews and analyses by Battye and Boyer (2002) and Bond et al. (2003); for PM size distribution we draw primarily on EPA's AP-42 and their *Air Emissions Species Manual* (Radian, 1990). (Note that Battye and Boyer [2002] and Bond et al. [2003] draw some of their data from AP-42 and an updated version of Radian [1990], so all the estimates in Table 41 may not be independent.)

Aerosols from biomass combustion have constituents other than OM (such as Na^+ and K^+) that tend to cool climate (Jacobson, 2002, 2003). However, the LEM does not have CEFs for these components. To account for the affects of these other constituents, the OM fraction of biomass aerosols is multiplied by an enhancement factor, which is assumed to be 1.35 in the case of aerosols from bio-fuel combustion, and 2.0 for agricultural residue burning.

Control of emissions from trains, ships, boilers, engines, etc.

In the previous version of the model, the user made a direct estimate of the average in-use emission factor for the trains, engines, industrial boilers, etc. This estimate was "direct" inasmuch as it was not built up from separate estimates of the uncontrolled emission rate and the extent and effectiveness of emission controls. Any effects of emission controls were built into, or written in with, the directly input emission factor -- for example, by dividing an uncontrolled emission rate by two.

Now, the model has a set of factors for uncontrolled emissions, and a separate set of population-wide average emission-reduction factors, due to controls, for trains, tankers, scrapers, loaders, off-road trucks, tractors, well equipment, industrial engines, pipeline engines and turbines, industrial boilers, and building heaters. These population-wide average emission-reduction factors are calculated on the basis of the extent and effectiveness of emission controls. The extent of controls, in turn, is estimated on the basis of the extent in some base year, and the rate of increase thereafter. Formally:

$$PER_{S,P,T} = 1 + \min\left[1.0, \max\left[0.0, FWC_{S,P,T_B} + \left(T - T_{S,P,B}\right)\cdot TO_{S,P}\right]\right]\cdot\left(ER_{S,P}\right)$$

eq. 102

where:

$PER_{S,P,T}$ = the population-average emission-reduction factor for emission source S and pollutant P in target-year T (total actual emissions in year T from all sources S, controlled and uncontrolled, divided by uncontrolled emissions).

FWC_{S,P,T_B} = the fraction of total fuel use, by emission source S, that is subject to emission control for pollutant P, in base year T_B (based on the analysis in DeLuchi et al., 1992, and other sources).

T = the target year of the analysis (input by the user).

$T_{S,P,B}$ = the base year for control of pollutant P from emission source S (I assume 1990 for industrial boilers; 1995 for trains and NG compressors; 2000 for all other sources).

$TO_{S,P}$ = the rate of adoption of emission controls for pollutant P from emission source S (fuel throughput newly subject to control in one year divided by total fuel throughput in a year) (my assumptions, based on the analysis in DeLuchi et al. [1992] and other sources).

$ER_{S,P}$ = The emission-reduction factor for controlled emissions of pollutant P from emission source S (controlled emissions from source S, per unit of fuel input or output, divided by uncontrolled emissions from source S, per unit of fuel input or output) (Table 36).

The min and max functions are required to keep the relevant fractions between 0 and 1.0.

Industrial boilers

In the previous version of the model, emissions from industrial boilers (used in a variety of fuel cycles) were estimated as follows:

coal: use the emission factors for utility boilers;

NG: use AP-42 (fourth edition) factors for small industrial boilers; assume HC and CO uncontrolled, NO_X controlled to level estimated by DeLuchi et al. (1992);

refinery gas: use emission factors for NG;

fuel oil: use AP-42 (fourth edition) factors for industrial boilers firing #5 or #6 fuel oil; assume HC and CO uncontrolled, NO_X controlled to level estimated by DeLuchi et al. (1992);

crude oil: use emission factors for fuel oil;

petroleum coke: use factors from AP-42, third edition.

LPG: not included in model.

wood waste: not included in the model.

The revised version features a number of minor changes to these factors:

Coal: The model no longer automatically uses the emission factors for utility boilers. Now, the user must input separate uncontrolled-emission factors and control factors for industrial boilers using coal. Presently, the uncontrolled-emission factors are those for dry-bottom, wall-fired, pulverized-coal boilers, which are used commonly by industry as well as by utilities. Emission factors for PM, PM_{10}, $PM2_{2.5}$, SO_x, and aldehydes have been added. Control factors are discussed elsewhere in this report. Also, to account for emissions from use of limestone to scrub sulfur, I have added to emissions from coal-fired industrial boilers the same limestone-related emissions estimated for coal-fired utility boilers (see Appendix D of DeLuchi [1993]).

NG: The NG factors remain the same: those for small industrial boilers (between 10 and 100 10^6 BTU / hour). Factors for PM, PM_{10}, $PM2_{2.5}$, and SO_x have been added. (Note that the AP-42 5th-edition emission factors are consistent with those recently estimated by Ferry et al. [1997].) Control factors are discussed elsewhere in this report.

Refinery gas: The emission factors now are calculated on the basis of the assumed composition of the refinery gas. In essence, there is a separate set of emission factors for each component of refinery gas (CH_4, LPG, H_2S, and H_2). The factors for each component are weighted by the energy share of the component (so that if methane is 40% of refinery gas on an energy basis, then the methane emission factors get a weight of 0.40), and the weighted factors are summed for all of the constituents to produce a weighted-average emission factor. Each set of emission factors (one set for each of the components, CH_4, LPG, H_2S, and H_2) is estimated as $NG_p \cdot K_{p\text{-}c}$, where NG_p is the emission factor for pollutant P from natural-gas-fired industrial boilers, and $K_{p\text{-}c}$ is emissions of P from component C (say, LPG) relative to emissions of P from natural-gas combustion. Thus, all emission factors are estimated relative to the natural-gas factors. The relative emission factors ($K_{p\text{-}c}$) are shown in Table 37. Sulfur emissions are calculated on the basis of the sulfur content of the gas, due to H_2S.

Fuel oil: The fuel-oil factors remain the same: those for industrial boilers firing #5 or #6 fuel oil. Factors for PM, PM_{10}, $PM2_{2.5}$, SO_x, and aldehydes have been added. Control factors are discussed elsewhere in this report.

Crude oil: Uncontrolled-emission factors (in g / 10^6 BTU) for CH_4, TOCs, CO, and NO_x still are assumed to be the same as those for fuel oil. SO_x emissions are calculated on the basis of the sulfur content. PM, PM_{10}, and $PM_{2.5}$ emissions are calculated on the basis of the sulfur content of the fuel, using the relationships defined for fuel oil (EPA, 1995, AP-42). The control factors are assumed to be the same as those for fuel oil.

Petroleum coke: The fifth edition of AP-42 does not have factors for petroleum coke, so the uncontrolled emission factors from the third edition remain in the model. Emission factors for PM and SO_x have been added. Control factors are discussed elsewhere in this report.

LPG: Emission factors from AP-42, fifth edition, were added. The propane emission factors were weighted by 0.9, and the butane factors by 0.1.

wood-waste: Emission factors from AP-42, fifth edition, 2003 supplement were used: uncontrolled CO, NO_X. and PM emission factors for dry-wood fired boilers; and uncontrolled TOC, CH_4, and N_2O emission factors for wood-residue combustion (Table 18). SO_X calculated from the sulfur content of the fuel. Controls on NO_X and PM assumed to be the same as for coal-fired plants; other pollutants not controlled.

Gasoline and diesel industrial engines and large stationary diesel engines

The post-control emission factors for industrial engines and large stationary diesel engines are equal to uncontrolled-emission factors multiplied by emission-reduction factors, which account for the projected use of emission controls. The uncontrolled-emission factors for NMOC, CH_4, CO, NO_X, PM, PM_{10}, and $PM_{2.5}$ are from the EPA's *Compilation of Air Pollutant Emission Factors* (AP-42, 1995). The emission factors for N_2O are my assumptions. Uncontrolled SO_2 emissions are calculated on the basis of the sulfur content of the fuel. The emission-reduction factors are discussed elsewhere in this report.

I assume that the EPA AP-42 emission factors for gasoline industrial engines (and for gasoline-powered tractors) are based on conventional gasoline (CG). Industrial engines and tractors that use reformulated gasoline (RFG) presumably would have lower emissions of CO, NMOCs, and NO_X. I assume that the ratio of tractor or industrial-engine emissions on RFG to tractor or industrial-engine emissions on CG is equal to this ratio for highway vehicles (shown in Table 12). I then weight the RFG and the CG emission factors by the RFG or CG fraction of the total gasoline energy used by the industrial engine or tractor. The RFG or CG fraction of the total gasoline energy used by the industrial engine or tractor is calculated from the RFG or CG fraction of the total fuel volume, which is specified by the model user.

Emission factors for gas-turbine and gas-engine pipeline compressors

I have input the EPA's (*Compilation of Air Pollutant Emission Factors*, AP-42, fifth edition, 1995) revised factors for uncontrolled emissions of CH_4, CO, and NMOCs from pipeline compressors. The CO, NO_X, and THC emission factors are consistent with those recently estimated by Ferry et al. (1997). The CH_4 emission factors are consistent with those used in the EPA / GRI's (1996) comprehensive analysis of methane emissions from the natural gas system. Control factors are discussed elsewhere in this report[77].

I assume that the AP-42 emission factors include any NMOC, CO, NO_x, SO_x, and PM emissions from the combustion of lubricating oil used in natural-gas-fired internal-combustion engines. In the case of hydrogen-fueled internal-combustion engines, I assume minor emissions to account for the combustion of lubricating oil:

- NMOC: 0.2 g / 10^6-BTU
- CO: 1.0 g / 10^6-BTU
- CH_4: 0.02 g / 10^6-BTU
- SO_x: 0.05 g / 10^6-BTU
- PM: 0.12 g / 10^6-BTU

These assumptions increase lifecycle CO_2-equivalent emissions by on the order of 1%.

Trains

The previous emission factors for trains (Table A.1 of DeLuchi [1993]) were from a table dated 1973 in the EPA's emission-factor handbook, AP-42 Volume 2, "Mobile Sources". In the early 1990s the EPA updated the emission factors for trains, as part of a general update of non-road emission factors required by the 1990 Clean Air Act Amendments. The updated emission factors are reported in Appendix F of the EIA's *Model Documentation Report* (1994).

Ships

EPA's emission factor handbook, AP-42, provides somewhat dated estimates of emission factors for a variety of marine vessels operating under a variety of conditions. The EIA's *Model Documentation Report* (1994) used the EPA factors to estimate a weighted-average emission factor for river, lake, and ocean-going vessels. These factors, which are similar to EPA's AP-42 factors for coastal vessels specifically, are shown in the table below.

Recently, Energy and Environmental Analysis Inc. (EEA), under contract to EPA, reviewed and analyzed available data on emissions from marine vessels (EEA, 2000).

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[77] I assume that more engines than turbines are controlled because uncontrolled NO_X emissions from turbines are nearly 10 times lower than uncontrolled emissions from engines to begin with.

They found that emissions of all pollutants except SOx were a nonlinear function of the load of the engine expressed as a fraction of the rated capacity of the engine. This function was valid regardless of the size and type of the engine. The following table shows the estimated emission factors at three fractional loads, along with the EIA's (1994) estimates and my assumptions (g-pollutant/g-fuel):

	This study	EIA (1994)	EEA (2000)		
			0.2	0.5	0.8
HC	0.0030	0.0065	0.0027	0.0008	0.0004
CO	0.0120	0.0136	0.0152	0.0072	0.0047
NOx	0.0460	0.0346	0.0401	0.0457	0.0475
PM	0.0020	0.0025	0.0012	0.0012	0.0012

The EIA (1994) estimates of CO and NOx are consistent with the EEA (2000) estimates, but the EIA (1994) estimates of HC and PM are higher than the EEA (2000) estimates. The EIA estimates also are broadly consistent with estimates for diesel engines in other, non-marine applications. I choose the EEA (2000) factors at 50% load, with higher values for HC and PM. (Note that in the LEM, the emission factors are input as g/10^6-BTU, not g/g.)

Leaks of gaseous fuels

The fuel cycle energy use and emissions model accounts in details for gaseous fuel leaks from the production, processing, transmission, and distribution of gaseous fuels such as natural gas. However, there also may be leaks of gaseous fuels from the end use of the fuel, such as in vehicles or heaters. Elsewhere, I discuss my assumptions regarding fugitive leaks of gaseous fuels from vehicle refueling stations, and from vehicles themselves. Here, I note that I assume that all other devices or processes that use natural gas, refinery gas, or LPG have a fugitive fuel loss rate of 0.05%. This has a negligible effect (less than 0.05%) on fuel cycle emissions.

Indirect energy use

Appendix E of DeLuchi (1993) cites Rose's (1979) citation of estimates of the ratio of "indirect" to "direct" energy for trains, ships, trucks, and pipelines, where indirect energy is that required to manufacture, repair, and service the mode, and direct energy is that consumed directly by the mode. I use these estimates to calculate "indirect" GHG emissions related to the use of trains, ships, trucks, and pipelines.

In the revised model, I have added indirect GHG emissions related to agricultural machinery and heavy off-road mobile equipment. Fluck's (1985) detailed analysis provides data that can be used to calculate the indirect/direct energy ratio for agricultural machinery. According to Fluck (1985), agricultural machines used 1.149 EJ directly in 1978, and consumed 0.362 EJ per year in manufacture, and 0.200 EJ per year for maintenance and repair. This indicates an indirect / direct ratio of (0.362+0.200) / 1.149 = 0.489, quite comparable to Rose's (1979) estimate of 0.429 for trucks, which seems reasonable. Similarly, Jensen and Hauggaard-Nielsen (2003) estimate that the energy embodied in farm machinery is 40-50% of the "direct" energy used for establishment, harvest, transport, drying, and other activities.. I assume a ratio of 0.45 for scrapers, wheeled loaders, and off-road trucks, and 0.49 for tractors.

The indirect / direct ratio for trains and ships has been reduced, because, with a simple calculation, I am unable to get within an order of magnitude of Rose's (1979) estimates (1.1 for trains, 0.9 for ships). Data from the EIA's *MCES 1991* 1994) and the Census' *1991 Annual Survey of Manufacturers* (1992) indicate that in 1991, the manufacture of railroad equipment consumed at most 6 trillion BTU of primary energy. (Data for 1986 indicate the same order of magnitude.) The transport of railroad equipment consumed on the order of 0.6 trillion BTU in 1993 (1.15 billion ton-miles [Bureau of the Census, *1993 CFS*, 1996] multiplied by my assumed average of 500 BTU / ton-mile). Assuming that maintenance, repair, servicing, and terminal operations consumed roughly as much as did manufacture and transport (to a first approximation, this appears to be true for motor vehicles and farm equipment), the grand total indirect energy consumption was 13 trillion BTU. In 1991, freight rail consumed 410 trillion BTU of energy directly (Davis and McFarlin, 1996; consumption averaged about 440 trillion BTU from 1982 to 1994). This implies an indirect / direct energy ratio of about 0.03! An analogous calculation for ship transport gives a similar result.

There are three likely explanations of the discrepancy between my estimates, which are less than 0.05, and Rose's estimates, which are around 1.0:

- Rose's (1979) source overestimates indirect energy;
- my accounting of indirect energy is incomplete;
- I have underestimated maintenance, repair, servicing, and terminal-operation energy.

I believe that all are true, and so have assumed that the true ratio is of the order of magnitude between my estimates and Rose's: about 0.20. This results in a 1-2% decrease in total fuel cycle emissions for gasoline.

GHG emissions are calculated from these indirect / direct energy ratios in the manner outlined in Appendix E of DeLuchi (1993). The addition of indirect emissions from the use of agricultural machinery increases fuel cycle emissions from the biomass pathways by a nontrivial amount: for example, by about 2% in the corn-ethanol cycle.

Note that I assume that the "direct" energy in the indirect / direct energy ratio includes any direct energy that is used as part of the "indirect" activities: for example, diesel fuel used by trucks used to transport trucks from plant to dealer.

Other

I corrected minor key-in errors for the emission factors for gasoline tractors (Table A.1 of DeLuchi [1993]).

In the previous version of the model, natural gas used in the recovery stage for any feedstock was assumed to be used in industrial boilers. Now, natural gas in feedstock recovery is assigned to natural-gas engines, rather than natural-gas boilers. (According to the EPA / GRI [1996] study, all NG used in NG recovery is used in compressor engines.) This results in an increase in emissions from all fuel cycles, because engines emit more CH_4 than do boilers. Similarly, I have switched the use of natural gas at NGL plants from boilers to compressor engines and turbines, in the proportion indicated by EPA / GRI (1996).

Emissions from hydrogen pipeline compressor turbines and engines have been added. (Originally, the estimate of emissions from hydrogen transmission referred to the emission factors for hydrogen power plants.)

EMISSION AND ENERGY-USE PARAMETERS FOR NONROAD ENGINES

The LEM includes and energy-use of and emissions from trains, ships, tractors and other nonroad engines as part of the lifecycle of transportation fuels and modes. It also represents lifecycle emissions from forklifts as an end-use.

Because emissions from non-road mobile sources, such as trains, depend greatly on the degree of emission control, which in turn depends on Federal and state emission standards, which change over time, I first review the regulation and control of non-road engines. Then, I review and analyze EPA data on energy use and emission factors for forklifts, and present my own assumptions (for forklifts) in this analysis.

Regulation of non-road engines

The 1990 amendments to the Clean Air Act directed the U. S. Environmental Protection Agency (EPA) to study the contribution of nonroad engines to urban air pollution, and, if warranted, regulate them (EPA, 1999a; Federal Register, 1999). Nonroad engines include those in forklifts, farm equipment, off-road construction equipment, recreational equipment, lawn and garden equipment, outdoor power equipment, and marine vessels (EPA, 1999a). (Locomotives and aircraft are treated separately in the 1990 Clean Air Act.) Up until the mid 1990s, no nonroad sources except aircraft were regulated (EPA, 1999a).

In 1991, EPA completed its *Nonroad Engine and Vehicle Emission Study* (EPA, 1991), which showed that nonroad engines are a significant source of nitrogen oxides (NO_X), volatile organic compounds (VOCs), and particulate matter (PM) (see footnote 2 in this report.) As a result of this study, EPA began the process of regulating nonroad engines. Today, the EPA regulates several categories of nonroad engines (EPA, 1999a; Federal Register, 2002; 2003):

- land-based non-road diesel engines (farm equipment, such as tractors; construction equipment, such as backhoes and bulldozers; material handling equipment, such as heavy forklifts; and utility equipment, such as pumps and generators).

- small land-based spark-ignition engines (less than 19 kW) (lawn and garden equipment, such as blowers, lawn mowers, chainsaws, and small tractors)
- large land-based spark-ignition engines (more than 19 kW) (forklifts, airport ground-service equipment, generators, and compressors).
- marine engines.
- recreational engines.
- locomotives.
- aircraft.

The Federal regulatory status for these engines is as follows:

Land-based non-road diesel engines (excluding locomotives)	In 1994, EPA adopted "Tier 1" standards for engines over 50 hp, to be phased in from 1996 to 2000. In 1998, EPA issued more stringent "Tier 2" standards for all engine sizes from 2001 to 2006, and yet more stringent "Tier 3" standards for engines rated over 50 hp from 2006 to 2008 (EPA, 1999a, 1998a; Federal Register, 1998). In 2003, EPA proposed "Tier 4" emission standards, to take effect in 2008 and beyond. The Tier 4 program also proposes large reductions in te sulfur content of diesel fuel for offroad use. See Table 38 for details.
Small land-based spark-ignition engines	Under Phase I of EPA regulations, new small SI engines must comply with standards for HC, CO, and NO_X beginning in 1997. Phase II standards are being developed (EPA, 1999a).
Large-land-based spark-ignition engines	In February 1999, EPA issued a Notice of Proposed Finding that these engines contribute significantly to air pollution (Federal Register, 1999; EPA, 1999b). This was the first step in the process of setting emission standards for these engines. EPA proposed standards in September 2001 and adopted final rules in November (2002) (Federal Register, 2002). California adopted standards in 1998 (Stout, 1999b).
Marine engines	Some marine engines are regulated, some are proposed to be regulated, and some are unregulated; see EPA (1999a, 1997a) for an overview. Recreational marine diesel engines are covered under the November 2002 rulemaking (Federal Register, 2002).

Locomotives	Three sets of emission standards: Tier 0 applies to engines manufactured between 1973 and 2001 , Tier 1 to 2 engines manufactured between 2002 to 2004 engines, and Tier 2 to engines made after 2005 (EPA, 1999a, 1997a).
Aircraft	Emission standards for gas turbine engines have been in place for about 20 years. In April 1997, EPA adopted the standards of the International Civil Aviation Organization (EPA, 1999a, 1997a).

As noted above, EPA recently has regulated emissions from gasoline, LPG, and NG spark-ignition (SI) nonroad engines. After California adopted requirements for large SI engines in October 1998, EPA made "extensive effort to coordinate its anticipated program" with the California program (Stout, 1999b). California's standards for large SI non-road engines, using any fuel, including LPG and NG, are as follows (Stout, 1999b):

	NO_X + NMHC	CO	PM	Evaporative
Limit (g/bhp-hr)	3.0	37.0	none	none
Rationale	capability of available control technologies (i.e., 3-way catalysts)	capability of available control technologies (i.e., 3-way catalysts)	SI engines with 3-way catalysts have inherently low PM	many of these engines use propane, which has low evaporative emissions

The emission limit for NO_X + NMHC represents about a 75% reduction in combined emissions (Stout, 1999b).

The California standards apply over the useful life of the engines, which CARB set at 5,000 hours. EPA adopted the California standards for model years 2004 to 2006, and more stringent standards for subsequent model years (Federal Register, 2002) (g/bhp-hr):

	NO_X + NMHC	CO
2004-2006	3.0	37.0
2007 +	2.0	3.3

For model years 2007 and later, manufacturers have the option of certifying to lower NOx + NMHC standards and higher CO standards, down to 0.6 for NOx + NMHC with 15.4 CO. EPA also adopted a diurnal evaporative emissions standards of

0.2 g / gallon-tank-capacity / day and set other requirements relating to fuel, fuel tanks, and fuel lines, effective in 2007 (Federal Register, 2002,. p. 68294, 68350).

Testing and control of nonroad engines

Test cycle. The current nonroad diesel test cycle consists of a limited combination of steady-state speeds and loads (Federal Register, 1998). However, EPA has been concerned that this test cycle does not include some of the operating modes that are commonly experienced in the field (Federal Register, 1998). Consequently, EPA announced in its 1998 rulemaking that it intended to develop a transient test cycle to supplement the steady-state test (Federal Register, 1998). Subsequently, EPA developed a Nonroad Transient Composite test cycle (NRTC), which it now proposes to use, along with a cold-start test, in future testing of nonroad diesel engines (Federal Register, 2003).

EPA has had similar concerns regarding the testing of large SI nonroad engines (Stout, 1999b). For model years 2004 to 2006, EPA adopted the steady-state duty cycles used by CARB (Federal Register, 2002). However, starting with 2007, EPA specified an expanded set of duty cycles, consisting of a warm-up segment (beginning with a cold start), a transient segment, and a steady-state segment (Federal Register, 2002). Furthermore, to address concerns that even the expanded test cycle does not cover some operating conditions experienced in the field, EPA adopted in-use “field testing” standards beginning with model year 2007 (Federal Register, 2002).

Control of nonroad SI engines. EPA asserts that the engines in nonroad equipment generally are larger than 1 liter and 19 kW, and typically are similar to automotive base engines (Stout, 1999b). As a result, EPA believes that they should be capable of using advanced emission control technologies similar to those used by automobiles:

> Many of the engines that would be affected by these new emission standards have counterpart engine models used in highway applications. While highway engines have seen extensive technological developments, the nonroad engine designs have changed little to reflect these improvements. Shifting toward these technologies that have been developed for cars and trucks, such as electronically controlled closed-loop injection systems with three-way catalytic converters, there is a great potential to dramatically improve engine performance and fuel economy in addition to the anticipated emission reductions (EPA, 1999b, p. 2).
>
> Manufacturers can upgrade engines from an open-loop fuel system to one with electronically controlled closed-loop operation...Gasoline-fueled engines can utilize established fuel injection technology, while LPG- and natural gas-fueled engines can likely achieve a comparable level of emission control with closed-loop carburetor-type fuel systems or new gaseous fuel injection systems. Injection systems for gaseous-fueled engines are becoming available, but have not proven themselves to the same degree as injection systems for gasoline-fueled engines (Stout, 1999b, p. 5).

Similarly, in its recent Notice of Proposed Finding in the Federal Register (1999), EPA believes that “manufacturers will generally be able to produce engine models with the projected control technologies that can be used in most applications in a category without significant modification" (p. 6011).

As regards LPG and NG, EPA notes:

> There is considerable variation in the quality of LPG and natural gas in the field, with a corresponding variation in the emissions from these engines...On the other hand, closed loop fueling technology has the potential to eliminate most of the sensitivity to varying fuel composition by making internal adjustments to ensure consistent air-fuel ratios. We will need to investigate the range of in-use fuel quality for LPG and natural gas to be able to specify fuel properties appropriate for certification fuel and the effect of different fuels on emission levels from closed-loop systems (Stout, 1999b, p. 8).

Emission factors for nonroad engines

The EPA has developed a national nonroad emissions model, "NONROAD"[78]. This model predicts emissions of CO, CO_2, SO_x, PM (TSP, PM_{10}, and $PM_{2.5}$), HC (total hydrocarbons [THC], total organic gases [TOG], non-methane hydrocarbons [NMHC], non-methane organic gases [NMOG], and volatile organic compounds [VOC]), and NO_x, by various levels of aggregation (by county, by type of equipment, by source category code, and so on) for all nonroad equipment categories except locomotives and aircraft (Pollack and Lindhjem, 1997). The model includes five general fuel categories: diesel (2-stroke and 4-stroke combined), gasoline 2-stroke, gasoline 4-stroke, CNG (2-stroke and 4-stroke combined), and LPG (2-stroke and 4-stroke combined) (Lindhjem, 1998).

For each specific type of nonroad equipment, NONROAD calculates emissions in a target year as the product of several factors (Stout, 1999a):

- equipment population in the target year
- the average load factor, expressed as a fraction of the available power
- the rated engine power
- operating hours per year for each unit
- in-use emission factors, accounting for emissions deterioration, and / or new standards

The equipment population is calculated by applying growth and scrappage rates to population estimates in a base year. The emission factors are based on the EPA's NEVES (EPA, 1991) and other studies done since 1991, as discussed below.

Because the fuel cycle analysis presented here estimates emissions per unit of work or fuel energy, and does not estimate an emissions inventory, we do not need the "activity" data inputs of NONROAD (equipment population, load factor, and activity hours). Thus, I extract from the NONROAD documentation the latest EPA emission

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[78] A draft of NONROAD was released in June 1998, and a revised version was released in 2002.

factors, in g/bhp-hr. I divide the g/bhp-hr factors by the brake-specific fuel consumption and then multiply by the higher heating value of the fuel, to convert the emission factors in g/10^6-BTU-fuel, which is the form used in the LEM:

$$EF_{NR,F,P,T} = \frac{EF^*_{NR,F,P,T}}{BSFC_{NR,F,T} \cdot HHV_F} \qquad \text{eq. 103}$$

where:

$EF_{NR,F,P,T}$ = the emission factor for pollutant P from nonroad engine NR using fuel F in year T (g/10^6 BTU).

$EF^*_{NR,F,P,T}$ = the brake-specific emission factor for pollutant P from nonroad engine NR using fuel F in year T (g/bhp-hr) (see the discussion in the text).

$BSFC_{NR,F,T}$ = the brake-specific fuel consumption of nonroad engine NR using fuel F in year T (lb/bhp-hr) (see the discussion in the text).

HHV_F = the higher heating value of fuel F (10^6-BTU/lb) (calculated from heating value data presented in this report).

Diesel engines. EPA's emission factors for compression-ignition onroad diesel engines are documented in EPA (2002b, 1998b, 1991), Beardsley and Lindhjem (1998b), and Pollack and Lindhjem (1997). My analysis here is based mainly on the data in EPA's NEVES (EPA, 1991) and on previous versions of the NONROAD model (Beardsley and Lindhjem, 1998b). For the most part, I have not incorporated revisions made with the most recent (2002) version of NONROAD (EPA, 2002b).

For nonroad diesels, EPA estimates emission factors by model year and engine size, for steady-state operation, and then estimates "in-use adjustment" factors meant to account for higher emissions during in-use transient operation. In the case of HC, CO, and NOx, their estimated emission factors take account of the recent Tier 1, 2, and 3 -- but not Tier 4 -- emission standards, shown in Table 38. In the 1998 version of NONROAD, EPA did not estimate deterioration factors for nonroad diesel engines. The factors used in NONROAD are equal to the steady-state emission factors multiplied by the in-use adjustment factors.

For nonroad diesel engines, I use the EPA's 1998 NONROAD emission factors for HC, CO, and NO_X. However, because the NONROAD model does not account for the impact of the new PM standards (Beardsley and Lindhjem, 1998b), I make my own estimates for PM and Tier 4-level emissions, as noted in Table 39. Also, I split the EPA's factor for total HC into a CH_4 factor and a NMHC factor. Finally, I add a factor for N_2O, because NONROAD does not report N_2O. My assumptions are shown in Table 39. As did EPA in prior versions of NONROAD, I assume that there is no deterioration in emissions from nonroad diesel engines.

Spark-ignition (SI) engines. HC, CO, NOx, and PM emission factors for nonroad SI engines are documented in EPA (1991, 2002c, 2002d, 2002e, 2002f, 2002g, 2002h), Beardsley and Lindhjem (1998a), Stout (1999a, 1999b), Pollack and Lindhjem (1997), and Harvey (1998). In general there are two major sources of estimates of these emission factors: the NEVES (EPA, 1991), and the current version of NONROAD (EPA, 2002g, 2002h), which relies in part on the NEVES.

NEVES estimates of exhaust emissions. For nonroad SI engines, EPA's NEVES estimated HC, CO, NOx, and PM exhaust emission factors by model year and engine size, for steady-state operation. The NEVES study used heavy-duty engine data to adjust emissions from SI engines for what it called "in-use effects (e.g., EPA, 1991, Table 2.07). (The previous version of NONROAD did not use such adjustment factors, because EPA believed that there was not a significant difference between steady-state emissions and transient emissions from SI engines [Beardsley and Lindhjem, 1998a], but, as indicated below, the current version of NONROAD does have a steady-state/transient adjustment factor [EPA, 2002h].)

The NEVES provides emission factors for each specific kind of equipment, such as 4-stroke spark-ignition (4SSI) propane-powered forklifts (EPA, 1991, Table 2.07). According to Beardsley and Lindhjem (1998a), the LPG and CNG emission factors in the NEVES were estimated by multiplying the factors for gasoline by some relative emissions factor. NEVES estimates the following emission factors, specifically for 4SSI engines in forklifts (EPA, 1991, Table 2.07) (g/bhp-hr):

Fuel	HC	CO	NO_X	PM
Propane	4.50	82.81	17.90	0.05
Gasoline	10.02	258.70	5.16	0.06

The differences between propane and gasoline shown here generally are consistent with the differences found with highway vehicles. I assume that these factors apply to uncontrolled engines not subject to emission standards.

NONROAD estimates of exhaust emissions. The current version of NONROAD estimates exhaust emissions factors for the general category of large, 4SSI gasoline, LPG, and CNG engines; it does not estimate factors for specific pieces of large equipment. The model accounts for Phase I and Phase II regulations for small SI engines, but does not fully account for recent new regulations for large SI nonroad engines (see the discussion of regulations, above) (EPA, 2002h).

NONROAD estimates an in-use emission factor as the product of a zero-mile steady-state emission rate, an adjustment for the difference between transient and steady-state emissions, and an emissions deterioration factor. The uncontrolled, zero-mile, steady-state emission factors for HC, CO, and NOX are based on a recent compilation of tests on large 4SSI engines (Stout, 1999a; EPA, 2002h); the PM emission factors are taken from the NEVES. (Beardsley and Lindhjem, 1998a, indicate that the study that produced the HC, CO, and NOX results did not measure PM emissions.) The factors are as follows (g/bhp-hr):

Fuel type	HC	CO	NO_X	PM
gasoline	6.22	203.4	7.130	0.06
LPG	1.68	28.23	11.99	0.05
CNG	24.64	28.23	11.99	0.05

EPA (2002h) then multiplies these by the following factors to account for the difference between transient operation and steady-state operation:

Fuel type	HC	CO	NO_X	PM	BSFC
all	1.30	1.45	1.0	1.0	1.0

Finally, NOROAD (EPA, 2002g) accounts for emissions deterioration over the life of the engine. In the 2002 version of NONROAD, EPA (2002g) estimates the following ratio of emissions at median life to emissions at the beginning of life, for all large 4SSI nonroad gasoline engines:

MY	HC	CO	NO_X	PM
pre-2004	1.26	1.35	1.03	1.26
2004-2006	1.64	1.36	1.15	1.64
2007+	1.64	1.36	1.15	1.64

In this, the EPA (2002g) *assumed* that the factors for PM were the same as for HC, that the factors for post 2007 model years were the same as those for 2004 to 2006, and that factors for CNG and LPG were the same as for gasoline.

The above factors result in the following estimates of in-use emissions at median life for uncontrolled 4SSI engines (g/bhp-hr)

fuel type	HC	CO	NO_X	PM
gasoline	10.19	398.16	7.34	0.08
LPG	2.75	55.26	12.35	0.06
CNG	40.36	55.26	12.35	0.06

The NONROAD generic emission factors can be compared with the emission factors in the NEVES, specifically for 4-stroke SI engines in forklifts (shown above). The NEVES emission factors differ somewhat from the current NONROAD emission factors.

EPA does not estimate N_2O or CH_4 emissions from nonroad engines. My estimates of these are discussed in the notes to Table 40.

Emission factors for post-2004 model years. The new emission standards (see the discussion above) for large SI gasoline nonroad engines are considerably lower than the emission factors estimated in NONROAD or NEVES. I assume that the emissions from model year 2004 and later SI nonroad gasoline engines are about equal to the proposed standards~~.~~

My assumptions and estimates are shown in Table 40.

Estimates of evaporative emissions. EPA (2002c, 2002d, 2002f) discusses seven sources of evaporative emissions from non-road gasoline spark-ignition vehicles: diurnal, hot soak, running loss, resting loss, crankcase, refueling-spillage, and refueling-vapor-displacement.

- Diurnal: The EPA's NEVES study assumed 3.0 g / day / gallon-tank-capacity for large engines, and 1.0 for small engines (EPA, 1991, 2002d). However, the equation that EPA uses in MOBILE6 to predict evaporative emissions from highway vehicles suggests that 1.0 g / day / gallon-tank is reasonable for larger non-road engines, too (EPA, 2002d). As discussed above, EPA has set a standard of 0.2 g / gallon / day beginning with the year 2007.
- Hot-soak, resting, and running losses: the NEVES and the current version of NONROAD assume zero emissions (EPA, 2002d). Harvey of EPA (1998) says that hot soak emissions probably are only about 1% of total HC emissions from gasoline nonroad engines, but EPA (2002d) cites a study that indicates that hot-soak emissions might not be negligible.
- Crankcase and refueling: The NEVES study assumes 2.69 g / bhp-hr crankcase+refueling emissions from gasoline-powered forklifts (EPA, 1991). The most recent version of NONROAD (year 2002) revises the methods in NEVES and estimates spillage and vapor displacement emissions from refueling as a function of tank volume (in the case of spillage) and temperature (in the case of vapor displacement) (EPA, 2002c, 2002f). However, because no new data were used in these revisions, it is likely that the resulting emission factors are not significantly different from those in the NEVES.

Fuel. California requires that nonroad engines use the same gasoline that highway vehicles use (Stout,1999b). Therefore, I assume that forklifts subject to the eventual Federal emission controls for large SI engines will use reformulated gasoline. I assume that pre-control engines use conventional gasoline.

My assumptions are shown in Table 40.

FUELCYCLE EMISSIONS FROM THE USE OF NATURAL GAS, ELECTRICITY, FUEL OIL, AND LPG FOR HEATING AND COOKING

Background

In 1993, U. S. households consumed 7.15 quads of energy for space heating and for water heating, broken down as follows (EIA, *Household Energy Consumption and Expenditures 1993*, 1995):

	NG	Electricity	Fuel oil	LPG	Total
Space heating	3.67	0.41	0.95	0.30	5.33
Water heating	1.31	0.34	0.12	0.05	1.82
Total	4.98	0.75	1.07	0.35	7.15

Commercial buildings consumed 1/5 to 1/3 as much as households:

	NG	Electricity	Fuel oil	District heat	Total
Space heating	1.09	0.11	not reported	not reported	1.70
Water heating	0.52	0.05	not reported	not reported	0.81
Cooking	0.20	0.02	not reported	not reported	0.22
Total	1.81	0.18	0.24	0.53	2.73

Cooking probably consumed an additional 0.5 quads. The grand total of about 10 quads -- or about 12 quads of primary energy -- is about 13% of total U. S. energy consumption.

Because heating can be provided by at least 4 different sources of energy -- natural gas, electricity, fuel oil, and LPG -- and is a major source of U. S. energy consumption, it is interesting to compare fuel cycle CO_2-equivalent emissions resulting from the use of different fuels.

Applying the model to estimate fuel cycle emissions for space heating and water heating

For this analysis, I constructed separate fuel cycles for heating end uses. I consider LPG (from natural gas, crude oil, and a combination of both), NG from natural gas, fuel oil from crude oil, and electricity from several sources. I assume that LPG is 95% propane and 5% butane.

For each heating fuel, the upstream portion of the fuel cycle (from feedstock recovery through fuel distribution) is the same as the upstream portion of the corresponding transportation fuel cycle, except for a few clear differences in fuel distribution and dispensing. For example, the upstream NG-to-heating fuel cycle is the same as the upstream NG-to-CNG fuel cycle, except that in the former there is no final high-pressure compression stage. Generally, I assume that the distribution of LPG or fuel oil to residential or commercial users is the same as the distribution of LPG or diesel fuel to motor-vehicle service stations. I also assume that the refinery processes that produce No. 2 distillate fuel oil for heating are the same as those that produce No. 2 distillate diesel fuel for highway trucks.

End-use emission factors for residential and commercial heating

The EPA emission-factor handbook, AP-42, contains some emission factors for residential furnaces burning natural gas or fuel oil. It does not contain any emission factors specifically for residential uses of LPG, but it does contain emission factors for LPG commercial and institutional boilers. Table 34 shows the pertinent emission factors from AP-42.

The EPA publishes another emission-factor sourcebook, the *AIRS Facility Subsystem Source Classification Codes and Emission Factor Listing for Criteria Air Pollutants* (EPA, 1990), based largely although not entirely on AP-42. This sourcebook does report emission factors for commercial and institutional "external combustion boilers -- space heaters," for NG, LPG, and fuel oil. Table 34 here also shows the emission factors from this sourcebook. EPA notes that "these factors, for the most part, are taken directly from AP-42. In certain cases, however, they may be (1) derived from information not yet incorporated into AP-42 or (2) based merely on the similarity of one process to another for which emissions information **does** exist" (EPA, 1990, p. 4, emphasis in original). Given that the fifth edition of AP-42 does not report emission factors for LPG used in residential furnaces, it is not clear whether the emission factors for LPG space heaters, reported in the 1990 EPA report, are based on old AP-42 factors for LPG used in commercial boilers, or rather actually are based on the EPA's judgment regarding emission factors for residential furnaces specifically.

Generally, the emission factors from EPA's (1990) source handbook are consistent with the emission factors from AP-42 (EPA, 1995). With one exception, I use the NMOC, CO, NO_X, and PM emission factors in the 1990 sourcebook (EPA, 1990), because they are specific to the use of LPG, NG and fuel oil for space heating. (Also, it is not clear if the AP-42 data specifically for residential furnaces is in fact more recent than the data in the 1990 EPA report. Much of the source material in AP-42 is old.) The exception is with regards to PM. As noted in Table 34, the most recent supplement to AP-42 (EPA, 1995) states that new residential burners emit much less PM than old ones, and reports a PM emission factor for fuel oil that is much lower than the one in the 1990 EPA report. My estimate of PM emissions is based on the AP-42 emission factor for fuel oil, assuming first that total PM emissions (filterable plus condensable) from fuel oil combustion are a bit higher than the reported filterable emissions, and then that PM emissions from NG and LPG combustion are less than PM emissions from fuel oil combustion, per 10^6 BTU of fuel burned.

To estimate CH_4, the EPA's 1990 estimates of NMOCs was multiplied by the CH_4/NMOC ratio implied by the AP-42 estimates.

The emission factors for N_2O are more problematic. The AP-42 emission factors result in LPG having 20 times the N_2O emissions of fuel oil, per 10^6 BTU burned. Data reviewed in Delucchi and Lipman (1997) indicate that combustion of natural gas and fuel oil emits 0.2 to 2.0 g/10^6 BTU. These data also indicate that in the few cases where emissions from both fuels were measured in the same project, there was no systematic difference between NG and fuel oil. Moreover, there is no reason to assume that N_2O emissions from LPG are dramatically different from N_2O emissions from NG. With these considerations, it seems most reasonable to assume a value of 1.0 g-N_2O/10^6-BTU-fuel for all fuels. (This value makes a small allowance for the possibility of secondary N_2O emissions.)

All assumptions are shown in Table 34. The emission factors are assumed to apply to water heating, as well as to space heating.

End-use efficiency

Emissions of criteria pollutants and GHGs should be estimated per unit of service provided, so that fuels and technologies can be compared holding at least the major "benefit" -- the service provided -- constant. In the case of transportation, the service is miles of travel, and hence the ultimate emission measure of interest is grams emitted per mile of travel. In the case of space heating and water heating, the service is useful heat: heat transferred from the heater to the air or surface.

Our final result, then, will be grams of pollutant (or CO_2-equivalents) per BTU of useful heat provided. This final result is calculated by dividing an intermediate result, grams per BTU of fuel or electricity, by the thermal efficiency of the heat source. The intermediate result, g/BTU-fuel or g/BTU-electric, is calculated with respect to the higher heating value of the fuel, or, for electricity, at 3413 BTUs/kWH. The thermal efficiency, discussed next, is defined as the ratio BTUs of useful heat provided to BTUs of fuel or electrical energy input to the heating device.

The thermal efficiency has two components: the efficiency of conversion of chemical or electrical energy to heat, and the efficiency of heat transfer to the air or surface. Fuel combustion and resistance heating are nearly 100% efficient, unless, in the case of fuel combustion, the burner is operating poorly or with insufficient air and a significant part of the fuel is not burned.

There is significant variability in the transfer efficiency. For electric heaters, which radiate directly into the space or onto the surface of interest, the transfer efficiency is close to 100%. However, for fuel burners, the transfer efficiency can vary from 60% to close to 100%, depending on how much heat and vapor is lost in combustion gases. Units that vent directly to the atmosphere, without dampers, are 60-70% efficient; units with dampers are about 80% efficient. Condensing or recuperative units, which capture most of the water vapor and heat that would normally be vented, are up to 97% efficient. Under Federal law, all gas furnaces manufactured after January 1, 1992, must have a thermal efficiency of at least 78%. (This and similar information is available from various web pages maintained by the California Energy Commission, for example: www.energy.ca.gov/efficiency/appliances/; or www.energy.ca.gov/title24/; see also the EIA's *Assumptions to the Annual Energy Outlook 2001*, 2001).

The EIA's *AEO* and corresponding *Assumptions to the Annual Energy Outlook 2001* report assumes that electric heaters are about 96% efficient, and that fuel-fired heaters are 75-96% efficient (typically 80-90% for new models)[79].

The efficiency trend is assumed to be a double-sided logistic function (Eq. 6). My assumptions are based on the *AEO*s projections of stock average efficiency of space heaters for commercial buildings (Supplemental Table 22 in the *AEO*):

	base year	maximum	minimum	k exp.
LPG	0.77	0.93	0.62	0.040
Natural gas	0.77	0.93	0.62	0.040
Fuel oil	0.77	0.93	0.62	0.040
Electric resistance heating	0.96	1.00	0.92	0.040

"OWN-USE" OF FUEL

Background

In many fuel cycles, the end-use fuel produced is used as a process fuel at some stage. For example, diesel fuel is used by trucks and engines at many points in the diesel fuel cycle. This use of fuel X as a process fuel in fuel cycle X has been called "own-use".

Own-use matters because it reduces the net output of the fuel cycle by the amount that is used internally, which of course increases the amount of feed and fuel that must be processed in order to provide net energy outside of the fuel cycle itself. There are different ways to account for own use, depending on the conventions of the analysis (see Appendix A of DeLuchi [1993]). In LEM, a revised treatment of "own use" has been used that is more consistent across fuel cycles, and that corrects a few simplifications.

The following exposition has two parts:

1) First, I show formally how own-use was handled in the previous version of the model. Even though the original method has been revised, the original

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[79] Actually, according to the CEC web site, efficiencies between 84% and 89% are not common, because they tend to result in acidic condensate. However, I am assuming a weighted average.

approach is shown because it offers the clearest representation of own use, and because the revised method can be shown to be equivalent to the original (i.e., gives the same answer).

2) Second, the new method, which is slightly easier to program, albeit less intuitive, is derived from the original method.

The original method

As shown in Appendix A of DeLuchi (1993), total GHG emissions from stage i of fuel cycle X, in grams of CO_2-equivalent emissions per BTU of end use fuel delivered to consumers, can be represented as:

$$G_{X,i} = \sum_{\mathrm{f}} EN_{X,i,\mathrm{f}} \cdot EM_{\mathrm{f}} \qquad \text{eq. 104}$$

where:
big subscript X = fuelcycle X.
subscript i = stage of fuelcycle X (all stages except end use by vehicles or power plants).
subscript f = process fuel f .
$G_{X,i}$ = g/BTU CO_2-equivalent emissions from stage i of fuelcycle X.
$EN_{X,i,f}$ = use of process fuel f (e.g., electricity, diesel fuel) at stage i of fuel cycle X: BTUs of process fuel f per 1.0 BTU of fuel X made available to end users outside of fuel cycle X.
EM_f = emission factor: grams of CO_2-equivalent emissions per BTU of process fuel f used.

Note that in this representation, $EN_{X,i,f}$ is BTUs of process fuel f per 1.0 BTU of fuel X made available to end users outside of fuel cycle X, *not* BTUs of process fuel per BTU of energy produced by stage i. Generally, 1.0 BTU of energy out of stage i might not end up as one BTU of fuel X made available to end users outside of fuel cycle X, because some of the energy output from stage i might be lost in stages downstream (for example, methanol production requires about 1.5 BTUs of natural-gas input to produce 1.0 BTU of methanol), and some might be used internally within the fuel cycle as a process fuel, and hence be unavailable outside of the fuel cycle. Therefore, given data on process fuel use at a particular stage of the fuel cycle, and energy output of the stage, the parameter $EN_{X,i,f}$ will be shown to be:

$$EN_{X,i,\mathrm{f}} = \frac{P_{X,i,\mathrm{f}} \cdot K^{*}_{X,i}}{1 - U_X} \qquad \text{eq. 105}$$

where:

$EN_{X,i,f}$ is as defined above

$P_{X,i,f}$ = BTUs of process fuel f used at stage i of fuel cycle X, per 1.0 BTU of energy of energy out of stage i (estimated from primary data).

$K^*_{X,i}$ = the stage i energy-conversion or energy-loss factor: BTUs of fuel or feedstock energy out of stage i of fuel cycle X per 1 BTU of fuel energy output from the final stage of the fuel cycle.

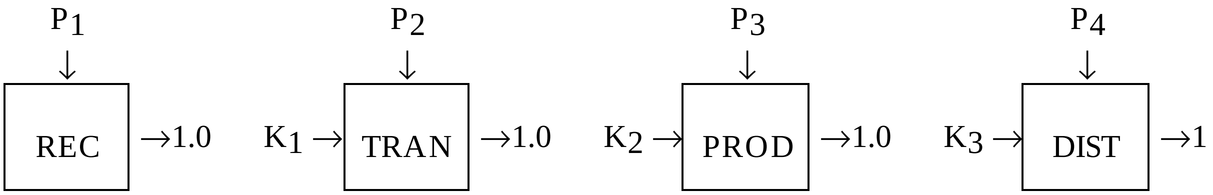
U_X = total internal use (own-use) of fuel X as a process fuel, in fuel cycle X: BTUs of own-use per 1 BTU of X output from the final stage of the fuel cycle X (estimated as a fraction of P_i, at each stage).

There is a subtle difference between the definition of $K^*_{X,i}$ and the definition of $U_{X,}$. In the definition of $K^*_{X,i}$, the 1 BTU of fuel energy output from the final stage of the fuel cycle does *not* include the amount of fuel lost at previous stages; it is, so to speak, the amount at the end of the pipe, after losses all along the pipe. However, in the definition of U_X, the 1 BTU of X output from the final stage includes the amount that is recycled internally, so that the amount available outside of the particular fuel cycle is $1.0 - U_X$.

With this difference in mind, we can see how the lost energy represented by the factor $K^*_{X,i}$ just as well can be counted as internal or own use, and so be incorporated into U_X. Consider, for example, fuel lost to evaporation or leakage during the fuel cycle. If amount of fuel lost from the fuel distribution stage is 5% of the net fuel output of the stage (i.e., the output net of the loss), then the K* factor, as defined above (and discussed further below), is (1+0.05·1)/1 = 1.05. But the lost fuel also can be counted as a sort of own use U (non-combustion own use, in this case). Remembering that in the case of own use the 1 BTU output includes the amount of own use -- in this case, the amount lost -- the parameter U is 0.05/(1+0.05), and the own-use factor 1/(1-U) is 1/(1-0.05/1.05) which equals 1.05, the same as the K* factor.

Estimation of own-use

The following diagram shows energy input and output for a simple four-stage fuel cycle (recovery, transmission, production, and distribution). P_i is the total amount of process energy (from all process fuels f) used in stage i per 1.0 BTU of energy output from stage i, and K_i (not K^*_i) is the number of BTUs from stage i needed as input to stage i+1 in order to produce 1.0 BTU from stage i+1.

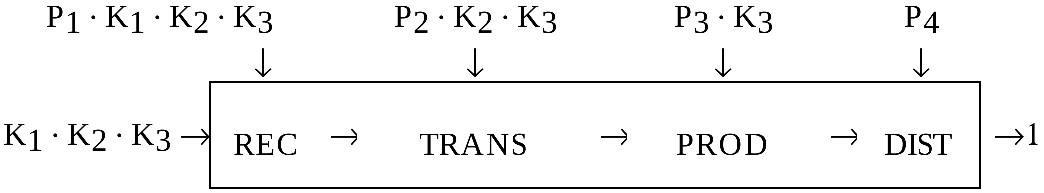


The process energy factors, P_i, are estimated from primary data on process energy use and fuel or feedstock output, at each stage. The conversion / loss factors, K_i, are estimated from energy-in / energy-out data for each stage, and typically are close to 1.0 for all stages except for fuel production. Note that K_i is expressed relative to 1.0 BTU output from stage i+1, whereas K^*_i is expressed relative to 1.0 BTU output from the final stage, such that K^*_i is the product of the K_i from stage i to the penultimate stage:

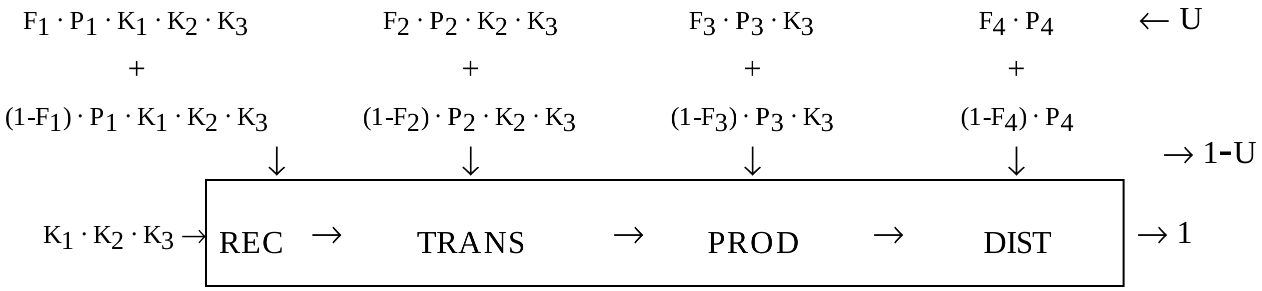
$$K^*_i = K_i \cdot K_{i+1} \cdot \ldots \cdot K_{\text{final-1}}$$

There being no K_{final} because K is expressed relative to 1.0 BTU output from stage i+1 and by definition there is no stage after the final stage.

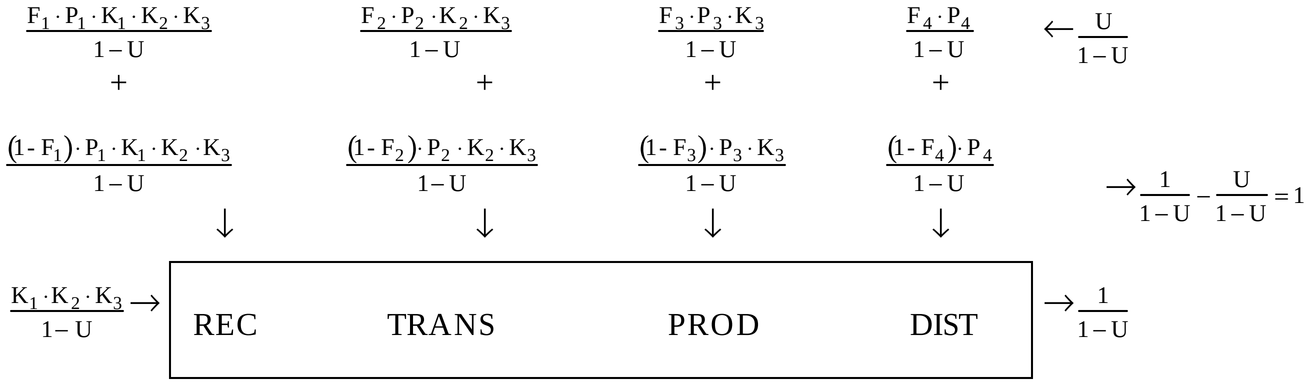
Recall that the overall objective is to express process energy inputs per BTU of final product delivered to consumers outside of the fuel cycle. In order to do this, we must account for the multiplicative effect of the K_i factors, and for own-use of final fuel. First I account for the multiplicative effect of the K_i factors, by representing the four separate stages as one system, the output of which is one BTU of fuel product.



Next, I account for own use. The 1.0 BTU of fuel output from the final stage includes some fuel that is recycled back to the stages of the fuel cycle, as process fuel. Hence, the amount of fuel available to end users outside of the fuel cycle is less than 1.0. Let F_i be the fraction of P_i that is the end use fuel *x that* comes out of the fuel cycle X. We now have:



This diagram shows that, given inputs P_i, the whole fuelcycle produces 1-U BTUs of *x* for end users outside of the fuelcycle itself, where U is the total amount of own use at all stages of the fuelcycle (equal to $F_1 \cdot P_1 \cdot K_1 \cdot K_2 \cdot K_3 + F_2 \cdot P_2 \cdot K_2 \cdot K_3 + F_3 \cdot P_3 \cdot K_3 + F_4 \cdot P_4$). Thus, to end up with 1 BTU of X for end users outside of the fuelcycle, we must scale all inputs by 1 / 1-U:



The foregoing shows energy flows, $EN_{X,i}$. The final step is to incorporate these expressions for $EN_{X,i}$ into the expression for CO_2 - equivalent emissions ($G_{X,i}$), by multiplying them by the appropriate emission factors EM. (Recall from above that $G_{X,i} = M \cdot \sum_f EN_{X,i,f} \cdot EM_f$.) For any fuel cycle X, the emission factor EM_f for any process fuel f that is not the output *x* of X -- i.e., for any non- “own-use” process fuel -- is the full fuel cycle emission factor, where the full fuel cycle includes emissions from production, distribution, etc., as well as from final end-use of the process fuel in fuel cycle X. This should be intuitively clear: for those process fuels outside of the fuel cycle in question, the entire fuel cycle emission must be counted. I designate such a full fuel cycle emission factor as EM_{FC}.

However, with this method, the emission factor EM for own-use fuel *x* in fuel cycle X is just the emission factor for final or direct use of the own-fuel as a process fuel within its fuel cycle. For example, in the method presented above, the appropriate emission factor for diesel fuel used by tanker trucks in the diesel fuel cycle is the emission factor for diesel end-use use by trucks -- *not* the full fuel cycle emission factor for diesel fuel. This is because, in this method, the emissions attributable to the making of the own-use fuel already are accounted for by virtue of the own-use fuel being subtracted from net output. I designate such an end-use-only emission factor as EM_{EU}.

Combining this with the derivation for EN, above, we now can derive the following expression for complete fuel cycle emissions of CO_2-equivalent GHGs (G_X):

$$G_X = \sum_i G_{X,i}$$

from above :

$$G_{X,i} = \sum_f EN_{X,i,f} \cdot EM_f$$

$$= \frac{P_{X,i} \cdot K_{X,i} \cdot F_{x,i}}{1-U_X} \cdot EM_{x,EU} + \frac{P_{X,i} \cdot K_{X,i}}{1-U_X} \cdot \sum_{\mathrm{f} \neq x} F_{\mathrm{f},i} \cdot EM_{\mathrm{f},FC}$$

substituting :

$$G_X = \sum_i \left(\frac{P_{X,i} \cdot K_{X,i} \cdot F_{x,i} \cdot EM_{x,EU} + P_{X,i} \cdot K_{X,i} \cdot \sum_{\mathrm{f} \neq x} F_{\mathrm{f},i} \cdot EM_{\mathrm{f},FC}}{1-U_X} \right)$$

recall that :

$$U_X = \sum_i P_{X,i} \cdot K_{X,i} \cdot F_{x,i}$$

hence :

$$G_X = \frac{U_X \cdot EM_{x,EU} + \sum_i \left(P_{X,i} \cdot K_{X,i} \cdot \sum_{\mathrm{f} \neq x} F_{\mathrm{f},i} \cdot EM_{\mathrm{f},FC} \right)}{1-U_X} \qquad \text{eq. 106}$$

where:

little subscript *x* = fuel x produced by fuelcycle X.
$G_{X,i}$, $EN_{X,i,f}$, EM_f, $P_{X,i}$, $K_{X,i}$, and U_X are as defined above
G_X = complete fuel cycle CO_2-equivalent emissions of greenhouse gases from the entire fuel cycle X, except end use, per BTU of fuel output.
$F_{x,i}$ = the fraction of P_i that is the end use fuel *x that* comes out of the fuel cycle X.
$F_{f,i}$ = the fraction of P_i that is process fuel f.
$EM_{x,EU}$ = emission factor for end use of own-fuel *x* (CO_2-equivalent g/BTU).
$EM_{f,FC}$ = emission factor for full fuel cycle production and use (including end use) of process fuel f (CO_2-equivalent g/BTU).

Development of an equivalent, simpler method

The method just developed is appealing because it is derived from a clear, general representation of a fuel cycle. It does, however, have two minor disadvantages. First, it requires that own-use U_X be estimated for the entire fuel cycle. Second, it requires two different kinds of emission factors: $EM_{X,EU}$ for own-use fuel, and $EM_{f,FC}$ for other fuels.

Because of these disadvantages, I have derived from the method above a simpler, but less intuitive method that does not require the estimation of own use U_X, or the use of different kinds of emission factors. The method is:

$$G_{X,i} = P_{X,i} \cdot K_{X,i} \cdot \sum_{\mathrm{f}} F_{\mathrm{f},i} \cdot EM_{\mathrm{f},FC}$$

$$G_X = \sum_i G_{X,i} \qquad \text{eq. 107}$$

Where all the terms are as defined previously, and the summation over f process fuels includes the own-use fuel *x*. The advantages of this method are that it does not require the estimation of U_X per se, or the designation of separate kinds of emission factors for own-use fuel. The notation and program is simpler than the original method.

This new method can be shown to be equivalent to the original method. First, expand the expression for $G_{X,i}$ into terms for own use fuel *x* and other fuels f:

$$G_{X,i} = P_{X,i} \cdot K_{X,i} \cdot F_{x,i} \cdot EM_{x,FC} + P_{X,i} \cdot K_{X,i} \cdot \sum_{\mathrm{f} \neq x} F_{\mathrm{f},i} \cdot EM_{\mathrm{f},FC}$$

where:

$EM_{x,FC}$ = emission factor for full fuel cycle production and use, including *end use*, of own fuel *x* (CO_2-equivalent g/BTU).

Now substitute this expression for $G_{X,i}$ into the expression for G_X:

$$G_X = \sum_i \left(P_{X,i} \cdot K_{X,i} \cdot F_{x,i} \cdot EM_{x,FC} + P_{X,i} \cdot K_{X,i} \cdot \sum_{\mathrm{f} \neq x} F_{\mathrm{f},i} \cdot EM_{f,FC} \right)$$

$$= EM_{x,FC} \cdot \sum_i P_{X,i} \cdot K_{X,i} \cdot F_{x,i} + \sum_i \left(P_{X,i} \cdot K_{X,i} \cdot \sum_{\mathrm{f} \neq x} F_{\mathrm{f},i} \cdot EM_{f,FC} \right)$$

$$= EM_{x,FC} \cdot U_X + \sum_i \left(P_{X,i} \cdot K_{X,i} \cdot \sum_{\mathrm{f} \neq x} F_{\mathrm{f},i} \cdot EM_{f,FC} \right)$$

Now, let:

$$EM_{x,FC} \equiv EM_{x,FC*} + EM_{x,EU}$$

$$\sum_{i} \left(P_{X,i} \cdot K_{X,i} \cdot \sum_{\mathrm{f} \neq x} F_{\mathrm{f},i} \cdot EM_{\mathrm{f},FC} \right) \equiv O$$

then:

$$\mathrm{G}_{\mathrm{X}} = \left(EM_{x,EU} + EM_{x,FC*} \right) \cdot U_X + O = U_X \cdot EM_{x,EU} + U_X \cdot EM_{x,FC*} + O$$

where $\mathrm{EM}_{\mathrm{x,FC*}}$ = complete fuel cycle emission factor for fuel *x*, except end-use emissions. Note, though, that $\mathrm{EM}_{\mathrm{x,FC*}}$ is just G_{X}: complete fuel cycle $\mathrm{CO_2}$-equivalent emissions of greenhouse gases from the entire fuel cycle X, except end use[80]. Hence, we have:

$$G_X = U_X \cdot EM_{x,EU} + U_X \cdot G_X + O$$

Rearranging, we get[81]:

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[80] This definition assumes that the own use of fuel *x* involves the same stages as other end uses of fuel *x*. Put another way, it means that adding or subtracting a stage from fuelcycle X results in a different fuelcycle. Thus, in principle, the natural gas-to-vehicles fuelcycle should be represented separately from the natural gas-to-power plants fuelcycle, because the vehicle cycle has two stages, low-pressure distribution system and natural gas compression, that the power-plant cycle doesn't. Accordingly, I have characterized several different natural gas fuelcycles.

[81] Alternatively, from this point, we can make an infinite number of substitutions of $U_X \cdot EM_{x,EU} + U_X \cdot G_X + O$ for G_{X}. After two more such substitutions we have:

$$G_X = U_X \cdot EM_{x,EU} + U_X \cdot \left(U_X \cdot EM_{x,EU} + U_X \cdot \left(U_X \cdot EM_{x,EU} + U_X \cdot G_X + O \right) + O \right) + O$$

$$= U_X \cdot EM_{x,EU} + U_X^2 \cdot EM_{x,EU} + U_X^3 \cdot EM_{x,EU} + U_X^3 \cdot G_X + O + U_X \cdot O + U_X^2 \cdot O$$

$$= U_X \cdot EM_{x,EU} \cdot \left(1 + U_X + U_X^2 \right) + O \cdot \left(1 + U_X + U_X^2 \right) + U_X^3 \cdot G_X$$

With an infinite number of substitutions for G_{X}, we have:

$$G_X = U_X \cdot EM_{x,EU} \cdot \left(1 + U_X + U_X^2 + U_X^3 + \ldots \right) + O \cdot \left(1 + U_X + U_X^2 + U_X^3 + \ldots \right)$$

$$= \left(U_X \cdot EM_{x,EU} + O \right) \cdot \left(1 + U_X + U_X^2 + U_X^3 + \ldots \right)$$

$$G_X - U_X \cdot G_X = U_X \cdot EM_{x,EU} + O$$

$$G_X = \frac{U_X \cdot EM_{x,EU} + O}{1 - U_X}$$

$$= \frac{U_X \cdot EM_{x,EU} + \sum_i \left(P_{X,i} \cdot K_{X,i} \cdot \sum_{f \neq x} F_{f,i} \cdot EM_{f,FC} \right)}{1 - U_X} \qquad \text{eq. 108}$$

with the last being the original expression derived earlier.

Application of the new method

The model now uses the new method -- $G_{X,i} = P_{X,i} \cdot K_{X,i} \cdot \sum_{f} F_{f,i} \cdot EM_{f,FC}$ -- to calculate g/BTU CO_2-equivalent emissions of GHGs from all stages of the fuel cycle except end use. It is evident from the demonstration above that this new method is circular, or recursive: emissions at each stage ($G_{X,i}$) depend on total fuel cycle emissions (G_X), which is the sum of emissions from each stage: $G_X = \sum_i G_{X,i};\ G_{X,i} = f(G_X)$. The spreadsheet handles this circularity by iterative calculations, and converges on a solution after 20 or so iterations (as revealed by comparing the results of the new method with the results of the old method, which is not circular in the same way). Thus, the new method in effect transfers some of the work of estimating fuel cycle GHG emissions from me to the spreadsheet.

In the new method, the factor K is used to account for energy lost by evaporation or leakage, and for energy lost in feedstock-to-fuel conversion processes. For example, in the conversion of natural gas to methanol, about 1.5 BTUs of natural gas are required to produce 1.0 BTU of methanol. Although it would be possible to treat 0.5 BTUs of natural gas as an additional fuel input used to "process" the 1.0 BTU of natural gas that emerges as 1.0 BTU of methanol, it would be awkward to do so.

In the case of fuel loss, by leakage or evaporation, the K factor for any particular stage i is equal to $1+L_{i+1}$, where L_{i+1} is the loss from stage i+1 as a fraction of the output from stage i+1. To see this, recall the definition of K_i: the number of BTUs from stage i needed as input to stage i+1 in order to produce 1 BTU from stage i+1. If

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The second term in this expression is the binomial expansion of $(1-U_X)^{-1}$. Hence:

$$G_X = \left(U_X \cdot EM_{x,EU} + O\right) \cdot \left(1 - U_X\right)^{-1}$$

we have an output from stage i+1 of 1.0, and a loss within stage i+1 that is some fraction L_{i+1} of the output of 1, then the total output of stage i needed as input to stage i+1, to produce 1 BTU from stage i+ 1, is $1 + L_{i+1} \cdot 1$ (because $1 + L_{i+1} \cdot 1 - L_{i+1} \cdot 1 = 1$). In this analysis, I estimate fuel loss as a fraction of output (net of loss)[82], so that the K_i factor is simply $1+L_{i+1}$.

Related changes

Where an energy source X is used to recover energy source X (e.g., coal used at the mine site as a source of energy), the fuel cycle emissions for such "own use" should not include emissions from a feedstock transmission stage. I have adjusted the model accordingly.

QUALITATIVE DISCUSSION OF RESULTS FROM THE REVISED GHG EMISSIONS MODEL

This final section presents a qualitative discussion of the results from the LEM. The discussion here refers to tables of results in the LEM itself (the results tables in the LEM are numbered), but only a few of these results tables are reproduced in this report. A complete set may be published in a separate report.

Energy efficiency and emissions of vehicles.

Vehicle efficiency is one of the most important calculated parameters in the GHG emissions model, because it linearly determines fuel cycle emissions of CO_2. In the model, the efficiency of the vehicle is determined by the mi/BTU efficiency of the AFV engine or powertrain relative to that of the baseline gasoline or diesel vehicle, the weight of the vehicle, and other parameters. The weight of the vehicle, in turn, is a function of the driving range, the characteristics of the fuel storage systems, and other factors.

The input parameters for the calculation of vehicle energy use are discussed above. The calculated weight results are shown in Table 50b, and the calculated overall efficiency and fuel-use results are shown in Table 50c. The efficiency of the EV relative to efficiency of the baseline gasoline vehicle has increased, and as a result fuel cycle GHG emissions from EVs are significantly lower.

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[82] Note that if the loss for stage i+1 is expressed as a fraction of the output from stage i -- which is the input to stage i+1 -- then the K_i factor is equal to output of stage i, O_i, divided by the output of stage i+1, which is equal to the input O_i minus the loss of $O_i \cdot L^*_{i+1}$: $O_i / (O - O_i \cdot L^*_{i+1})$, or $1/(1 - L^*_{i+1})$. Alternatively, and perhaps more simply, one can transform a loss given initially with respect to input into a loss with respect to the output: $L = L^*/(1-L^*)$, where the asterisk denotes the loss with respect to the input.

The single most important parameter here is the energy conversion efficiency of the vehicle: the relative thermal efficiency in the case of AF ICEVs, and relative powertrain efficiency in the case of EVs. Driving range and vehicle weight are less important because they affect vehicle efficiency only indirectly. (Over the typical range of variation of both driving range and fuel-storage characteristics, the fuel cycle CO_2-equivalent emissions vary by only 1-2%.)

The calculated g / mi emissions are shown in Table 50d.

Energy intensity of fuel cycles

Table 51a presents the new calculated energy intensities by stage of the fuel cycle, in BTUs of process energy used at each stage per BTU of fuel made available to end users. These results differ from the Table 3 results of DeLuchi (1991) because, as discussed above, the underlying assumptions and representations of process efficiency have changed. The most significant changes are those relating to the energy requirements of fuel production (e.g., methanol production from natural gas); less significant are those relating to the energy requirements of fuel and feedstock transport.

Table 51b shows BTUs of process energy consumed per vehicle mile of travel.

Kinds of process fuel used

Table 52 summarizes the calculated and input breakdown of the kinds of energy used at each stage of the fuel cycle. As noted above, this table has been broken into three parts: one for feedstocks, one for fuels, and one for distribution of liquid fuels. Virtually all of the changes calculated here have only a minor effect on fuel cycle CO_2-equivalent emissions. (An exception is the change in the mix of fuels used to provide process heat at corn-to-ethanol plants.)

Leaks of methane and CO_2

The data and methods used to estimate leaks from natural-gas systems, venting and flaring of gas associated with oil production, and methane emissions from coal mines have been completely revised. As a result, calculated venting and flaring emissions from oil wells have increased by a minor amount, calculated leaks from natural-gas systems have increased substantially, and calculated emissions from coal mining have decreased substantially. Table 24 shows parameters in the estimation of leaks from coal mining, and Table 28 shows parameters in the estimation of leaks from NG systems.

The increase in the calculated leakage rate from NG systems increases fuel-cycle emissions by about 7 g / mi, or 2%. The decrease in calculated methane emissions from coal mining decreases CO_2-equivalent emissions from the coal-to-electricity fuel cycle by about 2%.

Leaks of hydrogen

The data and methods used to estimate leaks from hydrogen stations, vehicles, and pipelines also have been completely revised. Moreover, as discussed in the section on CO_2-equivalency factors (CEFs), a CEF for hydrogen has been added, to account for the effect of hydrogen leaks on concentrations of methane and troposperic ozone. The following table shows the CO_2-equivalent gram / mile fuelcycle emissions (not including emissions from the lifecycle of materials or vehicles) without and with a CEF for hydrogen, and the resulting percentage increase in fuelcycle emission:

	Light-duty FCEV (H_2 / water)	Light-duty FCEV (H_2 / NG)	Heavy-duty ICE (H_2 / NG)
Compressed H_2	42.8, 44.5 (4.0%)	197, 198 (0.4%)	2497, 2507 (0.4%)
Liquefied H_2 (central.)	116.2, 119.2 (2.6%)	273, 276 (0.9%)	3345, 3375 (0.9%)

The increase in the CO_2-equivalent emissions due to assigning a non-zero CEF to hydrogen, compared with a CEF of zero, ranges from less than 1% in the case of vehicles using compressed hydrogen made from natural gas, to 3-4%, in the case of vehicles using liquid hydrogen made from electrolysis of water. The use of liquefied rather than compressed hydrogen results in higher leakage, and hence higher CO_2-equivalent emissions, because of boil-off losses associated with liquid-fuel transfers. The use of hydrogen made from water rather than from natural gas results in higher hydrogen leakage, and hence higher CO_2-equivalent emissions, because of the assumption that there are hydrogen pipelines in the case of hydrogen from water but not in the case of hydrogen from natural gas.

This analysis has explicit estimates of leakage from vehicular storage and fuel systems, fuel-cell stacks,fuel dispensing, other liquid-fuel transfers, pipeline distribution, pipeline transmission, and pipeline compressors. However, there are very few data on hydrogen leakage rates, and our estimates may be substantially wrong. Note, too, that as regards comparing lifecycle GHG emissions from hydrogen fuel-cell vehicles with lifecycle GHG emissions from fossil-fuel internal-combustion-engine vehicles, we have not included emissions of hydrogen from the incomplete combustion of fossil fuels. We do not know the magnitude of this source, and hence do not know how the omission might affect the comparison.

Electricity generation: efficiency and mix of fuels,

As discussed above, I have projected the efficiency of electricity generation and the mix of fuels used for generic national power. Tables 53a and 53b show the new projected efficiencies and fuel mixes.

For most years, the projected generation efficiency is higher than that assumed in Appendix D of DeLuchi (1993), and as a result emissions from fuel cycles that consume a lot of electricity (such as the EV fuel cycle) are lower.

The new national marginal recharging mix for EVs has more coal and less gas than did the one in the previously documented version of the model, and hence by itself results in higher fuel cycle GHG emissions from EVs.

Fuel cycle emissions from the use of electricity

As discussed above, I have updated most of the emission factors for power plants. Table 53c shows the new CO_2-equivalent emissions from power plants, by pollutant, and total fuel cycle emissions from the end use of electricity. The changes to the emission factors for utility boilers have only a minor effect on the CO_2-equivalent fuel cycle emissions.

Grams emitted per 10^6 BTU of fuel delivered to end users, by stage and feedstock/fuel combination.

Table 54 shows the new calculated CO_2-equivalent emissions per unit of energy delivered to end users, by stage of the fuel cycle and feedstock / fuel combination. These results are useful mainly for the purpose of estimating GHG emissions from non-transportation fuel cycles. For example, one can use the g / 10^6-BTU results for the NG fuel cycle to estimate emissions from use of NG for home heating. (One still must estimate emissions from final end-use combustion of the gas in the home, of course.)

Table 55 shows the calculated g / 10^6-BTU emissions of each individual greenhouse gas, without the equivalency factors applied. That is, Table 55 shows the actual mass emissions, not the CO_2-equivalents, of the different greenhouse gases, whereas 54 here and Table 10 in DeLuchi (1991) show the CO_2 equivalents. One can calculate CO_2 equivalents from the data of Table 55 simply by multiplying actual emissions by the CO_2-equivalency factors (Appendix D).

These unweighted emissions, by stage of the fuel cycle, can be used as part of an analysis of criteria-pollutant emissions.

Finally, Table 56 summarizes the results of Tables 54 and 55.

Upstream fuel cycle and material lifecycle emissions expressed relative to end-use emissions.

For perspective, Table 63 expresses upstream emissions of each pollutant as a percentage of vehicular emissions of the pollutant. Table 65 shows emissions from the materials lifecycle and vehicle assembly and transport, in the "natural" units of g/lb, and also as a percentage of vehicular emissions.

These relative percentages are interesting in several respects. In all cases, upstream emissions of CH_4 and SO_X exceed vehicular emissions by a wide margin. In most cases, upstream emissions of PM (BC+OM) exceed vehicular emissions. This is significant because all three are potent greenhouse gases, and because SO_X and PM are the most damaging of all urban pollutants, per kg emitted (Delucchi, 2000b). If humans, materials, crops, and other "recipients" of pollutant damage were as exposed to upstream emissions as to vehicular emissions, then upstream emissions probably would be more damaging (per mile of travel) than vehicular emissions. However, in most places, people are much more exposed to vehicular emissions than to emissions from, say, petroleum refineries or automobile plants, which generally are not located in the center of metropolitan areas (Delucchi and McCubbin, 1996). The remoteness of upstream sources greatly diminishes the impact of their relatively high emissions of SO_X and PM, with the result that the health-damage cost per mile of fuel-upstream and material-lifecycle emissions is considerably less than the damage cost per mile of vehicular emissions (McCubbin and Delucchi, 1999).

Upstream and material-lifecycle emissions of CO and N_2O are relatively minor, except for the ethanol fuel cycles, which produce large amounts of N_2O from the use of fertilizers for the biofuel crops. Upstream and material-lifecycle emissions of NO_X and NMOCs generally are significant fractions of vehicular emissions, and in some fuel cycles (e.g., ethanol) exceed vehicular emissions. Upstream CO_2, NO_X, and CO_2-equivalent emissions are large in those fuel cycles in which fuel production is relatively energy intensive (such as ethanol, methanol, and hydrogen from natural gas).

My findings with regards to emissions from the lifecycle of materials used in vehicles (Table 60) are similar to those in Maclean and Lave (1998) and Tahara et al. (2001). For example, Tahara et al. (2001) estimate that the lifecycle of automotive materials emits about 1.6 lbs of CO_2 per lb of vehicle, and that assembly emits about 1.0 lbs of CO_2 per lb of vehicle. I estimate that the lifecycle of materials emits about 1.5 lbs of CO_2 per lb of vehicle, and that assembly emits about 0.3 lbs of CO_2 per lb of vehicle. It is possible that my estimate of assembly energy do not account adequately for energy used to assemble parts at establishments not included in the automotive manufacturing sector.

Gram-per-mile emissions by vehicle/fuel/feedstock combination, and stage of the fuel cycle.

Table 57 presents the new final g/mi results by vehicle/fuel/feedstock, and stage of the fuel cycle. These can be compared with the results of the previous analysis (Tables 9 and 12 of DeLuchi [1991]).

Results of the analysis of fuels for space heating and water heating

Table 61 shows the total fuel cycle CO_2-equivalent emissions from the use of NG, LPG, fuel oil, and electricity for space heating and water heating. The results are shown in terms of grams of CO_2-equivalent emissions per million BTU of useful heat provided, counting the CO_2-equivalent effect of all of the pollutants included in the model, from all of the stages of the fuel cycle. Table 61 also shows the percentage difference between each fuel and natural gas, which has the lowest fuel cycle emissions. LPG has the next lowest, followed by fuel oil and then electricity from various sources. The differences in the results for different target years are not important.

There are two significant differences between the results estimated here for space heating and water heating, and the results estimated for the use of transportation fuels.

First, LPG fares slightly worse relative to NG in the space and water heating application than in the transportation application. This difference is due mainly to end-use emissions of methane: natural gas vehicles have relatively high emissions of CH_4, but natural gas heaters have very low emissions. For example, in the case of transportation, end-use emissions of CH_4 from CNG vehicles are, by themselves, more than 10% of total fuel cycle CO_2-equivalent emissions, and also 10 times higher than CH_4 emissions from LPG vehicles. However, in the case of space and water heaters, CH_4 emissions from natural gas are less than 0.1% of total fuel cycle emissions. Moreover, CH_4 emissions from natural gas heaters are, according to the EPA, slightly *less* than CH_4 emissions from LPG (Table 34). Another, less important factor is that in the case of transportation, there are significant emissions associated with compressing the natural gas at the end of the pipeline, whereas in the case of heating with NG there is not.

Is the difference in end-use emission factors reasonable? Heaters, like utility boilers, are external combustion devices, whereas car engines are internal combustion devices, and it does seem reasonable that external combustion is more complete, and hence produces less organic pollution (CH_4, CO, and NMOC), than does internal combustion. The EPA’s emission factors for utility boilers, which are based on a large number of tests, show the same pattern as do the emission factors for space heaters: CH_4 emissions are a tiny fraction -- less than 0.01% -- of fuel cycle CO_2-equivalent emissions for natural-gas power plants. Moreover, the CH_4 emission factors for natural-gas turbines, which are internal combustion devices, are about 100 times those for utility boilers. CO and NMOC emissions have the same patterns.

The second significant difference between the results for space-heating and water-heating fuels, and the results for transportation fuels, is the poor showing of electricity as a source of heat. In this analysis, electricity has fuelcycle emissions two to four times higher than those for NG, LPG, or fuel oil, whereas in the case of transportation, electric vehicles have lower fuel cycle emissions than do gasoline, diesel, CNG, or LPG vehicles. This is attributable to a dramatic difference in end-use efficiency. The electric vehicle is severalfold more efficient at converting a BTU of electricity (from the wall) into a mile of travel than an internal combustion engine vehicle is at converting a BTU of fuel into a mile of travel, but an electric resistance heater is only 10-20% more efficient at converting a BTU of electricity into a BTU of useful heat than a fuel burner is at converting a BTU of fuel into a BTU of useful heat.

Analytical issues

The quality of the model used in this analysis can be considered in terms of scope (what isn’t included that should be?), structure (which processes are not represented accurately?), and uncertainty of input parameters (what is not well known?).

Scope. An ideal analysis of fuel cycle emissions and energy use would include all energy-consuming and pollutant-emitting processes, and all pollutants, in complete and correct detail. With respect to this ideal, the model used in this analysis falls short in several ways:

- It includes only air-pollutant emissions; it does not include water pollutants, or other kinds of environmental impacts, such as soil erosion. A complete lifecycle environmental comparison should consider all environmental impacts
- It does not include at least two major kinds of air pollution: emissions of particulate matter in dust (e.g., dust from highways, agricultural operations, or coal mining), and emissions of volatile organic compounds from plants (e.g., terpenes from trees used in short-rotation intensive cultivation). Inclusion of these sources of pollutant could change the relative attractiveness of different fuel cycles.
- Although it includes emissions associated with materials manufacture and assembly for vehicles, trains, and ships, it does not include emissions associated with materials used for large construction projects such as power plants and refineries. Although generally these emissions are small compared to the emissions from fuel production and use (especially end use), they might add nontrivially to some fuel cycle totals.
- It includes only a few second-order “price effects”. All fuel cycles are part of an economic system as well as physical/technological system. When one makes, say, additional gasoline, one does not merely get the emissions associated with making versus not making the additional amount of gasoline; one also affects the price and therefore the consumption of other, economically related fuels. The change in the

consumption of these other fuels will affect air-pollutant emissions. Ideally, these market-driven changes in pollution should be considered along with the "first-order" emissions due to making versus not making the additional fuel. (For details, see Delucchi [2002].) This analysis considers only a few such effects, mainly as regards the marketing of the co-products of some production processes (e.g., the marketing of the co-products of corn-to-ethanol conversion). I do not know how a complete consideration of price or market effects would affect the results.

The structure of the analysis. Although most parts of the fuel cycle model contain reasonably detailed representations, there are a few important simplifications that can lead to misleading or internally inconsistent results:

- Generally, the model uses average rather than "marginal" emission-reduction factors. For example, the model calculates the average emissions for all coal-fired boilers used in industry, on the basis of the projected extent and effectiveness of emission controls. It does not distinguish industries or processes in which all boilers will be controlled from industries or processes in which few boilers will be controlled. This results in an overestimate of emissions from new sources, which are required to meet New Source Performance Review Standards, and an underestimation of emissions from old sources not subject to emission controls.
- The apportionment of refinery energy use and emissions to the different products (gasoline, distillates, residual fuel, and so on) is an input rather than an estimated parameter. Ideally, the fuel cycle model would contain a mini refinery model to calculate energy use and emissions attributable to each refinery product.
- A few important parameters are not projected year-by-year through 2050, as are many unimportant parameters are, but rather are fixed at year 2000 values.
- The calculation of second-order energy use and emissions related to the manufacture and servicing of transportation modes (trains, ships, trucks, and pipelines) also is an input rather than a calculated parameter, and might in fact be inconsistent with other calculations in the analysis.

Uncertainty in important parameter values. All parameter values are uncertain to some degree. In some cases, the uncertainty is great enough, and the parameter values important enough, to significantly affect the certainty of the overall results. The most important uncertainties in this analysis are:

- *The CO_2-equivalency factors (CEFs) for all non-CO_2 greenhouse gases.* The uncertainty in the CEFs for CH_4, N_2O, N (as NO_x, or nitrogen in fertilizer), SO_2, and PM can have a significant effect on the overall results. The uncertainty in the CEFs for CO and NMOCs is less important: varying these CEFs over their likely range of values does not significantly affect the results. In any case, the uncertainty in the CEFs runs deep: most of the existing estimates do not incorporate several important effects, and in many cases the effects considered are not well characterized.
- *Efficiency of end use.* In all fuel cycles, the efficiency of energy end use is important and still uncertain. In particular, in the EV cycle, the major uncertainty remains the relative energy use of EVs (both BPEVs andFCEVs) although the new

energy-use model described briefly in Appendix G has helped to narrow that uncertainty. The effect of the mix of fuels used to generate power is reasonably well reflected in the regional results.

There also is non-trivial uncertainty in the composition and cycle life of batteries for EVs. The cycle life is important because the shorter the cycle life (in miles of travel), the higher the g/mi lifetime emissions.

• *The evolution of fuel-production technology.* Generally, I have assumed that production processes will continue to get more efficient, and gradually switch from high-emitting to low-emitting process fuels. Historically there is some justification for these assumptions. For example, in the 1980s, high fuel prices led to considerable improvements in the fuel efficiency of corn-to-ethanol conversion processes, and environmental and other considerations spurred a switch from coal to natural gas. It is not clear, however, to what extent these trends can be expected to continue. And the problem of prediction is even more difficult for those technologies, such as wood-to-ethanol, that are still being developed.

• *Emissions related to changes in cultivation and land use.* In the biomass fuel cycles, the most uncertain and important parameters, aside from those mentioned above, are those that represent which land uses (e.g., forests, pasture land, or agricultural land) are replaced by which energy crop systems (corn, soybeans, switchgrass, or SRIC trees), and those pertaining to N_2O emission related to nitrogen fertilizer inputs. In some cases (e.g, the biodiesel fuel cycle), uncertainty regarding N inputs can have an enormous impact on fuel cycle CO_2-equivalent emissions.

• *The effect of quantity changes on prices and hence demand and, ultimately, supply in other markets.* In a few instances I account, crudely, for economic effects in the markets for products related to the co-products of fuel cycles (e.g., in markets for electricity affected by the generation of power from excess lignin in biomass-to-ethanol plants). The values of these parameters are uncertain and can signficantly affect fuelcycle CO_2-equivalent emissions.

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REFERENCES

See Appendix Z

<<TABLES 1A, 1B, 1C, AND 2 MOVED TO APPENDIX D>>

TABLE 3. COMPOSITION (VOLUME %) AND PROPERTIES OF CONVENTIONAL AND REFORMULATED GASOLINE

Assumed volume % of:	**Conventional**[a]	**Reformulated**[b]
alkanes	58.8	59.3
aromatics	32.0	25.4
olefins	9.2	4.1
MTBE	0.0	11.2
Calculated properties:[c]		
Density (g/liter)	749.1	738.7
Higher heating value (kJ/g)	46.5	45.6
Carbon weight fraction	0.866	0.842
sulfur weight fraction	0.000339	d
Higher heating value (10^6 BTU/gal)	0.1250	0.1208

[a] The national average composition in 1988 (Auto/Oil Air Quality Improvement Research Program, 1995, 1997).

[b] Except for the sulfur content, these are the characteristics of a gasoline blended to meet the California regulatory requirement for Phase II reformulated gasoline (Auto/Oil Air Quality Improvement Research Program, 1995, 1997). (This is similar to the specification in Singh and McNutt (1993): 11.7% MTBE, 10.3% olefins, 24.6% aromatics, and 55.4% alkanes.) The sulfur content is assumed to decline from the present average of 236 to ppm to 35 ppm.

[c] Calculated on the basis of the chemical properties of the individual compounds or classes of compounds. The calculated density and heating value match those reported by the Auto/Oil Program (1995).

[d] Assumed to decline from the present average of 236 to ppm to 35 ppm. See the text.

TABLE 4. PROJECTIONS OF COAL QUALITY

	1994	**% change/yr.**
Utility coal heating value (10^6 BTU / ton)[a]	20.673	-0.30
Utility coal carbon weight fraction[b]	0.586	-0.26
Utility coal ash weight fraction[c]	0.094	-0.90
Utility coal sulfur weight fraction[d]	0.0117	0.0040 / 0.04
Generic industrial coal heating value (10^6 BTU / ton)[a]	22.068	-0.30
Generic industrial coal carbon weight fraction[b]	0.6235	-0.26
Generic industrial coal ash weight fraction[e]	0.094	-0.90
Generic industrial coal sulfur weight fraction[e]	0.0117	0.0040 / 0.04

[a] The 1994 value is from EIA's *AER 1995* (1996). Over the past 20 years or so, the heating value has declined by about 0.34% per year (EIA, *AER 1995,* 1996). I project a similar rate for the next 20 years.

[b] The 1994 value is chosen so that it and the 1994 heating value result in the carbon / BTU value estimated by the EIA (*AER 1995,* 1996) for 1994. The projected change per year is that rate which, when combined with the historical change in heating value, results in the change in carbon / BTU values over the past 20 years or so (EIA, *AER 1995*, 1996). Note that generally, if the heating value decreases, the carbon content decreases (Hong and Slatick, 1994).

[c] The 1994 value is from the EIA's *Electric Power Annual 1994* (1995).The projected change per year is the average rate from 1984 to 1994 (EIA, *Electric Power Annual 1994,* 1995; EIA, *Coal Industry Annual 1993*, 1994).

[d] The 1994 value is from the EIA's *Electric Power Annual 1994* (1995). From 1984 to 1994, the sulfur weight fraction of coal declined by about 2.3% per year (EIA, *Electric Power Annual 1994*, 1995; EIA, *Coal Industry Annual 1993*, 1994). It appears that the EIA projects that the sulfur weight fraction will continue to decline, albeit at a slightly lower rate. In its *Supplement to the AEO 1996*, the EIA projects the production of low-sulfur (0-0.60 lbs S per 10^6 BTU), medium sulfur (0.61-1.67 lbs S per 10^6 BTU), and high-sulfur coal (more than 1.67 lbs S per 10^6 BTU) through the year 2015 (EIA, 1996). I estimate from these data that the sulfur weight fraction declines about 1.6% per year. Instead of specifying a percentage change per year, however, I use a one-sided logistic function (Eq. 4) with the lower limit shown to the left of the slash, and the k exponent (steepness parameter) shown to the right.

[e] Assumed to be the same as the sulfur content and ash content of utility coal.

TABLE 5. CHARACTERISTICS OF FUEL GASES

A. CHARACTERISTICS OF MOLECULAR CONSTITUENTS OF FUEL GASES

	HHV[a]	**Mass**	**Carbon**[b]	**Van der Waal's**[c]		**Conc.**[d]	**CEF**[e]
	kJ/mole	*g/mole*	*wt. %*	*a*	*b*	*m/L*	*mass*
CH_4	890.31	16.043	74.87%	2.253	0.04278	0.04230	21.00
C_2H_6	1559.84	30.07	79.89%	5.498	0.06380	0.04252	3.05
C_3H_8	2219.90	44.097	81.71%	8.664	0.08556	0.04272	3.24
C_4H_{10}	2877.40	58.123	82.66%	13.670	0.11840	0.04306	4.03
C_5H_{12}+(f)	3775.00	77.495	83.40%	21.060	0.15648	0.04360	5.06
CO_2	0.00	44.0098	27.29%	3.592	0.04267	0.04241	1.00
CO	0.00	28.0098	42.88%	1.485	0.03985	0.04225	4.00
N_2	0.00	28.014	0.00%	1.390	0.03913	0.04225	0.00
H_2	285.83	2.016	0.00%	0.244	0.02661	0.04218	0.00
H_2S	562.01	34.076	0.00%	4.431	0.04287	0.04247	-16.58
H_2O	0.00	18.0153	0.00%	5.464	0.03049	0.04257	0.00

B. MOLAR FRACTIONS OF MOLECULAR COMPOUNDS IN FUEL GASES

compound	**raw NG**[g]	**dry NG**[h]	**coalbed**	**refinery**	**LPG**	**H_2 /NG**[i]
CH_4	0.867	0.938	0.964	0.420	0.000	0.003
C_2H_6	0.043	0.032	0.002	0.420	0.000	0.000
C_3H_8	0.020	0.008	0.000	0.030	0.950	0.000
C_4H_{10}	0.013	0.003	0.000	0.009	0.045	0.000
C_5H_{12}+	0.010	0.002	0.000	0.001	0.005	0.000
CO_2	0.023	0.008	0.010	0.010	0.000	0.003
CO	0.000	0.000	0.000	0.000	0.000	0.003
N_2	0.015	0.009	0.024	0.000	0.000	0.003
H_2	0.000	0.000	0.000	0.100	0.000	0.990
H_2S	0.010	0.000	0.000	0.00005	0.000	0.000
H_2O	0.000	0.000	0.000	0.010	0.000	0.000

[a] At 298° K. From the *CRC Handbook of Chemistry and Physics* (1975, 1984), except:

- value for butane (C_4H_{10}) is average of value for butane and isobutane;
- value for pentanes-plus ($C_5H_{12}+$) is 62% of the value for n-pentane and 38% of the value for n-hexane;
- value for H_2S is calculated as the heat of formation of $SO_2 + H_2O - H_2S$.

[b] Equal to 12.011·n / Mass · 100%, where *n* is the number of carbon atoms in the molecular, and Mass is the molecular mass from the "Mass" column.

[c] The gas-specific constants in Van der Waal's equation for real gases (*CRC Handbook of Chemistry and Physics*, 1984). The constant *a* is a measure of the attractive forces between molecules; the constant *b* accounts for the finite volume of the molecules themselves. See the discussion in the text.

[d] The molar concentration, calculated using der Waal's equation for real gases. See the discussion in the text.

[e] The CO_2--equivalency factor, on a mass basis. The values for CH_4, CO, and C_2 to C_{5+} alkanes are derived in Appendix D. CO_2 is defined to have a CEF of 1.0. I assume that the CEFs for N_2, H_2, and H_2O are zero. The value for H_2S is equal to $64/34 \cdot CEF_{SO2}$, on the assumption that the H_2S forms SO_2.

[f] Assumed to be 62% pentane (C_5H_{12})and 38%hexane (C_6H_{14}) on a molar basis, the proportions assumed for pipeline natural gas.

[g] Calculated based on the assumed composition of dry gas, and the amount of material removed from raw gas. See the text.

[h] The Auto / Oil (1998) study shows the methane, ethane, nitrogen, and CO_2 content of "industry average" natural gas. The EIA's *Alternatives to Traditional Transportation Fuels* (1994) shows the propane, butane, pentane, and hexane content of "typical" gas.

[i] Hydrogen made from reforming natural gas.

TABLE 6. ENERGY USE OF MOTOR VEHICLES

A. FUEL ECONOMY PARAMETERS FOR BASELINE CONVENTIONAL PETROLEUM VEHICLES

Input parameters	**Gasoline**	**Diesel**
In-use fuel economy, conventional fuel, city (mpg)	25.0	5.5
In-use fuel economy, conventional fuel, highway (mpg)	38.0	7.0
Fraction of miles in city driving	0.55	0.67
Engine efficiency: brake-BTU / fuel-BTU	n.e.	0.28
Weight with no fuel or payload (lbs)	calculated[a]	25,000
Weight with payload and fuel (curb weight) (lbs)	calculated[b]	40,000
% decrease in mi / BTU per 1.0% increase in weight, city[c]	0.25 / 0.60	0.35
% decrease in mi / BTU per 1.0% increase in weight, highway[c]	0.25 / 0.45	0.35
Weight compounding factor (lb added structure per lb additional vehicle weight)[d]	0.10	same as gasoline
Calculated results		
In-use fuel economy on conventional fuel (mpg)	29.5	5.9
In-use fuel economy on reformulated fuel (mpg)	28.5	n.a.
In-use fuel economy of specified gasoline (R100 / Ox11) (mpg)	28.5	n.a.
In-use fuel economy, conventional fuel, city (mi / 10^6-BTU)	199.9	39.7
In-use fuel economy, conventional fuel, hwy (mi / 10^6-BTU)	303.8	50.5

n.e. = not estimated; n.a. = not applicable; Ox-- = percent oxygenate in reformulated gasoline; Rxx = percent reformulated gasoline in specified fuel mix.

[a] Calculated as the curb weight less the fuel and the payload.

[b] Calculated on the basis of a formula relating curb weight to the federal test-cycle 55 / 45 city / highway mpg:

$$CURB = 6675.54 - 167.57 \cdot FFE + 1.49 \cdot FFE^2$$

$$FFE = IUFE / IUR$$

where:

CURB = curb weight of vehicle (including fuel and 300-lb payload) (lbs)

FFE = the fuel economy over the federal test cycle (55/45 federal-city-cycle/federal-highway-cycle)
IUFE = the fuel economy in-use (55/45 in-use-city/in-use-highway; calculated from input in-use city fuel economy and in-use highway fuel economy)
IUR = the in-use deterioration factor: the ratio of the in-use 55/45 fuel economy to the federal test-cycle 55/45 fuel economy (assumed to be 0.8; see DeLuchi, 1991)

[c] The figure before the slash is for internal-combustion-engine vehicles (ICEVs); the figure after is for electric vehicles (EVs). Runs of Delucchi's (2000) detailed second-by-second simulation model of the energy use and performance of EVs and ICEVs indicate that a 1.0% increase in weight results in an 0.25% decrease in the fuel economy of ICEVs, and an 0.60% decrease in the fuel economy of EVs (Appendix G of this report). Yamane and Furuhama (1998) use a similar model to calculate the effect of weight on the fuel economy of ICEVs, and come up with a higher figure of about 0.7%decrease in fuel economy per 1.0% increase in weight.

[d] An analysis of the weight of an early electric vehicle, the ETX-1, reported in DeLuchi (1992), indicates a value of about 0.07. EEA (1998) suggest a rule of thumb that results in about 0.11. Berry and Aceves (1998) assume 0.30, and Maclean and Lave (1998) assume 0.50, but I believe that these are too high. I settle on 0.10.

B. MILE/BTU EFFICIENCY OF ALTERNATIVE-FUEL-VEHICLE POWERTRAINS RELATIVE TO THAT OF CONVENTIONAL PETROLEUM VEHICLES

AFVs vs. LDGVs	**Diesel**	**M100**	**CNG**	**LNG**	**CH_2**	**LH_2**	**E100**	**LPG**	**EV**[a]
City cycle MY 1995 (V_{TB})	1.20	1.10	1.05	1.06	1.10	1.12	1.10	1.05	part C
Highway cycle, MY 1995 (V_{TB})	1.20	1.10	1.05	1.06	1.10	1.12	1.10	1.05	part C
Input max. value (V_U) for city	1.20	1.25	1.20	1.20	1.30	1.30	1.25	1.20	part C
Input max. value (V_U) for highway	1.20	1.25	1.20	1.20	1.30	1.30	1.25	1.20	part C
City cycle k exp.	-0.030	0.024	0.024	0.024	0.024	0.03	0.024	0.024	part C
Highway k exp.	-0.030	0.024	0.024	0.024	0.024	0.03	0.024	0.024	part C

AFVs vs. HDDVs	**Gas**	**M100**	**CNG**	**LNG**	**CH_2**	**LH_2**	**E100**	**LPG**	**SD100**
City cycle, MY 1995 (V_{TB})	0.83	0.95	0.80	0.80	0.97	1.00	0.95	0.80	0.88
Highway cycle, MY 1995 (V_{TB})	0.83	0.95	0.85	0.85	1.02	1.05	0.95	0.85	0.88
Input max. value (V_U) for city cycle	0.83	1.05	0.95	0.95	1.10	1.10	1.05	0.95	0.95
Input max. value (V_U) for highway	0.83	1.05	1.00	1.00	1.10	1.10	1.05	1.00	0.95
City cycle k exp.	-0.030	-0.024	-0.03	-0.03	-0.03	-0.03	-0.024	-0.03	-0.030
Highway k exp.	-0.030	-0.024	-0.03	-0.03	-0.03	-0.03	-0.024	-0.03	-0.030

Gas = gasoline; SD 100 = 100% soy diesel; M100 = 100% methanol; CNG = compressed natural gas; LNG = liquefied natural gas; EV = electric vehicle; CH_2 = compressed hydrogen; LH_2 = liquefied hydrogen; E100 = 100% ethanol; E95 = 95% ethanol; E93 = 93% ethanol; LPG = liquefied petroleum gas (assume 95% propane, 5% butane); BD-20 = 20% biodiesel; LDGV = light-duty gasoline vehicle; HDDV = heavy-duty diesel vehicle; MY = model year.
V_U, V_{TB}, and k are parameters in Eq. 3 in the text. See the text for sources.

[a] The relative efficiency of EVs is calculated on the basis of the efficiency of the EV drivetrain and the efficiency of the LDGV powertrain. These parameters are shown in part C of this table. The calculated relative drivetrain efficiency is consistent with values calculated by a detailed energy-use model and summarized here in Table 7.

C. EFFICIENCY OF GASOLINE AND ELECTRIC POWER TRAINS (MI/BTU BASIS, HHVS)

	MY 1996		Max. value		Min. value		Exponent	
	City	Hwy	City	Hwy	City	Hwy	City	Hwy
LDGV powertrain	0.13	0.18	0.19	0.26	0.10	0.14	0.05	0.05
EV powertrain, w/regen.	0.72	0.76	0.92	0.91	0.50	0.56	0.05	0.05
EV powertrain, no regen.	0.65	0.75	0.80	0.88	0.48	0.56	0.05	0.05
Fuel cell	0.55	0.45	0.62	0.58	0.48	0.40	0.04	0.06
gasoline reformer	0.68	0.74	0.78	0.83	0.62	0.66	0.06	0.06
methanol reformer	0.70	0.76	0.82	0.88	0.66	0.70	0.06	0.06
ethanol reformer	0.68	0.74	0.78	0.83	0.62	0.66	0.06	0.06

HHV = higher heating value; MY = model year; Max. = maximum; Min. = minimum; exponent = “k” exponent in equation 3; City = city driving cycle; Hwy = highway driving cycle; LDGV = light-duty gasoline vehicle; EV = electric vehicle; regen. = regenerative braking; no regen. = no regenerative braking (for fuel-cell vehicles without electro-chemical energy storage).

See the text for discussion of parameter values.

TABLE 7. THE DRIVETRAIN EFFICIENCY OF EVS RELATIVE TO THAT OF GASOLINE ICEVS

	City cycle		**Highway cycle**	
	Escort	*Taurus*	*Escort*	*Taurus*
Long-term efficiency ratio	~ 8.0	~8.3	~ 5.2	~ 5.5
For reference: mpg of gasoline ICEV	27.4	20.2	41.3	32.5

As calculated from the detailed second-by-second energy use model documented in Delucchi et al. (2000). See the text for discussion.

TABLE 8. PROJECTIONS OF EV AND BATTERY PARAMETERS

	Baseline V_{TB} **(MY 1996)**	**Max.** V_U	**Min.** V_L	**Shape exponent k**
Weight reduction of EV powertrain, body, and chassis (lbs)[a]	275	500	150	0.060
Desired range of EV, city driving (mi)[b]	70	300	50	0.140
Efficiency of battery recharging[c]	0.85	0.95	0.70	0.120
Specific energy of battery (Wh / kg)[d]	45	200	30	0.180
Battery efficiency[d]	0.80	0.95	0.65	0.080
Battery cycle life[d]	300	2,000	200	0.160

The parametersV_U, V_L, V_{TB}, and k are those in the Eq. 3 in the text. MY = model year, which is not necessarily the same as the target year (see the discussion in the text).

[a] Parameter values based on a detailed EV design and performance model documented in Delucchi (2000a). These values include the extra weight of the chassis required to support the heavy EV batteries.

[b] I chose the parameter values so that the total battery weight stayed within reasonable limits, given the corresponding battery specific energy. Note that the shape parameter for the projection of driving range is slightly less than shape parameter for the projection of battery specific energy, which means that most -- but not all -- of the increase in battery energy density is translated directly into an increase in range. A small portion of the increase in specific energy results in a decrease in vehicle weight.

[c] These parameter values reflect the anticipated improvement in EV controllers.

[d] These estimates are based on the battery data summarized in Table 9. The 1996 parameter values (V_{TB}) are those of commercially available Pb / acid batteries. The maximum parameter values (V_U) are those projected for lithium batteries.

TABLE 9. EV BATTERIES: PRESENT AND FUTURE CHARACTERISTICS

Battery type	**Specific energy** (Wh/kg)	**Specific volume** (Wh/l)	**Specific power** (W/kg)	**Energy efficiency** (%)	**Cycle life**
Commercial Pb/acid (1992-1995)	25-40	60-90	50-150	70-85	150-400
Advanced Pb/acid (2000+)	50	100+	300+		500+
Ni-metal hydride (1992-1995)	50-60	150-210	100-200	70-80	400-1000
Future Ni-metal hydride (2000+)	90+	190+	200+		1000+
Lithium ion (2000+)	100+	150+	300+	95+	1200+
Lithium polymer (long term)	200+		200+		

Sources: Delucchi (2000a); Burke (1995), DeLuca et al. (1992), Dickinson et al. (1994), U. S. DOE (1995b), Kalhammer et al. (1995), Moore (1996), OTA (1995), Ovshinsky et al. (1993).

TABLE 10. FUEL STORAGE, WEIGHT, AND RANGE OF ALTERNATIVE-FUEL-VEHICLES

	Gas	Dsl.	SD100	M100	CNG	LNG	CH_2	LH_2	E100	LPG
Range, LDVs (mi)[a]	380	456	n.a.	300	250	300	200	250	350	300
Range, HDVs (mi)[a]	510	600	600	500	450	500	350	450	550	550
Difference in powertrain and body weight, LDVs[b]	base-line	100	n.a.	0	0	0	0	0	0	0
Difference in powertrain and body weight, HDVs[b]	-200	base-line	0	-200	-200	-200	-200	-200	-200	-200
Lb storage system per lb fuel, LDVs[c]	0.40	0.40	n.a.	0.36	4.39	1.60	25.0	6.2	0.36	1.33
Lb storage system per lb fuel, HDVs[c]	0.18	0.18	0.18	0.16	3.14	1.15	20.0	4.0	0.16	0.75

Gas = gasoline; Dsl = diesel fuel; SD100 = 100% soy diesel; M100 = 100% methanol; CNG = compressed natural gas; LNG = liquefied natural gas; CH_2 = compressed hydrogen; LH_2 = liquefied hydrogen; E100 = 100% methanol; LPG = liquefied petroleum gases; “baseline” = the vehicle with respect to which the difference is estimated. Note that for EVs, the attributes are estimated with Eq. 3 and the parameter values shown in Table 8 .

[a] These estimates are my judgment, based on economic and technical attributes of storage systems, and driving requirements of light-duty and heavy-duty vehicles.

[b] I assume that in an LDV, a diesel CI engine weighs 100 lbs more than a SI engine, and that in an HDV, a diesel CI engine weighs 200 lbs more than a SI engine (Energy and Environmental Analysis [1991] reports that mid-size 1987 diesel passenger cars weigh 125 lbs more than their gasoline counterparts.). Otherwise, I assume that the body and powertrain in all ICEVs weigh the same. Note that these estimates here do not include any extra weight of chassis and suspension needed to support any extra weight of fuel-storage systems. That extra weight is estimated separately.

[c] Gasoline and diesel: From DeLuchi (1991) Table 2.
SD100: Assume values for oil (gasoline or diesel).
M100: Minor revision to values in DeLuchi (1991) Table 2.

CNG: Value for LDVs based on data in Richards et al. (1996); see the discussion in the text. Value for HDVs equal to value for LDVs multiplied by 3.14/4.39, which is the ratio of the HDV lb/lb to the LDV lb/lb for CNG tanks in DeLuchi (1991) Table 2.
LNG: Minor revisions to DeLuchi (1991) Table 2 values, on the basis of data reported in Powars et al. (1994): 1.29 for an early-generation 18-gallon tank by Beech, 1.76 for an early-generation 18-gallon tank by Essex, and 1.26 for a recent 48-gallon tank by Essex. These values probably do not include all relevant hardware.
CH_2: Value for LDVs based on data in Berry and Aceves (1998); see the discussion in the text. Value for HDVs equal to value for LDVs multiplied by 3.14/4.39, which is the ratio of the HDV lb/lb to the LDV lb/lb for CNG tanks in DeLuchi (1991) Table 2.
LH_2: DeLuchi (1991) Table 2 reports 5.7 lbs/lb for LDV tanks, and 3.6 lbs/lb for HDV tanks. Berry and Aceves (1998) report a value of 6.3 lbs/lb, from a 1996 DOE study of onboard hydrogen storage systems; Ewald (1998) reports a value of 6.5 lbs/lb, and Chalk et al. (1998) report 5.2. It is likely that advanced hydrogen storage tanks will be closer to the values of DeLuchi (1991) and Chalk et al. (1998). For example, Wetzel (1998) describes a recently developed LH_2 refueling system that eliminates the need for a gaseous hydrogen line and a solenoid cryovalve and associated controls and wiring.
E100: Minor revision to values in DeLuchi (1991) Table 2.
LPG: From DeLuchi (1991) Table 2.

TABLE 11. BLANK

TABLE 12. EMISSIONS FROM PETROLEUM AND ALTERNATIVE-FUEL VEHICLES: INPUT DATA

	LDGV emissions deterioration[a]					**LDGV zero-mile emissions**[a]				
	V_L	V_U	V_B	T_B	k	V_L	V_U	V_B	T_B	k
Fuel evap.[d]	0.010	0.100	0.030	2000	-0.08	0.050	8.000	0.300	2000	-0.08
NMOC exhaust	0.015	0.400	0.070	2000	-0.08	0.033	8.000	0.240	2000	-0.08
CH_4 exhaust	0.001	0.020	0.002	2000	-0.06	0.010	0.500	0.040	2000	-0.06
CO exhaust	0.100	2.500	0.900	2000	-0.08	0.152	80.00	3.800	2000	-0.06
N_2O, MY=2005	0.000	0.010	0.008	1985	0.40 -	0.000	0.065	0.060	1985	0.40 -
N_2O, MY> 2005	.0016	0.010	0.006	2015	0.20	0.024	0.065	0.040	2015	0.20
NO_2 exhaust	0.008	0.180	0.090	2000	-0.07	0.050	6.000	0.360	2000	-0.07
PM exhaust[i]	0.001	0.008	0.003	2000	-0.05	0.003	0.800	0.012	2000	-0.05

	AFV emissions relative to conventional gas[b]						
	R100	*diesel*	*M100*	*CNG*	*CH2*[c]	*E100*	*LPG*
Fuel evap.[d]	0.85	0.05	calc.	calc.	calc.	calc.	calc.
NMOC exhaust	0.70	0.50	0.90	calc.[e]	calc.	0.90	0.50
CH_4 exhaust	1.00	0.50	0.50	15.00	calc.	1.50	1.00
CO exhaust	0.80	0.20	0.60	0.60	calc.	0.60	0.60
N_2O exhaust	1.00	0.25	1.00	0.75[f]	0.00	1.00	1.00
NO_2 exhaust	0.85	1.50	0.90	0.90	0.90	0.90	0.90
PM exhaust[i]	1.00	10.00	0.40	0.20	calc.	0.40	0.25
lube oil[g]	1.00	1.00	1.00	0.50	1.00	1.00	0.75

TABLE 12, CONTINUED.

	HDDV parameters[a]				Emissions relative to diesel[b]						
	EM	*DR*	*?ZM*	*ZM*	*F-T D*	*SD100*	*M100*	*CNG*	*CH2*[c]	*E100*	*LPG*
NMOC exh.	0.92	0.006	-1.00	0.80	0.81	0.30	2.00	2.50	calc.	2.00	2.50
CH_4 exh.	0.032	0.0001	-0.50	0.03	0.90	0.30	1.00	30.00	calc.	3.00	1.00
CO exh.	5.46	0.030	-1.00	5.00	0.65	0.50	1.30	0.10	calc.	1.30	0.10
N_2O exh.	0.022	0.000	0.00	0.02	1.00	1.00	1.00	1.00	0.95	1.00	1.00
NO_2 exh.	6.60	0.000	-5.00	6.00	0.95	1.10	0.50	0.50	0.50	0.50	0.50
PM exhaust[i]	0.21	0.001	-8.00	0.50	0.76	0.50	0.20	calc.[h]	calc.	0.30	calc.[h]
lube oil[g]	6.45	n.a.	n.a.	6.45	1.00	1.50	1.00	0.50	1.00	1.00	0.75

LDGV = light-duty gasoline vehicle; HDDV = heavy-duty gasoline vehicle; R100 = 100% reformulated gasoline; SD100 = 100% soy diesel; F-T D = Fischer-Tropsch diesel; M100 = 100% methanol; NG = natural gas; H_2 = hydrogen; E100 = 100% ethanol; LPG = liquefied petroleum gas (assume 95% propane, 5% butane); V_L, V_U, V_B, T_B , k = parameters in Eq. 3 (see the text); EM = emissions in target year (g / mi for LDVs, g / bhp-hr for HDVs); DR = the deterioration rate in emissions (g / mi / 10,000-mi for LDVs; g / bhp-hr / 10,00-mi for HDVs); ?ZM = the annual percentage change in the zero-mile emission rate, with each new model year; ZM = the zero-mile emission rate from a base-model-year vehicle (g / bhp-hr for HDVs); evap. = evaporative; exh. = exhaust; calc. = calculated (not input directly; see the text or Appendix B of DeLuchi [1993]).

[a] See the discussion in the text.

[b] Based on estimates cited in Appendix B of DeLuchi (1993), and other literature published since then. See the discussion in the text and other notes to this table for details. See Appendix F of this report for analysis of CH_4 and N_2O emissions database.

[c] Because hydrogen fuel contains essentially no carbon, I assume that exhaust emissions of carbon-containing compounds (CH_4, NMOC, CO, and PM) from hydrogen vehicles come from combustion of lubricating oil. Therefore, I calculate emissions of these species from hydrogen vehicles as: the product of the emission rate due to oil combustion in a gasoline vehicle without a catalyst, and the relative oil consumption rate for hydrogen vehicles. I use the emission rate for non-catalyst equipped vehicles because I assume that hydrogen ICEVs will not have a catalytic converter.

[d] Resting-loss, running-loss, hot-soak, and diurnal emissions from ambient-temperature liquid fuels; boil-off of cryogenic fuels, and leakage of gaseous fuels. Emissions from vehicle refueling are included in the "fuel dispensing" stage. Emissions from trucks refilling the tanks at service stations, and from other upstream activities, are included in the "fuel distribution" stage.

In Appendix B of DeLuchi (1993), I assumed that evaporative emissions of methanol and ethanol relative to evaporative emissions of gasoline are proportional to the relative volatility of the pure fuel, which is quite low. However, I also acknowledged that factors other than relative volatility determine evaporative emissions. Recent tests of evaporative emissions from M85 (Kelly et al., 1996a) and E85 (Kelly et al., 1996b; Baudino et al., 1993) indicate that evaporative emissions of ethanol and methanol can be quite variable, and in some cases greater than evaporative emissions of reformulated gasoline. I now assume that g/gal evaporative emissions of methanol are 60% of g/gal evaporative emissions of conventional gasoline, and that g/gal evaporative emissions of ethanol are 40% of g/gal evaporative emissions of conventional gasoline.

See the text for explanation of calculated values for gaseous-fuel vehicles.

[e] The Auto/Oil Study (1996) measured emissions from NGVs using gas of four different compositions: 86%, 90%, 94%, and 97% methane. Emissions of CH_4, CO, and NO_X were not related to the methane content of the gas, but emissions of NMHC were. Emissions tests presented in Bevilacqua (1997) also show that emissions of CO and NO_X from dedicated NGVs are not related to the methane content of the gas, but that NMOG emissions, in some cases, decrease with increasing methane content.

On the basis of the Auto/Oil study, and other emission tests, I estimate the following relationship between NMHC emissions from NGVs as a fraction of NMOCs from gasoline vehicles, and the methane content of the gas:

$$\text{NMHC, relative to gasoline} = 1.90 - 1.794 \cdot \text{methane fraction}$$

[f] The sparse available data (e.g., Battelle, 1995; see Appendix F of this report) indicate that CNGVs have a slightly lower emission rate; therefore, I have assumed that N_2O emissions from CNGVs are 75% of N_2O emissions from gasoline LDVs.

[g] The input zero-mile lube-oil consumption rates (ZM) for the diesel and gasoline vehicle do not include any adjustments (deductions) for oil that is recycled or disposed improperly. Emissions related to the production lifecycle are accounted elsewhere. See the text for details.

The relative rate for AFVs is meant to be the g/mi oil consumption of the AFV relative to the g/mi oil consumption of the petroleum vehicle.

[h] The test data cited in the text indicate that emissions of PM from heavy-duty CNG and LPG vehicles are not really related to PM emissions from diesel counterparts, but rather tend to be fixed around 0.01 g/bhp-hr. This most likely is because PM emissions from gaseous-fuel vehicles are inherently low, on account of the gaseous state and simple chemical structure of the fuel. Therefore, I assume that PM emissions from HD CNGVs are equal to $0.004+0.02 \cdot PM_{diesel}$, and that PM emissions from HD LPG vehicles are equal to $0.006+0.02 \cdot PM_{diesel}$, in g/bhp-hr.

[i] For the purpose of CO_2-equivalency calculations, exhaust PM is assumed to comprise black carbon and organic matter in the proportions shown in Table 41. See Appendix D for discussion of CEFs

TABLE 13. ANNUAL VMT AND SURVIVAL PROBABILITY AS A FUNCTION OF AGE, FOR THE REFERENCE MODEL YEAR (1990) VEHICLE

Annual VMT[a]			Survival probability[b]			Age (yrs)
LDVs	*LDTs*	*HDTs*	*LDVs*	*LDTs*	*HDTs*	
0	0	0	1.000	1.000	1.000	0
13,100	13,700	76,000	0.995	0.998	0.994	1
12,995	13,590	72,200	0.987	0.994	0.986	2
12,801	13,380	67,756	0.977	0.988	0.976	3
12,526	13,076	63,045	0.963	0.979	0.964	4
12,177	12,686	58,267	0.943	0.967	0.947	5
11,762	12,218	53,546	0.920	0.948	0.926	6
11,290	11,682	48,963	0.890	0.924	0.898	7
10,770	11,088	44,574	0.853	0.892	0.863	8
10,210	10,448	40,415	0.807	0.852	0.822	9
9,620	9,774	36,508	0.754	0.806	0.774	10
9,009	9,078	32,866	0.692	0.755	0.722	11
8,385	8,369	29,492	0.625	0.702	0.669	12
7,757	7,659	26,385	0.554	0.649	0.616	13
7,133	6,959	23,537	0.481	0.597	0.565	14
6,519	6,276	20,939	0.409	0.548	0.516	15
5,923	5,618	18,580	0.341	0.502	0.471	16
5,348	4,992	16,446	0.278	0.459	0.429	17
4,800	4,403	14,522	0.223	0.419	0.390	18
4,282	3,854	12,794	0.176	0.383	0.355	19
3,798	3,348	11,247	0.137	0.349	0.322	20
3,347	2,887	9,865	0.100	0.319	0.293	21
2,932	2,471	8,636	0.070	0.291	0.266	22
2,554	2,098	7,544	0.040	0.265	0.242	23
2,210	1,768	6,578	0.010	0.242	0.219	24
1,901	1,478	5,725	0.000	0.220	0.199	25
1,626	1,227	4,973	0.000	0.190	0.171	26
1,382	1,010	4,312	0.000	0.150	0.135	27
1,167	825	3,733	0.000	0.110	0.098	28
980	668	3,225	0.000	0.060	0.053	29
817	537	2,783	0.000	0.020	0.018	30

[a] The mileage driven by the end of the indicated year of age. The values are estimated as follows:

- MY 1990 LDVs and LDGTs, age 1: The *Residential Transportation Energy Consumption Survey (RTECS)* of the EIA reports the average annual mileage per household vehicle, according to odometer readings. Davis (1998) summarizes the fleet (car-and-truck)-average annual mileage rate of vehicles 1-year old or less in the five years that the RTECS has been done:

1983	*1985*	*1988*	*1991*	*1994*
13,400	12,700	12,900	13,400	15,220

RTECS data indicate that new household light trucks are driven ever so slightly more than are new household passenger cars. Using data from a special tabulation from the 1988 *RTECS* public use tape (Davis, 1992), I calculate that in 1988, a 1-year old household passenger car traveled 12,764 miles, and a 1-year-old household light truck traveled13,345 miles. The household fleet average was 12,910 miles, consistent with the 12,900 shown in the mini-table above. The *RTECS* data for 1991 indicate a smaller gap between cars and light trucks, but the data for 1994 indicate a slightly larger gap (about 1,000 miles/year) (EIA, *Household Vehicles Energy Consumption 1994*, 1997; *Household Vehicles Energy Consumption 1991*, 1993). The EPA (1993, p. G-1) assumes that in 1990, a1990 model-year (MY) LDGV travels13,118, and a 1990 MY LDGT 15,640 miles. Given all of these data, I assume that a 1-year old 1990 MY LDV travels 13,100 miles, and a 1990 MY LDT 13,700 miles.

- MY 1990 HDTs, age 1: The EPA (1993, p. G-1 and G-2) assumes that in 1990, a 1990 MY class VIIIA diesel truck travels 43,946 miles, and a 1990 MY class VIIIB diesel truck 86,375 miles. Data from the 1992 *TIUS* (Bureau of the Census, 1995) indicate that a MY 1992 truck traveled 77,142 miles in 1992. I assume that a 1-year old MY 1990 heavy truck travels 76,000 miles.

- all MY vehicles, ages 2-30: I estimate annual VMT by age, for a particular model year, with the following equation:

$$VMT_{MY,A} = VMT_{MY,A-1} \cdot (1-K \cdot (A-1)^{C})$$

where:

$VMT_{MY,A}$ = annual VMT by model year MY at vehicle age A
A = the age of a particular model-year vehicle
K, C are assumed to be as follows:

	LDVs	LDTs	HDTs
K	0.0085	0.0085	0.055
C	0.95	1.00	0.35

The values of K and C for LDVs and LDTs result in a yearly decline in VMT with age that is consistent with the patterns that can be inferred from the five *RTECS* (Davis, 1998).

Note that I assume that the mileage traveled in the first year changes from one model year to the next. Data from the RTECs (Davis, 1998), summarized in the mini-table above, indicate that the annual mileage of 1-year old vehicles has increased on the order of 1% with each new model year. There is some indication that the rate of change is slightly greater for light trucks. I assume the following rates of change in the annual mileage of 1-year-old vehicles, with respect to the assumed annual mileage of 1-year old 1990 MY vehicles (explained above): LDVs, 0.65% / MY; LDTs, 0.70% / MY; HDTs, 0.50% / MY.

[b] The *Transportation Energy Data Book* (Davis, 1998) reports the survival probability over 20 years of model-year 1970, 1980, and 1990 automobiles, the survival probability over 25 years of light trucks registered between 1978 and 1989, and the survival probability over 25 years of all trucks registered in 1966 to 1973, 1973 to 1978, and 1978 to 1989. Given these data, my estimates for a model-year 1990 are as follows:

LDVs	LDTs	HDDVs
0-20 years: Davis (1998) data for MY 1990 autos	0-25 years: Davis (1998) data for 1978-1989 LDTs	all years: $LDTS_a \cdot 0.996^A$, where $LDTS_a$ is the LDT survival at age A, and the exponent A is the age
21-30 years: my estimates	26-30 years: my estimates	

I specified the formula for estimating the HDDV survival probability on the basis of the difference between the reported survival probability for 1978-1989 light trucks and the reported survival probability for all 1978-1989 truck (Davis, 1998).

The ORNL data indicate that the survival probabilities increase with model year. I estimate the survival probabilities for model years other than 1990, relative to the probabilities for MY 1990:

$$SP_{MY,A} = SP_{1990,A}^{(1-(MY-1990)/K)}$$

where:

$SP_{MY,A}$ = survival probability of model year MY at vehicle age A
$SP_{1990,A}$ = survival probability of model year 1990 at vehicle age A
MY = model year
K is assumed to be as follows:

LDVs	LDTs	HDTs
1.55	1.65	1.65

These parameter values result in survival probability schedules consistent with those reported by Davis for 1970 and 1980 MY LDVs.

TABLE 14. EMISSIONS FROM REFINERY PROCESS-AREA, YEAR 2000 (GRAMS-POLLUTANT/10^6-BTU-PRODUCT OUTPUT)

	CG	RFG	dist.	ULSD	fuel oil	LPG
Fuel evap. or leakage	10.6	11.2	9.1	9.1	8.1	14.4
NMOC exhaust	0.0	0.0	0.0	0.0	0.0	0.0
Evap. + NMOC exhaust	10.6	11.2	9.1	9.1	8.1	14.4
C in evap. + NMOC exhaust	8.8	9.3	7.5	7.5	6.7	11.9
Ozone-weighted total NMOC	8.0	8.4	6.8	6.8	6.1	10.8
CH_4 (exhaust)	2.1	2.2	1.8	1.8	1.5	2.5
CO	11.2	13.3	6.8	6.8	0.3	0.4
N_20	0.6	0.7	0.4	0.4	0.2	0.1
NO_x (NO_2)	2.2	2.4	1.8	1.8	1.2	1.9
SO_x (SO_2)	7.7	7.0	4.8	8.1	1.5	7.9
PM (combustion-like) (a)	0.9	1.1	0.5	0.5	0.0	0.0
PM_{10} (combustion-like) (a)	0.9	1.1	0.5	0.5	0.0	0.0
$PM_{2.5}$ (combustion-like) (a)	n.e.	n.e.	n.e.	n.e.	n.e.	n.e.
CO_2 from emission control	994.5	1162.6	641.9	641.9	122.6	183.6

CFG = conventional gasoline; RFG = reformulated gasoline; dist. = distillate fuel; ULSD = ultra-low-sulfur distillate fuel; LPG = liquefied petroleum gases; evap. = evaporative; C = carbon. See the text for details of estimation.

[a] For the purpose of calculating CO_2-equivalent emissions, the LEM has CEFs for black carbon (BC) aerosols from combustion, organic-matter (OM) aerosol from combustion, and dust (which generally comprises earth-crustal material) (Appendix D). Thus, in order to be able to calculate CO_2-equivalent emissions, it is necessary to classify all PM emissions as either dust-like or from combustion (or combustion-like) processes. Lacking data to the contrary, I have assumed that all PM emissions from refinery process areas are combustion-like and comprise BC and OM in the proportions shown in Table 41.

TABLE 14A. COMPARISON OF GM ET AL. (2002C) AND LEM ESTIMATES OF REFINERY EMISSIONS (G-POLLUTANT/KG-FUEL)

	GM et al. (2002c)[a]				LEM[b]	
	Diesel		**Gasoline**		**Diesel**	**Gasoline**
	low	*high*	*low*	*high*		
CO_2	134.6	707.9	259.6	901.5	263.4	553.9
CH_4	0.000	0.090	0.000	0.100	0.695	2.045
N_2O	0.000	0.000	0.000	0.000	0.022	0.013
SO_x	0.133	0.247	0.164	0.291	0.574	0.566
NO_x	0.092	0.125	0.146	0.184	0.491	1.124
CO	0.055	0.216	0.105	0.286	0.408	0.562
NMOVs	0.000	0.000	0.000	0.000	0.182	0.127
PM[c]	0.007	0.008	0.009	0.012	0.096	0.158

[a] GM et al. (2002c) report emissions in g/kWh-fuel, LHV basis (Tables 37-40 in their report). I convert to g/kg-fuel using their figure of 11.8 kWh/kg for gasoline (LHV basis), and my estimate of 11.7 kWh/kg for diesel fuel (LHV basis).

GM et al. (2002c) represent a refinery as series of linked input/output processes. One set of linked processes represents a “gasoline” refinery, and another set a “diesel” refinery. Emissions estimates for the processes are based on a 1999 report which apparently uses data from a modern refinery built in Germany in 1997.

The “low” estimates assume 0.3% sulfur in crude oil, no partial oxidation plant (visbreaker residue sold as a fuel). The “high” estimates assume 1.6% sulfur in crude oil, and a partial oxidation plant to convert the visbreaker residue to hydrogen for use in the refinery.

[b] I ran the LEM for the U. S. in the year 2010, and converted the results from g/10^6-BTU to g/kg-fuel. The results shown are for the “fuel production” stage of the lifecycle of gasoline and diesel fuel. This includes emissions from refinery boilers, emissions from refinery process areas, fugitive emissions, and emissions from the generation of electricity used by refineries.

The sulfur content of crude oil used in the U. S. in 2010 is projected in the LEM to be about 1.3%, in between the “low” and the “high” cases of GM et al. (2002c), but closer to the “high” case. In the LEM, average refinery emissions in the year 2010 are similar to emissions from a new refinery in 1995.

[c] This is given as “dust” in GM et al. (2002c).

TABLE 15. MIX OF ELECTRIC POWER USED TO RECHARGE ELECTRIC VEHICLES

EPRI Region	**GWh/d**	**Percentage of recharging power from:**					
Weekday[a]		*Coal*	*Oil*	*Gas*	*Nuclear*	*Other*	
Northeast	24.54	33.6	62.9	3.5	0.0	0.0	
East Central	8.681	100.0	0.0	0.0	0.0	0.0	
Southeast	13.626	86.8	5.9	7.3	0.0	0.0	
West Central	10.292	100.0	0.0	0.0	0.0	0.0	
South Central	4.972	25.5	0.0	74.5	0.0	0.0	
West	24.51	51.7	15.2	33.1	0.0	0.0	
United States	86.621	61.2	23.0	15.8	0.0	0.0	
Weekend[a]							
Northeast	11.658	41.3	46.6	1	11.1	0	
East Central	4.124	100	0	0	0	0	
Southeast	6.474	100	0	0	0	0	
West Central	4.89	96.8	0	0	3.2	0	
South Central	2.362	25.7	0	74.3	0	0	
West	11.644	64.9	0.9	34.2	0	0	
United States	41.152	68.8	13.5	14.2	3.5	0.0	
All days[b]		*Coal*	*Oil*	*Gas boiler*[c]	*Gas turbine*[c]	*Nuclear*	*Other*
Northeast	36.198	36.1	57.7	1.6	1.1	3.6	0.0
East Central	12.805	100.0	0.0	0.0	0.0	0.0	0.0
Southeast	20.1	91.1	4.0	2.9	2.1	0.0	0.0
West Central	15.182	99.0	0.0	0.0	0.0	1.0	0.0
South Central	7.334	25.6	0.0	43.4	31.1	0.0	0.0
West	36.154	56.0	10.6	19.5	14.0	0.0	0.0
United States	127.773	63.6	20.0	8.9	6.4	1.1	0.0

[a] The weekday and weekend GWh/day results are the 101-City, year-2010, Case-B scenario of Yao et al. (1993). For the purpose of calculating the all-days power mix

(bottom part of this table), the choice of year and scenario from the Yao et al. (1993) analysis doesn't matter; the results will be same in any case. (Also, in the Yao et al. analysis, the marginal power mix apparently is the same for all years and all scenarios.)

[b] Calculated from the data on GWh / day and recharging mix for weekdays and weekends.

[c] Split of gas between boilers and turbines is my estimate for the year 2015.

TABLE 16. CH_4 AND N_2O EMISSION FACTORS FOR UTILITY BOILERS(G/10^6-BTU-FUEL).

	CH_4			N_2O		
	IPCC generic	*AP-42, utilities*	*assumed here*	*IPCC generic*	*AP-42, utilities*	*assumed here*
Coal	1.1	0.2 - 1.4[b]	0.9	1.5	0.7 - 2.1[b]	0.9
Oil	3.2	0.8	0.8	0.6	0.3	0.3
NG	1.1	1.0	1.0	0.1	0.3, 1.0[c]	0.6
wood	32	n.e.	0.04	4.2	n.e.	0.02

Sources: “IPCC generic”: In its “simple” guidelines, the IPCC (1997) uses its judgment to “average” across fuel and boiler varieties and establish generic emission factors for the use of coal, oil, or gas, in what it refers to as the “energy industry,” which includes much more than electric utilities. “AP-42, utilities”: the EPA’s emission factors specifically for electric-utility boilers. NG = natural gas.

[b] Depends on the type of fuel used and the firing configuration.

[c] The lower figure applies to low-NO_X combustion, the higher to uncontrolled boilers.

TABLE 17. FEEDSTOCK AND PROCESS ENERGY REQUIREMENTS OF ALTERNATIVE-FUEL PRODUCTION PLANTS

Fuel -->	**FTD100**	**H_2**	**oil**	**M100**	**M100**	**M100**
Feedstock -->	NG	NG	coal	NG	coal	wood
Base year -->	1994	1996	2000	1994	1994	1994
Inputs (below) per unit of output (to right)	*gallons*	*10^6 BTU*	*gallons*	*gallons*	*gallons*	*gallons*
Electricity (kWh) (base yr.)[a]	0.00	8.50	0.00	0.06	0.00	0.61
Percent change per year	0.20	-0.20	0.00	-0.60	0.00	1.50
Calculated value in 2020	0.00	8.10	0.00	0.05	0.00	0.90
Diesel (gallons) (base yr.)	0.000	0.070	0.010	0.000	0.010	0.016
Percent change per year	-0.30	-0.30	-0.30	-0.30	-0.30	-0.30
Calculated value in 2020	0.00	0.07	0.01	0.00	0.01	0.01
Natural gas (SCF) (base yr.)	223.90	1,062	0.00	100.00	0.00	0.00
Percent change per year	-0.60	-0.20	0.00	-0.50	0.00	0.00
Calculated value in 2020	191.47	1,013	0.00	87.78	0.00	0.00
Coal (lbs) (base yr.)	0.00	0.00	30.00	0.00	10.50	0.00
Percent change per year	0.00	0.00	-0.50	0.00	-0.50	0.00
Calculated value in 2020	0.00	0.00	27.14	0.00	9.22	0.00
Wood, grass, crop residue (lbs) (base yr.)	n.a.	0.00	n.a.	n.a.	n.a.	16.00
Percent change per year	n.a.	-0.25	n.a.	n.a.	n.a.	-1.00
Calculated value in 2020	n.a.	0.00	n.a.	n.a.	n.a.	12.32
Crop (bu) (base yr.)	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Percent change per year	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Calculated value in 2020	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Total 10^6 BTUs input per unit output, year 2020	200,049	1,094,600	260,742	91,882	89,395	107,995
Ratio of total energy to fuel output energy	1.57	1.09	1.89	1.47	1.39	1.67
Ratio of feedstock energy to fuel output energy[b]	1.53	1.06	1.88	1.42	1.37	1.60

TABLE 17, CONTINUED.

Fuel -->	**E100**	**E100**	**E100**	**SD100**	**SG**
Feedstock -->	corn	grass	wood	soy	wood
Base year -->	1996	2000	2000	1994	1994
Inputs (below) per unit of output (to right)	*gal*	*gal*	*gal*	*gal*	*10^6 BTU*
Electricity (kWh) (base yr.)[a]	1.15	0.00	0.00	2.92	8.79
Percent change per year	-0.60	0.00	0.00	-0.80	-1.40
Calculated value in 2020	1.00	0.00	0.00	2.37	6.09
Diesel (gallons) (base yr.)	0.010	0.008	0.016	0.010	0.070
Percent change per year	-0.30	-0.30	-0.30	-0.30	-0.30
Calculated value in 2020	0.009	0.008	0.015	0.009	0.06
Natural gas (SCF) (base yr.)	22.45	0.00	0.00	46.37	0.00
Percent change per year	-0.30	0.00	0.00	-0.90	0.00
Calculated value in 2020	23.26	0.00	0.00	36.66	0.00
Coal (lbs) (base yr.)	1.62	0.00	0.00	0.00	0.00
Percent change per year	-0.40	0.00	0.00	-1.00	0.00
Calculated value in 2020	1.64	0.00	0.00	0.00	0.00
Wood, grass, crop residue (lbs) (base yr.)	0.00	28.09	29.57	0.00	167.66
Percent change per year	0.00	-1.65	-1.65	0.00	-0.25
Calculated value in 2020	0.00	20.14	21.20	0.00	157.10
Crop (bu) (base yr.)	0.385	n.a.	n.a.	0.679	n.a.
Percent change per year	-0.20	n.a.	n.a.	-0.20	n.a.
Calculated value in 2020	0.367	n.a.	n.a.	0.645	n.a.
Total 10^6 BTUs input per unit output, year 2020	41,537	152,096	179,111	47,674	1,341,560
Ratio of total energy to fuel output energy	0.49	1.80	2.12	0.36	1.34
Ratio of feedstock energy to fuel output energy[b]	0.00	1.79	2.09	0.00	1.31

FTD100 = 100% diesel fuel from natural gas via the Fischer-Tropsch process; NG = natural gas; M100 = 100% methanol; E100 = 100% ethanol; SD100 = 100% soy-derived diesel; SG = synthetic gas; n.a. = not applicable.

[a] This is purchased power only. Excess power marketed to the grid is not shown here, but is of course taken into account in the analysis.

[b] In the case of corn and soybeans, this is bushels per BTU.

Basis of estimates:

F-T diesel from NG: See the discussion in the text.

Oil from coal: See the discussion in the text.

Hydrogen from NG: See the discussion in the text.

Methanol/NG and methanol/coal: For 1994, I assume values for current technology (Tables J.1, J.3, and J.4). I pick the %/change per year so that by 2020 the resultant energy-use values approach those estimated for advanced technologies (Tables J.1, J.3, and J.4).

Methanol/wood: see the discussion in the text.

Ethanol/corn. See the discussion in the text.

Ethanol/wood. See the discussion in the text.

Ethanol/grass. Riley and Schell (1992) estimate inputs and outputs for grass-to-ethanol and wood-to-ethanol plants. I multiply my wood-to-ethanol parameter values by the grass/wood input/output ratios from Riley and Schell (1992).

Biodiesel/soybeans. My estimates are based on the input/output data reviewed in Appendix A to this report. I assume that biodiesel plants will use natural gas rather than coal to provide process heat and steam, because it is easier to meet emission requirements with natural gas.

SNG/wood: For 1994, I assume values towards the high end of the range of Table K.11 of DeLuchi (1991). I pick the %/change per year so that by 2015 the resultant energy-use values approach those at the lower end of the range of Table K.11.

Note that in all cases, I have included a small amount of diesel fuel input, as the energy used by loading and delivery trucks at the plant.

In some cases, the percentage change per year shown applies only through the year 2020. See the text for further discussion.

TABLE 18. ESTIMATES OF EMISSIONS FACTORS FOR ALCOHOL FUEL PRODUCTION PLANTS

A. GRAMS PER 10^6-BTU FEEDSTOCK INPUT TO PLANT OR 10^6-BTU FUEL INPUT TO BOILER

Reference	Product/process	Feedstock	HCs	CO	NO_x	SO_x	PM	CH_4
Intech (1990)	methanol / steam reforming plant	NG	0.2	1.4	68.7[a]	n.e.	n.e.	
Mueller (1990)	methanol / steam reforming plant	NG	0.2	5.5	53.7[a]	n.e.	n.e.	
Heath (1991)[b]	methanol / steam reforming plant	NG	neg.	25.6	30.9	neg.	neg.	
Ecotraffic AB (1992)[c]	methanol / steam reforming plant	NG	15.0	15.0	82.4[a]	<1.0	n.e.	3.0
Darrow (1994)[d]	methanol / steam reforming plant	NG	0.3	3.6	3 / 14.3	0.1	0.1	
Texas Air Board (1990)[e]	methanol / steam reforming plant	NG	n.e.	n.e.	n.e.	n.e.	n.e.	1-10
IPCC (1997)[f]	methanol / steam reforming plant	NG	n.e.	n.e.	n.e.	n.e.	n.e.	60
U.S. DOE (1988)	steam / uncontrolled industrial boiler	wood	16-118	15-2000	6-104	0.5-24	104-1360	n.e.
U.S. DOE (1988)	steam / controlled industrial boiler	wood	n.e.	n.e.	n.e.	n.e.	23	n.e.
Tellus (1993)[g]	steam / boilers	ag. wastes	n.e.	1710	90	n.e.	n.e.	15
AP-42 (EPA, 1995)	steam / stoker boilers	wood & waste	8.2[h]	272	100, 222[i]	4[j]	25-182[k]	9.5 [h]
AP-42 (EPA, 1995)	steam / fluidized-bed combustion	wood & waste	n.e.	77	n.e.	n.e.	n.e.	n.e.
Ismail and Quick (1991)[l]	steam / fluidized-bed combustion	wood	128	30	37	n.e.	n.e.	n.e.

Notes: see next page.

n.e. = not estimated, NG = natural gas.

[a] These NO_X emission factors seem relatively high. However, the high temperature of steam reforming, about 1500° F, could cause relatively high NO_X emissions. Low-temperature processes, or processes using pure oxygen, would have lower emissions.

[b] Heath (1991) cites a 1989 study that estimates emissions from eight different sources in a methanol plant. Her estimates were expressed per unit of methanol output; we converted to emissions per unit input assuming a 65% (HHV) conversion efficiency.

[c] The estimates by Ecotraffic AB (1992) appear to be based on emission factors (cited in Swedish studies) for heaters and flares used in the recovery of crude oil. Ecotraffic expressed its estimates per unit of methanol output; we converted to emissions per unit input, using Ecotraffic's estimated 70% (LHV) conversion efficiency. Ecotraffic's estimate of CH_4 emissions is based on an assumed 0.1% gas leakage rate. Its estimate of HC emissions is a "hydrocarbon equivalent," in which any methanol emissions are multiplied by 0.19 (methanol's O_3-forming potential relative to gasoline's).

Ecotraffic AB (1992) estimates N_2O emissions of less than 1.0 g/10^6-BTU for the NG-to-methanol conversion process.

[d] Darrow's (1994) estimates are based on emission factors for gas boilers. The low NO_X-emissions estimate assumes emission controls in the year 2000; the high estimate assumes no controls today.

[e] Data on CH_4 emissions from plants that produce methanol *and* other products (Texas Air Control Board, 1990) combined with data on the production capacity of methanol facilities (U. S. Department of Commerce, 1985) indicate that CH_4 emissions may be on the order of 1-10 grams per 10^6 BTU of methanol. However, it is not clear how CH_4 emissions should be allocated among the multiple products. Note, though, that this range for methanol (1-10) is consistent with the range estimated for petroleum refineries (0.24-2.4), because methanol plants process natural gas, whereas methanol plants process crude oil, and one would expect higher CH_4 emissions from a facility that process natural gas.

[f] The IPCC (1997) recommends an emission factor of 2.0 g-CH_4/kg-methanol produced. Given 46.45 kg/10^6-BTU-methanol, and assuming a 65% NG-to-methanol energy conversion efficiency, the emission factor is 93 g-CH_4/10^6-BTU methanol or 60 g-CH_4/10^6-BTU-NG input to the plant. The IPCC factor is mentioned in the EIA (1998), and others.

[g] The Tellus (1993) estimates are from an EPA data base.

[h] The HC figure is NMOCs. The high CH_4 emission rate is supported by data in Dahlberg et al. (1988), which indicate that CH_4 emission from the combustion of wood chips is almost 100 times higher than from combustion of fossil fuels -- 300 ppmv in effluent gas vs. 5 ppmv.

[i] The lower figure is for bark and wet-wood fired boilers, with no controls; the higher figures is for dry-wood fired boilers, with no controls. Note that the relevant New Source Performance Standards for any industrial-steam-generating unit is 136 g/10^6 BTU.

[j] If one calculates emissions on the basis of the sulfur content of wood (0.09% for poplar, as discussed elsewhere in this report), assuming no emission controls, then SO_2 emissions are on the order of 90 g/10^6-BTU.

[k] The low factor is emissions from wood-and-bark-fired boiler with an electrostatic precipitator; the high end is uncontrolled emissions from a dry-wood fired boiler.

[l] Ismail and Quick's (1991) data pertain to a plant in Fresno, California. Wood-fired FBC power plants in Maine must meet a 68 g/10^6 BTU standard for NO_x, CO, and NMHCs.

TABLE 18, CONTINUED

B. GRAMS PER 10^6-BTU FUEL OUTPUT

Reference	Product/process	Feedstock	HCs	CO	NO_x	SO_x	PM	CH_4
Sperling (1988)	methanol / gasification and synthesis	coal (0.4-0.6% sulfur)	100-500[a]	n.e.	15-150[a]	30-200	1-25	n.e.
Sperling (1988)	ethanol / fermentation	corn	5-140	10-170	100-830	37-1500	45-370	n.e.
USDOE (1983)	ethanol / fermentation	corn	432[b]	n.e.	174	227	76[c]	n.e.
USDOE (1988)	ethanol / fermentation: distillation and dehydration only	corn	n.e.	n.e.	n.e.	n.e.	n.e.	0.22
USDOE (1983)	methanol / gasification and synthesis	wood	n.e.	n.e.	18	n.e.	n.e.	n.e.
Sperling (1988)	methanol / gasification and synthesis	wood	n.e.	n.e.	10-200[a]	neg.	0-30[a]	
Ecotraffic AB (1992)[d]	methanol / gasification and synthesis	wood	8	19	11	3	n.e.	n.e.
USDOE (1988)	ethanol / gasification and fermentation	biomass	n.e.	n.e.	53	104	3	n.e.
Ecotraffic AB (1992)[e]	ethanol / conversion of lignocellulose	tree residues	28	81	93	11	n.e.	32
Ecotraffic AB (1992)[d]	ethanol / conversion of lignocellulose	SRIC trees	121	258	137	42	n.e.	n.e.
Riley & Schell (1992)[f]	ethanol / acid hydrolysis & fermentation	switchgrass wheat grass	6	36	25	9	16	n.e.
Riley & Schell (1992)[f]	ethanol / acid hydrolysis and fermentation	cottonwood and alder	9	53	30	4	20	n.e.

Notes: see next page.

n.e. = not estimated; SRIC = short-rotation intensive cultivation; NREL = National Renewable Energy Laboratory. Use of controls is mentioned if known.

[a] According to Sperling (1988), the upper bounds are “suspect.”

[b] Ethanol.

[c] Includes fugitive dust.

[d] Ecotraffic AB (1992) assumes that emissions from the methanol and ethanol conversion processes arise from the combustion of biomass (lignin) for process heat. They calculate these emissions by multiplying emission factors for lignin combustion (g/BTU-lignin) by lignin-use factors (BTU-lignin/BTU-fuel). The emission factors for lignin combustion are undocumented assumptions, and are the same for both processes. However, the lignin-use factor is much higher for the ethanol process (1.63:1) than for the methanol process (0.12:1). Hence, the considerable difference in g/BTU-fuel emission rates is due entirely to the considerable (unexplained) difference in assumed lignin usage rates.

[e] Ecotraffic AB (1992) assumes that emissions from this process arise from the combustion of biomass (lignin) and biogas for process heat. They calculate these emissions by multiplying emission factors for lignin combustion (g/BTU-lignin) by lignin-use factors(BTU-lignin/BTU-fuel) and emission factors for biogas (g/BTU-gas) by biogas-use factors (BTU-gas/BTU-fuel). The emission factors for lignin combustion are undocumented assumptions. Their emission factors for biogas combustion are from the product specifications of an engine.

[f] This study includes emissions from all operations at the plant site, including evaporative emissions from storage tanks, emissions from diesel loading equipment, fugitive emissions from vents, emissions from on-site utilities, and more. It is by far the most detailed study of emissions from biomass conversion that we have seen.

TABLE 18, CONTINUED

C. ASSUMPTIONS IN THIS ANALYSIS (G/10^6-BTU-FEEDSTOCK, EXCEPT AS NOTED)

	Ethanol		Methanol		
	corn[a]	*wood or grass*[b]	*NG*[c]	*coal*[d]	*wood*[e]
Aldehyde (as HCHO) exhaust	n.e.	n.e.	n.e.	n.e.	n.e.
Fuel evaporation or leakage[f]	432.0	4.5	14.5	4.5	4.5
NMOC exhaust[g]	0.0	0.0	0.3	88.2	2.1
Evaporation + NMOC exhaust	432.0	4.5	14.7	92.8	6.7
Carbon in evap. + NMOC exh.	225.4	2.4	5.6	54.6	2.8
Ozone-weighted total NMOC	198.7	2.1	2.6	58.1	2.1
CH_4 (exhaust)[h]	0.2	1.0	10.0	9.3	1.4
CO	0.0	26.6	4.0	7.6	15.6
N_20 [i]	0.0	1.1	0.5	1.4	0.8
NO_x (NO_2)	0.0	16.5	30.0	29.4	11.1
SO_x (SO_2)	n.e.	4.1	0.1	29.4	0.9
PM (combustion-like) [j]	n.e.	11.0	0.1	5.9	5.6
PM_{10} (combustion-like) [j]	n.e.	n.e.	0.1	4.4	4.2
$PM_{2.5}$ (combustion-like) [j]	n.e.	n.e.	n.e.	n.e.	n.e.
PM (dust-like) [j]	n.e.	n.e.	n.e.	n.e.	n.e.

n.e. = not estimated; NG = natural gas; evap. = evaporation; exh. = exhaust.

[a] Emissions from process areas (not emissions from boilers, which here are estimated separately), in g/10^6-BTU-fuel-output. I assume that the only significant emissions from process areas are evaporative/leakage emissions of the product ethanol, as estimated by USDOE (1983) in part B of this Table.

[b] The values shown here are for a 50%-wood/50%-grass feedstock mix. Except as noted, my assumptions are based on the estimates in Riley and Schell (1992) converted from a fuel-output basis (as shown in Part B of this table) to a feedstock-input basis, and weighted 50% grass 50% wood.

[c] Except as noted, my assumptions are based on the lower end of the estimates shown in Part A of this table, converted from a fuel-output basis to a feedstock-input basis assuming 1.5 BTUs-gas / BTU-methanol.

[d] Except as noted, my assumptions are towards the lower end of the range shown by Sperling (1988) (Part B of this table), converted from a fuel-output basis to a feedstock-input basis assuming 1.7 BTUs-coal / BTU-methanol.

[e] Except as noted, my assumptions are towards the lower end of estimates shown in Part B of this table, converted from a fuel-output basis to a feedstock-input basis assuming 1.8 BTUs-wood / BTU-methanol.

[f] Evaporative or leakage loss of ethanol or methanol. In the case of NG-to-methanol, this includes leaks of NG feedstock, assuming 0.05%leakage (1 / 2 of the rate assumed by EcoTraffic in part A of this table), as well as evaporation of methanol. The rate for corn / ethanol is from USDOE (1983) in part B of this table; the rate for wood or grass / ethanol is from Riley and Schell (1992) in part B; the rate for methanol I assume is the same as the rate for wood / ethanol.

[g] NMOC missions from process areas and boilers except evaporative and leakage emissions. (In the case of corn / ethanol, emissions from process areas only.)

[h] Process-area emissions of methane. In the case of NG-to-methanol, this line excludes leaks of natural gas feedstock, which leakage is estimated separately, as noted above. In the case of corn / ethanol, this line excludes CH_4 emissions from boilers, which are estimated separately. My assumptions for corn / ethanol and NG-to-methanol are based on the estimates in Parts A and B of this Table. For coal-to-methanol, I assume that CH_4 is 10% of total evaporative+exhaust NMOC emissions. For wood-to-ethanol and wood-to-methanol, I assume that the ratio of CH_4:NMOC is the same as the CH_4:NMOC ratio for fluidized-bed combustion of wood waste, which according to our estimates based on EPA AP-42 data is 2.0 / 9.6 = 21%.

[i] For wood-to-ethanol and wood-to-methanol, I assume that the ratio of N_2O:NO_X is the same as the N_2O:NO_X ratio for fluidized-bed combustion of wood waste, which I assume is 3.4 / 50.4 = 7%. (The 50.4 g-NO_X / 10^6-BTU is an estimate of controlled NO_X emissions, 1 / 2 of the uncontrolled emission factor from Part A of this table.)

[j] For the purpose of calculating CO_2-equivalent emissions, the LEM has CEFs for black carbon aerosols from combustion, organic-matter aerosol from combustion, and dust (which generally comprises earth-crustal material) (Appendix D). Thus, in order to be

able to calculate CO_2-equivalent emissions, it is necessary to classify all PM emissions as either dust-like or from combustion (or combustion-like) processes. Lacking data to the contrary, I have assumed that all PM emissions reported here are combustion-like.

TABLE 19. FERTILIZER USE IN CORN AND SOYBEAN FARMING

A. Corn

Year[a]	Yield	Fertilizer and pesticide applied (lb/bu-harvested)[b]			
	bu/acre[b]	*Nitrogen*	*Phosphate*	*Potash*	*Pesticide[c]*
1970-96	103.5	1.22	0.55	0.63	n.e.
1980-96	111.6	1.20	0.49	0.60	n.e.
1985-96	116.4	1.15	0.45	0.55	n.e.
1990-96	119.8	1.10	0.42	0.51	0.026
1965-1969	78.5	1.08	0.65	0.61	n.e.
1970-1974	84.1	1.25	0.69	0.70	n.e.
1975-1979	95.1	1.25	0.61	0.68	n.e.
1980-1984	100.2	1.32	0.58	0.72	n.e.
1985-1989	111.6	1.23	0.49	0.60	n.e.
1990-1996	119.8	1.10	0.42	0.51	0.026

B. Soybeans

Year[a]	Yield	Fertilizer and pesticide applied (lb/bu-harvested)[b]			
	bu/acre[b]	*Nitrogen*	*Phosphate*	*Potash*	*Pesticide[c]*
1970-96	31.0	0.11	0.41	0.67	n.e.
1980-96	32.7	0.11	0.39	0.70	n.e.
1985-96	34.4	0.10	0.34	0.65	n.e.
1990-96	36.0	0.10	0.31	0.60	0.033
1965-1969	25.7	0.09	0.35	0.41	n.e.
1970-1974	26.7	0.12	0.43	0.57	n.e.
1975-1979	29.4	0.12	0.47	0.68	n.e.
1980-1984	28.5	0.12	0.50	0.83	n.e.
1985-1989	32.1	0.09	0.39	0.72	n.e.
1990-1996	36.0	0.10	0.31	0.60	0.033

n.e. = not estimated. See the text for methods of estimation and sources of data.

[a] Crop production data are reported for a "marketing year," which begins on September 1. Thus, the year 1990 in this table corresponds to the marketing year September 1 1990 to August 31 1991. However, the production data -- acres planted, acres harvested, and bushels harvested -- actually apply to the crop that will be marketed in the 1990/91 marketing year. This crop will be planted in spring and harvested in fall. In essence, then, the production data for marketing year 1990 representing plantings and harvesting in calendar year 1990 (Riley, 1997).

Now, the fertilizer-use data nominally apply to a fertilizer year ending on June 30th of the year shown. However, in practice they include all fertilizer applied during the growing season (Taylor, 1997). Hence, the fertilizer-use data and the production data apply to the same crop year.

[b] The yield and the application rate are expressed per harvested acre, not per planted acre. Some acres are planted but ultimately abandoned and not harvested. Presumably, less fertilizer, pesticide, and energy is used on acreage that ultimately is abandoned. See the discussion in the text.

[c] Herbicides, insecticides, fungicides, and other chemicals.

TABLE 20. CURRENT AND PROJECTED MATURE DRY HARVEST YIELDS FROM SWITCHGRASS AND HYBRID POPLAR ENERGY-CROP PLANTATIONS, FROM ORNL

	Yield (dry tons/acre/year)					**?/yr.**[a]
	~1996[b]	*2005*	*2010*	*2015*	*2020*	
Switchgrass, cropland	4.9	5.2	5.5	5.7	5.9	0.77%
Switchgrass, pastureland	4.3	4.1	4.4	4.5	4.7	0.43%
Switchgrass, all land[c]	*4.7*	*4.9*	*5.1*	*5.3*	*5.6*	*0.67%*
Hybrid poplar, cropland	4.7	5.2	5.6	6.0	6.5	1.36%
Hybrid poplar, pastureland	3.7	4.1	4.4	4.7	5.1	1.36%
Hybrid poplar, all land	*4.5*	*4.9*	*5.3*	*5.7*	*6.2*	*1.36%*

Source: Calculated from the projections of experts who participated in a recent review of biomass cultivation practices (Walsh, 1997a). Walsh reports the mature, bone-dry, harvest yield, for switchgrass and hybrid poplar, on crop land and pasture land, in every state with some land suitable for energy crop production. She also reports the number of acres of crop land and pasture land suitable for switchgrass and hybrid poplar production in each state. I have weighted each state's projected per-acre yield by its share of the total suitable acreage nationwide. To the extent that the distribution of acreage that actually will be used for energy crop production (which is what we really wish to know) is not the same as the distribution of suitable (or "potential") acreage, the acreage weights and hence the acreage-weighted national yields will not be correct.

[a] Calculated average over the period1996 to 2020.

[b] Walsh (1997a) shows this as "current," which I interpret to mean around 1996.

[c] The summary tables in Walsh (1997a) report the mature yield. In the case of switchgrass, however, the first harvest brings only 30% of the mature yield, and the second harvest 67% (Walsh, 1997a). For the next 8 years of the 10-year rotation, the full mature yield is harvested. Therefore, the average yield over the 10-year rotation is 89.7% of the mature yield shown in their summary tables (Walsh, 1998). The estimates shown here account for this: they are equal to the projected mature yields multiplied by 0.897.

In the case of hybrid poplar, the actual yield at the end of every rotation is equal to the mature yield projected in Walsh (1997a).

TABLE 21. INPUTS TO ENERGY-CROP FARMING

A. Fertilizer, pesticides and seeds (lbs/bu-corn, lbs/bu-soy, lbs/net-ton-wood, lbs/net-ton-grass)

Input-----> ***Feedstock***	**N** *lbs*	**P_2O_5** *lbs*	**K_2O** *lbs*	**Lime** *lbs*	**Sulfur** *lbs*	**Pestic.** *lbs*	**Seeds** *lbs*
Corn (per bushel)							
Base-year value[a]	1.122	0.429	0.520	0.337	0.010	0.027	0.04
Percent change / year[b]	-0.50	-1.00	-1.00	-2.00	-2.00	-0.30	0.00
Year 2015 value[c]	1.010	0.347	0.421	0.220	0.007	0.025	0.04
Soybeans (per bushel							
Base-year value[a]	0.102	0.316	0.612	0.000	0.000	0.034	0.05
Percent change / year[b]	-0.50	-1.00	-1.00	-2.00	-2.00	-0.30	0.00
Year 2015 value[c]	0.092	0.256	0.496	0.000	0.000	0.032	0.05
Wood (per net ton)[d]							
Base-year value[a]	2.05	1.53	1.11	39.2	0.00	0.13	0.00
Percent change / year[b]	-1.36	-1.36	-1.36	0.00	0.00	-1.36	0.00
Year 2015 value[c]	1.79	1.33	0.96	39.19	0.00	0.11	0.00
Grass (per net ton)[d]							
Base-year value[a]	20.40	0.79	0.73	36.8	0.00	0.09	0.09
Percent change / year[b]	0.00	-0.67	-0.67	-0.67	0.00	-0.67	0.00
Year 2015 value[c]	20.40	0.74	0.68	34.40	0.00	0.08	0.09

Notes after part B of table.

B. Fuel and electricity use

Input-----> ***Feedstock***	**Diesel** *gal*	**Fuel oil** *gal*	**NG** *10^3CF*	**Coal** *lbs*	**Power** *kWh*	**Gas-oline** *gal*	**LPG** *gal*	**Bio-fuel**[e] *gal*
Corn (per bushel)								
Base-year value[a]	0.066	0.000	0.002	0.000	0.373	0.034	0.033	0.000
Percent change / year[b]	-0.30	-0.30	-0.30	-0.30	-0.30	-0.30	-0.30	-0.30
Year 2015 value[c]	0.062	0.000	0.002	0.000	0.351	0.032	0.031	0.000
Soybeans (per bushel)								
Base-year value[a]	0.177	0.000	0.002	0.000	0.139	0.105	0.012	0.000
Percent change / year[b]	-0.30	-0.30	-0.30	-0.30	-0.30	-0.30	-0.30	-0.30
Year 2015 value[c]	0.166	0.000	0.002	0.000	0.130	0.099	0.011	0.000
Wood (per net ton)[d]								
Base-year value[a]	2.20	0.00	0.00	0.00	5.00	0.30	0.00	0.00
Percent change / year[b]	-0.30	-0.30	-0.30	-0.30	-0.30	-0.30	-0.30	-0.30
Year 2015 value[c]	2.13	0.00	0.00	0.00	4.85	0.29	0.00	0.00
Grass (per net ton)[d]								
Base-year value[a]	1.70	0.00	0.00	0.00	5.00	0.30	0.00	0.00
Percent change / year[b]	-0.30	-0.30	-0.30	-0.30	-0.30	-0.30	-0.30	-0.30
Year 2015 value[c]	1.65	0.00	0.00	0.00	4.85	0.29	0.00	0.00

Pestic. = pesticide

[a] Base years are 1994 for corn and soybeans, and 2005 for wood and grass. The base-year estimates of fertilizer and pesticide use on corn and soybeans are the average application rates (lb / bu) over the period 1990 to 1996 (Table 19; see discussion in the text). The base-year estimates for fuel and electricity use for corn and soybean farming are derived from data in the *FCRS* (Ali and McBride, 1994a, 1994b). Rates have been adjusted from a basis per harvested bushel to a basis per bushel into the fuel production plant, after losses.

The base-year estimates for wood and grasses are from analyses and reviews done by ORNL and others; see the text for further details.

[b] These are my assumptions, based on past trends and my judgment in the case of corn and soybeans, and projections for wood and grass fuels. See the discussion in the text.

[c] Equal to: $V_T = V_{T_B} \cdot \left(1 + \frac{PC}{100}\right)^{T - T_B}$, where V_T = the value in the target year, V_{T_B} = the value in the base year, PC = the percentage change per year, T = the target year, and T_B = the base year (see note a).

[d] Net ton delivered to the fuel-production plant; equal to gross tonnage produced less losses during harvesting and transport.

[e] In the case of the corn / ethanol cycle, this is corn-derived ethanol; in the biodiesel cycle, it is soy-derived biodiesel; in the wood and grass fuel cycles, it is ethanol.

TABLE 22 HAS BEEN MOVED TO APPENDIX H.

TABLE 23. VENTING AND FLARING OF GAS ASSOCIATED WITH OIL PRODUCTION

	Baseline input data, 1995[a]						**rate of change[b]**	
	10^9 SCF VF	under report factor	10^3 bpd crude	10^3 tpd crude	calc. SCF/ ton	flare fract.	SCF/ bbl % ?	flare (k exp)
U. S. -- domestic production	297	1.05	6,560	1,004	850	0.870	0.0	-0.005
Canada	89	1.05	1,805	277	925	0.870	0.0	-0.005
Mexico	78	1.10	2,618	414	567	0.850	0.0	-0.010
Northern Europe (U. K., Norway)	95	1.05	5,257	764	358	0.870	0.0	-0.005
Venezuela	126	1.10	2,750	440	863	0.830	-0.5	-0.010
North Africa (Algeria, Libya)	314	1.20	2,592	365	2,827	0.830	-1.0	-0.010
Nigeria	926	1.25	1,933	291	10,885	0.800	-2.0	-0.015
Indonesia	177	1.10	1,503	219	2,433	0.850	-1.0	-0.010
Persian Gulf OPEC[c]	892	1.10	17,166	2,584	1,040	0.830	0.0	-0.010
Other Middle East[d]	24	1.10	1,771	266	272	0.830	0.0	-0.010
Other Latin America[e]	124	1.10	1,692	262	1,424	0.830	-1.0	-0.010
Other Africa[f]	195	1.25	1,011	150	4,439	0.800	-1.0	-0.015
Other Asia (China, Malaysia)[g]	600	n.a.	3,672	553	2,972	0.830	-1.5	-0.010
Former Soviet Union (Russia)[g]	1,000	n.a.	5,995	903	3,034	0.830	-2.0	-0.010
Total world	3,828	n.e	60,213	n.e.	n.e.	n.e.	n.a.	n.a.

SCF = standard cubic feet; VF = vented or flared; bpd = barrels per day; tpd = tons (2000 lbs) per day; fract. = fraction; Calc. = calculated; exp = exponent; bbl = barrel; ? = change; n.e.= not estimated; n. a. = not applicable.

[a] The EIA's *International Energy Annual 1996* (1998) reports venting and flaring of associated gas (the column "10^9 SCF VF") and crude oil production (the column "10^3 bbd crude"), by country, in 1995. The vented or flared figure shown here for the U. S. includes our estimate of gas vented and flared from U.S. Federal offshore oil platforms. We assume that 48 SCF of NG is vented or flared per bbl of crude oil from Federal offshore wells, and that crude oil production from Federal offshore wells is 11% of total U. S. crude oil production. See Appendix E for more details.

However, it is likely that the amount of venting and flaring is underreported. For example, in the U.S., six states do not report venting and flaring emissions to the EIA (1995c)\ . We estimate that venting and flaring emissions in these states are about 2% of reported venting and flaring in all other states. On the assumption that the states that do report venting and flaring might under-report slightly, we assume that the true venting and flaring emissions in the U.S. are 5% higher than the amount reported to the EIA.

We assume that underreporting is higher in South America and the Middle East than in the industrialized nations of the West, and highest in Africa. (The under-reporting factor is not applicable in the case of "other Asia" and "Former Soviet Union" because for these countries no venting and flaring data are reported.)

In order to estimate venting and flaring emissions per unit mass rather than per unit volume, we convert barrels of crude oil to tons of crude oil by multiplying by tons/bbl, for each country, as reported by the EIA's *International Energy Annual 1996* (1998). The result is tons per day, in the column "10^3 tpd crude".

The column "calc. SCF/ton" is equal to reported SCF vented or flared, multiplied by the underreporting factor, divided by tons produced per year.

Finally, our assumptions regarding the fraction that is flared, rather than vented (the column "flare frac.") are explained in Appendix E.

[b] The column "SCF/bbl % ?" shows the assumed percentage change per year in the venting and flaring emission rate for each country. I assume that the SCF/bbl emission rate remains constant in areas with relatively low rates in 1995 and relatively well developed oil fields and gas markets (U.S., Canada, Mexico, Northern Europe, and the Middle East), and declines slightly in areas with high SCF/bbl emission rates in 1995 and relatively poorly developed gas markets. (The rates in Africa and the Former Soviet Union are assumed to decline the most.)

I represent the fraction flared, over time, as a logistic function that approaches 1.0 asymptotically (Eq. 6). The parameter "k" is the exponent in the logistic function. The larger the absolute value of k, the greater the rate of change in the flared fraction. My assumptions for k follow my assumptions for the rate of change in SCF/bbl. .

[c] Saudi Arabia, Kuwait, Iran, Iraq, UAE, and Qatar.

[d] Oman, Yemen, and Syria.

[e] Colombia, Ecuador, and Argentina. Prior to January 1, 1993, Ecuador was a member of OPEC (EIA, *PSA 1997*, 1998).

[f] Angola and Gabon. Prior to January 1995, Gabon was a member of OPEC (EIA, *PSA 1997*, 1998).

[g] The EIA (1998) reports zero SCF vented and flared, but this presumably means that data were unavailable. There undoubtedly is considerable venting and flaring of associated gas in Russia, Asia, and Central Asia. For example, in a discussion of potential future markets for natural gas in Russia,

another EIA report (2002) says “Russian oil companies currently produce approximately 2.2 TCF of associated natural gas that could be treated and exported rather than flared off” (p. 5). The International Energy Agency’s (IEA, 2002) review of the energy situation in Russia cites a government report that Russian oil companies flared 7.2 billon cubic meters (Bcm) in 1999 (about 250 BCF), but suggests that “the volume of flared gas could be as high as 25 to 30 Bcm” (p. 250) (about 1000 BCF). Considering this, I have estimated considerable venting and flaring emissions in Russia (which I use to represent the Former Soviet Union) and Asia.

TABLE 24. VENTING AND FLARING OF GAS FROM COAL MINES.

	BTU /ton	SCF/ton-coal		Gas volume fraction			Sur-face	Calculated SCF/ton	
Producing region	a	b	c	d	e	f	g	h	i
U. S.	1.00	510	40	0.11	0.05	0.75	0.69	138.4	21.2
Canada	1.00	510	40	0.11	0.05	0.75	0.65	152.5	23.4
Colombia	1.10	510	40	0.00	0.05	0.75	0.60	226.9	11.9
N. Europe	1.00	510	40	0.20	0.05	0.75	0.50	149.7	45.5
Poland, Czech republic	1.20	230	1	0.00	0.05	0.75	0.34	151.5	8.0
South Africa	1.10	510	40	0.00	0.05	0.75	0.60	226.9	11.9
Australia	1.00	310	63	0.10	0.05	0.75	0.70	138.9	19.7
Indonesia	1.10	510	40	0.05	0.05	0.75	0.60	201.2	18.4
Former Soviet Union	1.10	850	80	0.05	0.05	0.75	0.42	465.6	42.5
China	1.10	510	40	0.05	0.05	0.75	0.60	201.2	18.4
Other	1.10	510	40	0.05	0.05	0.75	0.60	201.2	18.4
Target developed	1.00	510	40	0.10	0.05	0.75	0.70	138.9	19.7
Target LDC	1.10	510	40	0.05	0.05	0.75	0.60	201.2	18.4
Year of baseline data	n.a.	1994	1994	1994	1994	1994	n.a.	n.a.	n.a.
Percentage change p.a. year	n.a.	0.20	0.00	3.00	0.00	0.00	n.a.	n.a.	n.a.

Source: estimates for the U. S. are discussed in Appendix E. Estimates for other countries are my assumptions, in some cases explained in the notes. n. a. = not applicable; p.a. = per anum; target developed = the target country designated for analysis, when the target country is a developed country; target LDC = the target country designaged for analysis, when the target country is a less-developed country.

[a] The energy intensity of coal production, in BTU-process energy per ton of coal produced, *relative* to that estimated for the U. S. These are my estimates.

[b] Emissions of coalbed gas from underground mines, in SCF per ton of gas, in the baseline year. The values for the U. S. are discussed in the text and Appendix E. For other countries I assume U. S. values except as follows:

Poland, Czech Republic: Poland's National Fund for Environmental Protection and Water Management (2001) has completed a GHG emissions inventory for the United Nation's Convention on Climate Change. In this inventory, Poland used the following CH_4 emissions factors for coal mining: underground mining: ~230 SCF/ton (value varies slightly year-by-year); surface mining: 0.6 SCF/ton.

Australia: The Australian Greenhouse Office (2002) national greenhouse gas inventory reports 6.64 kg-CH_4/tonne-coal and 1.36 kg-CH_4/tonne-coal (p. B-12).

Former Soviet Union: The Russian Federal Service for Hydrometeorology and Environmental Monitoring (1997) performed a detailed estimate of methane emissions from coal mining in Russia, and reported total coal production and total methane emissions from underground and surface mining, from which I calculate the methane emission factor:

	reported 10^6 tonnes coal	reported 10^9 g CH_4	my estimate of SCF-CH_4/ton-coal
underground mining	147.6	2,693	850
surface mining	105.5	181	80

These values are similar to those calculated for the U. S.

[c] Emissions of coalbed gas from surface mines, in SCF per ton of gas, in the baseline year. See note b.

[d] Of the total coalbed gas generated, the volume fraction that is used as a fuel. I assume the same values for all countries, except given consideration of the following.

China: The IEA *Energy Policiesof IEA Countries 2001 Review* reports that new Chinese energy policy calls for the development of coal-bed methane (p. 84). Given this, I assume relatively high value for "use" of coalbed methane (as opposed to venting or flaring) in China.

U. K.: The U. K.'s Department for Environment, Food, and Rural Affairs (2001) reports that methane emissions from coal mining declined 62% from 1990 to 1999 due to reduced coal production and increased use of methane for energy (p. 20). The Department also reports a projection that these emissions from coal mining will continue to decline rapidly through 2020 (p. 52).

Germany: In Germany, a new company called "Minegas" has been founded to exploit the minegas from operational and closed mines for electricity generation (EIA, *Country Analysis Briefs, Germany*, 2001).

[e] Of the total coalbed gas generated, the volume fraction that is flared rather than vented.

[f] Of the total coalbed gas generated and used as a fuel, the volume fraction that displaces other production of gas (the remainder going to satisfy new induced demand).

[g] The ratio of tons of coal produced from surface mines to total tons of coal produced from all mines. I assume the same values for the U. S., except in the following cases:

General. The IEA's *Energy Policies of IEA Countries Japan 1999 Review* (1999) states that there are 2 underground and 11 surface mines in Japan (p. 119). In the U. S. most of the mines are surface also. The IEA's *Energy Policies of IEA Countries Turkey 2001 Review* (2001) states that 90% of lignite production in Turkey comes from open-cast mines (p. 53).

Australia: The IEA's *Energy Policies of IEA Countries Australia 2001 Review* (2001) states that over 70% of hard-coal production in Australia comes from open-cast mines (p. 59). The Australian Greenhouse Office reports that in 2000 68%of coal production came from surface mines (p. B-12).

Poland: The Office of Fossil Energy (1996) reports data that indicate that about 3/4 of the coal in Poland and less than 1/2 of the coal in the Czech Republic come from underground mines.

Former Soviet Union: see note b.

[h] Gas vented in the year 2000. Calculated from the data and assumptions in the other columns.

[i] Gas flared in the year 2000. Calculated from the data and assumptions in the other columns. Gas that is burned but that simply displaces other gas that would have been burned instead is not counted as a net incremental burning.

[j] Mainly Colombia.

[k] Mainly Germany and the United Kingdom.

[l] Mainly Russia and Poland.

[m] Mainly China, India, and Indonesia.

TABLE 25. OIL PRODUCTION BY COUNTRY AND TYPE OF RECOVERY (ONSHORE CONVENATIONAL OIL, OFFSHORE CONVENTIONAL OIL, AND HEAVY OR ENHANCED OIL RECOVERY)

Crude oil produced in:	**Fraction of tonnage from:**		
	onshore conventonal[a]	*offshore conventional*[b]	*heavy/ enhanced*[c]
U. S.	function	function	function
Canada	function	function	function
Mexico	0.60	0.40	0.00
Northern Europe (U. K., Norway)	0.05	0.95	0.00
OPEC	n.a.	n.a.	n.a.
Venezuela	0.40	0.30	0.30
North Africa (Algeria, Libya)	0.80	0.20	0.00
Nigeria	0.80	0.20	0.00
Indonesia	0.80	0.20	0.00
Persian Gulf (Saudi Arabia, Kuwait, Iran, Iraq, UAE, Qatar)	0.85	0.15	0.00
Other Middle East (Oman, Yemen, Syria)	0.80	0.20	0.00
Other Latin America (Colombia, Ecuador, Argentina)	0.80	0.20	0.00
Other Africa (Angola, Gabon)	0.80	0.20	0.00
Other Asia (China, Malaysia)	0.80	0.20	0.00
Former Soviet Union	0.50	0.50	0.00

[a] Oil produced from conventional onshore reserves. This fraction is estimated as 1−offshore−heavy.

[b] Oil produced from conventional offshore reserves.

U. S.: before 1990, 0.15; from 1990 to 2008, increasing from 0.15 to 0.30; after 2008, decreasing from 0.32 in 2009 to 0.15 at the rate of -0.005 per year (estimated on the basis of historical data in the EIA's *AER 1997* [1998], and projections in the EIA's *AEO 1999* [1998]).

Canada: According to the EIA (*Canada*, 1997), offshore oil production is expected to increase from essentially nothing in the early 1990s to over 300,000 bbl/day, or on the order of 10% of total production, by 2001. I assume that the offshore share increases from 1% in 1995 to 10% in 2004.

Mexico: The Office of Fossil Energy (2001) and the EIA's *Mexico* Country Analysis Brief (2001) state that 3/4 of Mexico's oil comes from offshore sites in Campeche Bay in the Gulf of Mexico, and that 52% of the oil Mexico produces is heavy "Maya-22". I infer from the EIA and Office of Fossil energy reports that there is offshore production outside of Campeche Bay, and that virtually all of the heavy "Maya-22" comes from offshore sites.

Northern Europe: Virtually all of the production from Northern Europe (Norway and the U. K.) is from the North Sea (EIA, *North Sea*, 1998). In the U. K, 95% of the production is offshore, and in Norway the percentage apparently is similar (EIA, *North Sea*, 1998).

Persian Gulf: Data in the EIA's *Persian Gulf Oil and Gas Exports Fact Sheet* (2002) suggest that offshore production is less than 20% of total production.

Former Soviet Union: The EIA's *Caspian Sea Region* (2002) states that "most of Azerbaijan's oil resources...and perhaps 30-40%of the total oil resources of Kazakhstan andTurkmenistan are offshore" (p. 2). However, most of Russia's oil production comes from Western Siberia (EIA, *Country Analysis Briefs:Russia,* 2002).

All other countries: my assumptions.

[c] Heavy oil, or enhanced oil recovery.

U. S.: The EIA (*AEO 1999*, 1998) projects that enhanced oil recovery will account for about 14% of U. S. production in 2020, up from about 9% in 1997. I assume that the share increases by 0.002 per year, from 0.04 in 1970, up to a maximum of 0.25.

Canada: According to the EIA (*Canada*, 1997), production from tar sands accounts for about 19% of Canada' s oil supply, and will increase dramatically in the future. I assume that the share increases by 0.006 per year, from 0.11 in 1985, up to a maximum of 0.35.

Venezuela: The major Orinico oil-producing region has heavy crude oil. I assume that it accounts for 60% of Venezuela's output. .

All other countries: my assumptions.

TABLE 26. PRODUCTION OF NATURAL GAS AND NATURAL GAS LIQUIDS IN THE U. S., 1982, 1987, 1992 (10^3 TONS)

	Bureau of the Census			**EIA**		
	1982	*1987*	*1992*	*1982*	*1987*	*1992*
Marketed production[a]	462,231	438,252	463,185	456,232	431,206	462,446
Unprocessed dry gas[b]	150,062	147,730	165,151	103,467	110,351	65,887
Processed wet gas[c]	312,169	290,623	298,034	352,766	320,855	396,560
NGL production[d]	61,746	62,325	65,236	57,734	59,531	63,505
Residue dry gas[e]	250,423	228,297	232,798	295,031	261,324	333,055
Dry gas production[f]	400,485	376,027	397,949	398,498	371,675	398,942

[a] The net marketed production of the gas field: gross withdrawals less gas used for repressuring, quantities vented and flared, and nonhydrocarbon gases removed in treating or processing operations (EIA, *Natural Gas Annual 1995*, 1996). It consists of dry natural gas, which will be shipped to end users via pipeline, and natural gas liquids, which are extracted at natural gas processing plants. It is estimated here as: tdry gas production plus NGL production.

[b] The amount of gas that is sent directly from the production field to consumers, bypassing NGL processing plants. It is estimated here as: total dry gas production less the dry gas output ("residue dry gas") of NGL plants.

[c] The amount of gas sent to NGL processing plants. Estimated here as: marketed production less unprocessed dry gas.

[d] The output of natural gas liquids from NGL processing plants. Estimated here as: the reported volume of each kind of NGL (bbls of ethane, propane, normal butane, isobutane, and pentane; EIA, fax data transmittal, 1997; Bureau of the Census, *1992 Census of Mineral Industries, Industry Series, Natural Gas Liquids, Industry 1321*, 1995; Bureau of the Census,*1987 Census of Mineral Industries, Industry Series, Natural Gas Liquids, Industry 1321*, 1990) multiplied by the ton / bbl liquid density.

[e] The dry gas output of NGL processing plants. Estimated here as: the reported processed volume, in billion cubic feet (BCF) (data from EIA and Census sources cited in note d), multiplied by $22.36 \cdot 10^3$ tons-dry-gas / BCF-dry-gas.

[f] Total dry gas production. Estimated here as: the reported production volume, in BCF (EIA, *AER 1996*, 1997; Bureau of the Census, *1992 Census of Mineral Industries, Industry Series, Crude Petroleum and Natural Gas, Industry 1311*, 1995; Bureau of the Census, *1987 Census of Mineral Industries, Industry Series, Crude Petroleum and Natural Gas, Industry 1311*, 1990) multiplied by $22.36 \cdot 10^3$ tons-dry-gas / BCF-dry-gas.

TABLE 27. ESTIMATED ENERGY INTENSITY OF NATURAL GAS TRANSMISSION IN THE U. S. BY END-USE SECTOR, IN 2015

End use sector[a]	**Normalized distance**[b]	**Share of consumption**[c]	**Energy intensity**[d]
Residential	1.00	0.201	0.033
Commercial	1.00	0.126	0.033
Industrial	0.95	0.366	0.031
Electric Generators	0.90	0.269	0.030
Lease and Plant Fuel	0.00	0.000	0.000
Pipeline Fuel	0.35	0.031	0.012
Transportation end use[e]	1.00 / 0.95	0.008	0.033
Average/total	*0.94*	*1.000*	*0.031*

[a] These are the sectors in the EIA’s *AEO* projections.

[b] The average transmission distance from producing field to end user, relative to the distance to transportation (CNG, LNG) end users. These are my assumptions, reasoned as follows:

Residential and commercial: these end users will be intermingled with CNG and LNG stations.

Industrial and electric generators: these tend to be located on the fringes of urban areas, or well outside of urban areas, and hence presumably are slightly closer to gas producing fields.

Lease and plant fuel: this is consumed at the producing field, prior to gas transmission

Pipeline fuel: it seems likely that total pipeline compressor horsepower is greater in the first half of a transmission system than in the last half; if so, then the consumption-weighted mean distance from field to the pipeline compressor station will be less than half the transmission distance

[c] Equal to the end-use consumption in each sector, in year T, divided by total end-of-pipe consumption in all sectors in year T, as projected by the EIA’s *AEO*. Values shown are for the year 2015. The share for lease and plant fuel is zero because it is consumed before the transmission and distribution system.

[d] Calculated with Eq. 93.

[e] For CNG stations, small-scale LNG stations (where the gas is liquefied at the refueling site), and facilities that produce hydrogen from natural gas, the relative distance is 1.00; for centralized LNG plants, the relative distance is 0.95 (the same as for industrial end uses).

TABLE 28. ESTIMATION OF EMISSIONS FROM THE U. S. NATURAL GAS SYSTEM, 1992

Stage	CH_4 lost (BCF)[a] [$CH4L_{i,US,92}$]	Output (BCF)[b] [$TP_{i,US,92}$]	% change per year[c] [?GL_i]	I/O factors[d] [IO_i]	HHV ratios[e] [MP_i]
Distribution systems	74.6	17,786	-1.00	1.00	1.00
Transmission and storage	109.3	17,863	-0.50	1.00	1.00
Processing	29.5	14,894	-0.50	0.70	0.85
Recovery	71.6	17840	-0.50	1.00	0.89

BCF = billion cubic feet. Variables used in the equation in the text are shown in brackets.

[a] Vented and fugitive emissions in 1992. I estimate this in three steps. First, I deduct from the EPA / GRI (1996) estimates of total (vented+fugitive+unburned methane) emissions the estimates of unburned methane from engines, because in this analysis those are accounted for in the methane emission factors for the engines. Then, I account for emissions from foreign pipelines carrying gas to the U. S., by multiplying the EPA / GRI estimate of 105.1 BCF for the transmission and storage stage by a "net-import adjustment factor" of 1.05, derived as follows: I assume estimate that BCF-miles of foreign pipelines that deliver gas to the U. S., less BCF-miles of U. S. pipelines that export gas, is roughly 5% of total BCF-pipeline miles in the U. S. (BCF-miles is equal to volume of gas transmitted in a year multiplied by the average shipping distance.) Third, I multiply the resultant estimates for each stage by the ratio of the EIA's (*Emissions of Greenhouse Gases in the United States 1997* (1998) estimate of total methane emissions for that stage to the original EPA / GRI (1996) estimate of total emissions for that stage. I do this because the EIA (and also the EPA [1998c] refined the EPA / GRI (1996) estimates for 1992 by using better data on numbers of wells, miles of pipeline, gas throughput, and so on. These ratios are 0.92 for recovery, 1.0 for processing, 0.99 for transmission and storage, and 0.97 for distribution.

[b] From EIA's *Natural Gas Annual 1995* (1996), except residue gas datum, which is from a fax data transmittal from EIA (1997). For distribution systems, the output is gas delivered to all U. S. consumers at the end of the pipeline. (For this purpose, consumption of gas by pipeline compressors and gas field facilities and processing plants is excluded, because these are not at the end of a distribution pipeline.) For transmission systems, the output is the amount delivered to distribution systems, which is presumed to be equal to the total consumption at the end of the pipeline, plus the leakage loss from distribution systems. For processing, the throughput is dry residue gas from NGL plants. For production, the output is total dry gas production.

Methodologically (but not practically, given the small loss rates), it is important to note that these are the outputs of and not the inputs to each stage.

[c] My assumptions. Note that this is the percentage change in the loss rate (gas lost divided by gas throughput), not the percentage change in the total amount of gas lost. See the text for some discussion.

[d] The ratio of the output of the stage shown to the output of the previous stage, not counting lost fuel or own-use fuel, which are treated separately. In general, this is 1.0 for all stages except natural gas processing. Not all produced gas goes to processing plants; some is clean enough to go directly to transmission plants. The relevant ratio of the dry gas output of processing plants to total dry gas production can be estimated on the basis of the data in Table 26 .

[e] The HHV of NG output from stage i divided by the HHV of all products output from stage i. Because the transmission and distribution stages produce only NG, this parameter is 1.0 for these stages. However, the production stage and the processing stage produce natural-gas liquids as well as natural gas. The data of Table 26, along with data on heating values for NG and NGLs (EIA, data transmittal, 1997; EIA, *AER 1996*, 1997), can be used to estimate the energy ratios.

TABLE 29. WATER AND PIPELINE SHIPMENT OF PETROLEUM, 1994

A. Domestic waterborne shipment

Product[a]	Produced	Shipped by water				TS/ TP[f]
	10^3 tons[b]	10^3 tons[c]	10^6 ton-miles[c]	by barge[d]	miles[e]	
Crude oil	791,706	114,064	256,655	0.021	2,250	0.14
Finished petroleum products excl. still gas, other hydrocarbons	1,031,826	276,689	156,555	0.423	566	0.27
Motor gasoline, aviation gasoline, natural gasoline, gasoline blending components, naptha-type jet fuel	364,827	99,898	73,594	0.344	737	0.27
Distillate fuel	183,209	60,197	30,097	0.393	500	0.33
Kerosene, kerosene-type jet fuel	78,440	2,102	491	0.733	234	0.03
LNG, LPG[g]	104,796	2,401	799	0.983	333	0.02
Napthas, solvents, pentanes	31,427	6,316	3,616	0.511	573	0.20
Residual fuel, unfinished oils, other oils	117,534	78,980	31,806	0.459	403	0.67
Light products (gasoline, distillate, kerosene, LPG)[h]	731,272	164,598	104,981	0.363	638	0.23
Heavy products (all non-light)[h]	300,554	112,091	51,574	0.547	460	0.37

B. Pipeline shipments

Product	Produced	Shipped by pipeline, tram, or conveyor		TS/ TP[f]
	10^3 tons[b]	10^3 tons[i]	miles[i]	
Crude oil, 1994	791,706	450,000	800	0.57
Finished petroleum products excl. still gas, other hydrocarbons, 1994	1,031,826	500,000	400	0.48

LNG = liquefied natural gas; LPG = liquefied petroleum gas; excl. = excluding; n.e. = not estimated

[a] Product grouping is based on detailed commodity mapping provided by the Army Corps of Engineers (data transmittal, 1997).

[b] Crude oil: equal to field production + imports+unaccounted for - stock change - exports (EIA, *PSA 1994*, 1995).

Petroleum products: equal to field production + refinery production + imports - stock change - exports (EIA, *PSA 1994*, 1995). I exclude exported petroleum from this calculation because it is not part of the supply of petroleum to the US, and because most likely they it is not shipped by pipeline.

[c] From the Army Corps of Engineers (1995). All waterborne commerce is either domestic commerce or foreign commerce. Domestic commerce includes domestic traffic to and from Alaska, Puerto Rico, Hawaii, the Virgin Islands, Guam, American Somoa, Wake Island, the U. S. Trust Territories, and any of the lower 48 States. Thus, crude oil shipped from Alaska to the lower 48, or from the lower 48 to Hawaii, is domestic commerce.

By contrast, foreign commerce is waterborne import and export traffic between any foreign country and the United States, Puerto Rico, and the Virgin Islands. A foreign commodity off-loaded at a U. S. port onto a domestic carrier -- for example, foreign oil off-loaded at Baton Rouge to a Mississippi river barges -- at that point becomes domestic commerce.

Presently, essentially no foreign commodities go directly to interior ports (e.g., St. Louis); all are off-loaded to domestic carriers at coastal ports (Army Corps of Engineers, personal communication, 1997). This is consistent with my assumption that all crude oil imported to the U. S. is off-loaded at coastal cities (Houston, New York, or Los Angeles).

[d] Of total domestic ton-miles in 1995, the fraction that went by barge. Calculated from 1995 ton-mile data (Army Corps of Engineers, data transmittal, 1997).

[e] Equal to ton-miles shipped divided by tons shipped.

[f] Tons shipped per ton produced.

[g] This must be all LPG, because LNG is shipped from Algeria or the United Arab Emirates to coastal ports in Massachusetts or Louisiana (EIA, *Natural Gas Annual 1996*, 1997), and hence is foreign, not domestic, commerce. Furthermore, LNG is transported in specialized cryogenic vessels, not barges.

[h] Similar but not identical to the “light” and “heavy” categories used in the EIA’s *AEO* projections.

[i] See the discussion in the text.

TABLE 30. PRIMARY SOURCES OF DATA ON DOMESTIC COAL TRANSPORTATION

A. Ton and ton-mile data, 1993 and 1995

Source	Rail		Water		Truck		Tram/conveyor	
	10^6 ton	*10^6 ton-mi*	*10^6 ton*	*10^6 ton-mi*	*10^6 ton*	*10^6 ton-mi*	*10^6 ton*	*10^6 ton-mi*
EIA, 1995[a]	592	n.e.	234	n.e.	105	n.e.	100	n.e.
EIA, 1993[a]	532	n.e.	215	n.e.	116	n.e.	96	n.e.
Census, 1993[b]	700	418,000	130	50,000	230	10,000	140	9,000
ICC, 1993[c]	n.a.	n.a.	n.e	n.e	n.e	n.e	n.e	n.e
ACE, 1993[d]	n.e	n.e	215	99,172	n.e	n.e	n.e	n.e
ACE, 1995[d]	n.e	n.e	223	96,881	n.e	n.e	n.e	n.e

B. Calculated tons-shipped/ton-produced (TS/TP)and average length of haul one way (LH1W)

Source	Rail		Water		Truck		Tram/conveyor	
	TS/TP[e]	*LH1W*[f]	*TS/TP*[e]	*LH1W*[f]	*TS/TP*[e]	*LH1W*[f]	*TS/TP*[e]	*LH1W*[f]
1995	0.57	n.e.	0.23	430	0.10	n.e.	0.10	n.e.
1993	0.56	600	0.23	380,460	0.12	43	0.10	64

n.e. = not estimated; n.a. = not available

[a] EIA *Coal Industry Annual 1993* (1994) and *Coal Industry Annual 1995* (1996). The EIA sends form EIA-6, "Coal Distribution Report," to all U. S. coal producers or distributors that own or purchase and distribute more than 50,000 short tons annually. The survey excludes imported coal, which however is less than 1% of domestic coal production.

For a small amount of coal (less than 1% of the total), the EIA reports the mode of distribution as "unknown." I have distributed this amount to the four mode categories proportionately.

[b] Bureau of the Census, *1993 CFS* (1996). For my purposes, the *CFS* data have several serious shortcomings compared to the EIA's data. The ultimate problem is that the CFS does not report, or provide enough data to enable one to calculate, total tons and ton-

miles for the aggregate generic modes “truck,” “water,” “rail,” and “pipeline.” There are three reasons for this. First, for some of the detailed mode categories in the *CFS*, such as “private truck,” the *CFS* does not report shipment data, so as not to disclose data about individual companies. Such gaps make it impossible to estimate the total shipped by any kind of truck. Second, the *CFS* has a category “other and unknown,” with a rather substantial amount of tonnage in it. Third, the *CFS* reports shipments by multiple modes, such as “rail and water”. The ton-mile data for multiple-mode shipments cannot be allocated to individual modes.

Because of these deficiencies in the reporting of the *CFS* data, I have had to estimate the totals for the generic categories of this table. The estimate are my judgment. I have assumed that the “other and unknown modes” are pipelines, tramways, and conveyors.

[c] I was not able to obtain railroad waybill data from the Interstate Commerce Commission (ICC).

[d] Army Corps of Engineers *Waterborne Commerce of the United States Calendar Year - 1993* (1994), and *Waterborne Commerce of the United States Calendar Year - 1995* (1996). The figures shown are for domestic transportation only. See also the notes to Table 29.

[e] Equal to total tons shipped (TS) by each mode, as reported by the EIA (part A of this table), divided by total domestic mine production (TP), as reported by the EIA: $945 \cdot 10^6$ in1993 (EIA, *Coal Industry Annual 1993*, 1994), and $1033 \cdot 10^6$ in 1995 (EIA, *Coal Industry Annual 1993*, 1994). I exclude imports from the production denominator (TP) because imported coal is not included in the EIA’s estimate of coal distribution (TS, the numerator). (Imports are less than 1% of domestic production.) However, I do not exclude exports because apparently coal that is to be exported is included in the EIA’s estimate of coal distribution.

[f] The average length of haul is calculated by dividing ton-mile by tons, from part A of the table. The *CFS* ton-mile and ton data indicate 600 miles by rail, 380 miles by water, 43 miles by truck, and 64 miles by what I have presumed to be pipeline, tram, or conveyor, in 1993. The ACE *Waterborne Commerce* data indicate 460 miles in 1993, and 430 miles in 1995.

TABLE 31. DATA USED TO CALCULATE TON-MILES OF SHIPMENT OF PETROLEUM PRODUCTS BY TRUCK, 1992

Average weight when loaded (lbs)[a]		Assumed weight in class[b]	Reported truck miles in class (10^6)[a,c]	Assumed weight when empty[b]	Product weight (fraction of total)[c]
From:	*To:*	*(lbs)*	*(10^6)*	*(lbs)*	
0	6,000	5,000	1,005.5	3500	0.30
6,000	10,000	8,000	664.3	4000	0.50
10,000	14,000	12,000	102.5	5000	0.58
14,000	16,000	15,000	83.4	6000	0.60
16,000	19,500	17,750	132.4	7000	0.61
19,500	26,000	22,750	571.7	9000	0.60
26,000	33,000	29,500	375.3	11000	0.63
33,000	40,000	36,500	148.7	14000	0.62
40,000	50,000	45,000	286.0	17000	0.62
50,000	60,000	55,000	253.1	20000	0.64
60,000	80,000	70,000	2,160.0	23000	0.67
80,000	100,000	90,000	130.4	26000	0.71
100,000	130,000	115,000	179.2	31000	0.73
130,000	--	150,000	15.4	33000	0.78

[a] From the Census *Truck Inventory and Use Survey 1992* (1995).

[b] My assumptions. Note that my assumption for the 60,000-80,000 lb weight class, which has the most truck miles by far, results in 46,000 lbs of product being carried, which is consistent with the EIA's (*Alternatives to Traditional Transportation Fuels,* 1994) statement that tank trucks typically carry 5,000 to 10,000 gallons -- about 31,000 to 62,000 lbs of gasoline.

[c] These are total odometer miles during the year, and hence include empty backhauls. To calculate loaded ton-miles of travel, I divide this total truck-mileage by two, on the assumption that for half of the miles, petroleum trucks are unloaded.

[d] Calculated from the assumed weight in the class and the assumed weight when empty.

TABLE 32. CALCULATION OF ELECTRICITY AND FUEL USE IN SICS 517, 554, 55 (EXCEPT 554) AND 75, IN 1987

SIC: description	Electricity		Natural gas		Fuel oil	
	expense (10^6\$)[a]	*price (\$/kWh)*[c]	*expense (10^6\$)*[a,b]	*price (\$/SCF)*[d]	*expense (10^6\$)*[a,b]	*price (\$/gal)*[e]
517: Petroleum marketing	151	0.0600	84	0.00400	25	0.60
554: Service stations	666	0.0677	112	0.00563	33	0.71
55[f]: Motor vehicles, parts	750	0.0677	243	0.00563	73	0.71
75: Auto repair and service	467	0.0677	187	0.00563	56	0.71

[a] These data are from the Bureau of the Census' quinquennial surveys: data for SIC 517 are from the *1987 Census of Wholesale Trade, Subject Series, Measures of Value Produced, Capital Expenditures, Depreciable Assets and Operating Expenses* (1991); data for SICs 554 and 55 except 554 are from the *1987 Census of Retail Trade, Measures of Value Produced, Capital expenditures, Depreciable assets, and Operating Expenses* (1991); and data for SIC 75 are from the *1987 Census of Service Industries, Capital expenditures, Depreciable assets, and Operating Expenses* (1991).

The expenditure estimates published from these surveys are actual, direct payments for electricity and fuel; they do not include the cost of any electricity and fuel that was included in normal lease or rental payments or franchise fees. Therefore, the published expenditure estimates need to be scaled up to account for the use of electricity and fuel that was paid for in lease, rental, or franchise fees and hence did not show up in the published expenditures. Because the Census does not have any data on the cost of energy included in lease, rental, or franchise fees, this scaling must be done indirectly, as explained next.

The Census does have unpublished that allow one to calculate the ratio of: total operating expenses for all firms in the SIC of interest (that is, operating expenses of firms that paid for electricity and fuel, *plus* the operating expenses of firms whose electricity and fuel use was covered by lease, rental, or franchise fees) to the operating expenses of firms that reported only direct payments for electricity and fuel (Bureau of the Census, Business Division, data transmittal, 1993). Now, I assume that this ratio is equal to the ratio that I would really like to know, namely: payments for all electricity and fuel (including the cost of electricity and fuel covered in lease, rental, or franchise fees) to reported actual payments for electricity and fuel. Therefore, I multiply reported direct payments for electricity and fuel in each SIC by the ratio of total operating expenses of all firms to operating expenses of firms that reported direct payments for electricity and fuel, in each SIC.

[b] The Census shows only total expenditures for all fuels other than electricity; it does not distinguish natural gas from fuel oil. I use data from the EIA's *Manufacturing Energy Consumption Survey* to estimate the portion of fuel expenditures that is for natural gas, and the portion that is for fuel oil. In 1986, mercantile and service commercial buildings in the U.S. consumed 0.536 quads and 10.58-billion-dollars-worth of electricity, 0.332 quads and 1.61 billion-dollars-worth of natural gas, 0.105 quads and 0.489 billion-dollars-worth of fuel oil, 0.012 quads of district heat, and 0.017 quads of propane (EIA, *AER 1993*, 1994). Based on this, I assume that in 1987, 23% of the payment for "other fuels" as reported by the Census was for fuel oil, and that 77% was for natural gas.

The Census also provided information on operating expenses that included use of "fuels not applicable." I have assumed that this refers to highway fuels, which I wish to include in my totals, so I have estimated payments for these fuels and have included them in the totals shown for fuel oil.

[c] In 1987, the average electricity price in the U.S. in the commercial sector as a whole was $0.0708 / kWh, and in 1986 the average electricity price to mercantile and service commercial buildings specifically was $0.0686 / kWh (EIA, *AER 1993*, 1994; the figure for 1986 is from the Commercial Buildings Energy Consumption Survey, which was done in 1986 and 1989 but not 1987). The price to mercantile and service buildings in 1987 can be approximated as the price in 1986 multiplied by the ratio of the price to the commercial sector as a whole in 1987 to the price to the commercial sector as a whole in 1986. This results in $0.0677 / kWh, which I use as the average electricity price in SICs 554, 55 except 554, and 75.

I assume that the price to SIC 517 is between the commercial-sector average price of $0.0708 / kWh and the industrial-sector average price of $0.0477 / kWh (EIA, *AER 1993*, 1994).

[d] I estimate the average natural-gas price using the same data source (EIA, *AER 1993*, 1994) and methods that I used to estimate the average electricity price (footnote c). The relevant price data for natural gas are: $4.77 / 1000-SCF (Standard Cubic Feet) to the commercial sector in 1987 and $5.08 in 1986; $5.29 / 1000-SCF to mercantile and service buildings in 1986; and $2.94 / 1000-SCF to the industrial sector in 1987.

[e] I estimate the average fuel-oil price using the same data source (EIA, *AER 1993*, 1994) and methods that I used to estimate the average electricity price (footnote c).The relevant price data for fuel oil are: $0.803 / gallon for residential heating oil in 1987, and $0.836 in 1986; $0.685 / gallon for "fuel oil" sold to mercantile and service buildings in 1986 (I assume 140,00 BTU / gallon HHV); and $0.527 / gallon for No. 2 fuel oil sold from refiners to resellers in 1987, and $0.486 / gallon in 1986.

[f] Excluding SIC 554, which is covered separately.

TABLE 33. ENERGY USE PER UNIT OF ACTIVITY FOR PETROLEUM MARKETING, SERVICE STATIONS, AUTOMOBILE SERVICES, AND MOTOR-VEHICLE AND PARTS SALES

	Electricity	**Natural gas**	**Fuel oil**
energy use per gallon of fuel	*(kWh)*	*(SCF)*[a]	*(gallons)*
Petroleum marketing (SIC 517)[b]	0.0113	0.0947	0.0002
Service stations (based on SIC 554)[c]	0.1015	0.2045	0.0005
energy use per vehicle mile			
Motor vehicle and parts (based on SIC 55 except 554)[d]	0.0060	0.0232	0.00005
Auto repair, parking, service (SIC 75)[e]	0.0036	0.0173	0.00004

All values shown are equal to dollar expenditures on electricity or fuel divided by price (per kWh, SCF, or gallon) divided by total activity (miles or gallons). Expenditure and price data are from Table II above. Activity data are documented in the notes to this table.

[a] Standard Cubic Foot.

[b] I assume that electricity and fuel use at bulk-storage facilities is proportional to the amount of fuel handled. In 1987, SIC 517, petroleum bulk storage, sold 222.7 billion gallons of fuel (Bureau of the Census, *1987 Census of Wholesale Trade, Miscellaneous Subjects,* 1991). I assume that all highway fuels pass through a bulk-storage facility, and that no gallon of any fuel is sold twice within SIC 517. I also assume that SIC 517 handles only petroleum products, and that there is no bulk storage of highway fuels outside of SIC 517. With these assumptions, the amount of energy used at bulk storage facilities per unit gallon of highway fuel consumed by end users -- which is the number that I want -- equals the total amount of energy (of each kind) consumed in SIC 517 divided by the total amount of gallons sold in SIC 517.

The electricity and fuel-use and the gallon-sales data for SIC 517 are from the same general survey, but it appears that the definition of “petroleum bulk stations and terminals” used in the electricity and fuel-use part of the survey (Bureau of the Census*1987 Census of Wholesale Trade, Measures of Value Produced, Capital expenditures, Depreciable assets, and Operating Expenses,* 1991) is slightly different than the definition used in the gallon-sales part of the survey (Bureau of the Census, *1987 Census of Wholesale Trade, Miscellaneous Subjects,* 1991). Nevertheless, I use electricity and fuel use data from the *Measures of Value Produced...*report, and gallon data from the *Miscellaneous Subjects* report. I have scaled the reported gallon sales by the ratio of total sales to reported sales.

c I assume that electricity and fuel use at service stations is proportional to the amount of fuel dispensed. In 1987, service stations in SIC 554 sold 87.26 billion gallons of fuel (Bureau of the Census, *1987 Census of Retail Trade, Measures of Value Produced, Capital expenditures, Depreciable assets, and Operating Expenses*, 1991). The gallon-sales data and the electricity and fuel-use expenditure data are from the same survey (Bureau of the Census, *The 1987 Census of Retail Trade*, 1991*)* and pertain to the same population of service stations. However, businesses in SIC 554 sell more than just highway fuels, repair services, and automotive supplies: in 1987, food, drinks, drugs, household merchandise, and other non-automotive goods were slightly more than 10% of the sales in SIC 554 (Bureau of the Census, *1987 Census of Retail Trade, Merchandise Line Sales*, 1990). On the assumption that people would buy these non-automotive products elsewhere if they did not drive, I deduct the product's share of electricity and fuel usage, which I assume is equal to the products' share of total sales. Therefore, I allocate 90% of electricity and fuel use at service stations (SIC 554, which includes truck stops) to the 87 billion gallons of fuel sold in this SIC in 1987.

d I assume that the amount of electricity and fuel used at motor-vehicle dealerships and automotive parts stores is related indirectly to total vehicle miles of travel. (Energy use probably is more directly related to the total numbers of vehicles sold, but VMT in turn probably is related to vehicle sales, and in any case is easier to work with). The 1987 Census (Bureau of the Census, *1987 Census of Retail Trade, Measures of Value Produced, Capital expenditures, Depreciable assets, and Operating Expenses*, 1991) provides data which allow me to calculate electricity and fuel use in SIC 55 (except 554), but does not report total VMT or motor-vehicle sales. Therefore, I adjust the electricity and fuel consumption in SIC 55 (except 554) to approximate electricity and fuel use associated with the sale of all motor-vehicles and parts and supplies, and then divide by the 1.9212 trillion total VMT in 1987 (Federal Highway Administration, *Highway Statistics 1988*, 1989). I adjust electricity and fuel consumption by multiplying electricity and fuel consumption in SIC 55 (except 554) by the ratio of dollar sales of all automotive merchandise lines in all SICs (except 554) to dollar sales of all merchandise in SIC 55 (except 554). (1.034; Delucchi, 1996).

e I assume that electricity and fuel use associated with motor-vehicle services(such as automobile repair and parking) done in this SIC are related indirectly to vehicle miles of travel. (Energy use probably is more directly related to total expenditures on maintenance and repair, but VMT in turn probably is related to these expenditures, and in any case is easier to work with). I further assume that all motor-vehicle services that are not performed by establishments in SIC 55 are performed by establishments in SIC 75, and that establishments in SIC 75 perform only motor-vehicle services. With these assumptions, I can use electricity and fuel data for SIC 75 without adjustment. Thus, I divide the estimated electricity and fuel consumption in SIC 75 by the 1.9212 trillion total VMT in 1987 (Federal Highway Administration, *Highway Statistics 1988,* 1989).

This estimate does not include energy used for home garages, free parking, road lighting, in-house maintenance and repair by businesses, or public motor-vehicle agencies.

TABLE 34. EMISSION FACTORS FOR NATURAL GAS, LPG, AND DIESEL-FUEL FOR SPACE HEATING

A. EPA (1990, 1995 [AP-42]) EMISSION FACTORS FOR RESIDENTIAL FUEL USE

Pollutant	**Natural gas** (lbs / 10^6 SCF)		**LPG** (lbs / 10^3 / gal)		**Fuel oil** (lbs / 10^3 / gal)	
	EPA (1990)	*AP-42*[a]	*EPA (1990)*	*AP-42*[b]	*EPA (1990)*	*AP-42*[c]
CH_4	n.r.	3.7 / 2.3	n.r.	0.2	n.r.	1.8
N_2O	n.r.	n.r / 2.2[d]	n.r.	0.9[e]	n.r.	0.05[f]
Total NMOCs[g]	5.3	7.3 / 8.7	0.5	0.3[h]	0.70	0.7
CO	20	40 / 40	1.95	1.9	5.0	5.0
NO_x (NO_2)	100	94 / 94	7.5	14	18.0	18.0
PM	3.0	11.2 / 7.6[i]	1.85	0.4[j]	2.50	0.4[k]
PM_{10}	3.0	11.2 / 7.6[i]	1.85	n.r[l]	1.25	n.r[m]
$PM_{2.5}$	n.r.	11.2 / 7.6[i]	n.r.	n.r.	n.r.	n.r[m]
Rating[n]	n.r.	B-D	n.r.	E	n.r.	A[o]

B. EPA (1990, 1995 [AP-42]) EMISSION FACTORS CONVERTED TO GRAMS/10^6-BTU

Pollutant	**Natural gas**		**LPG**		**Fuel oil**	
	EPA (1990)	*AP-42*	*EPA (1990)*	*AP-42*	*EPA (1990)*	*AP-42*
CH_4	n.r.	1.6 / 1.0	n.r.	1.0	n.r.	5.9
N_2O	n.r.	n.r. / 1.0	n.r.	4.5	n.r.	0.2
Total NMOCs	2.3	3.2 / 3.8	2.5	1.5	2.3	2.3
CO	8.7	17.4 / 17.4	9.7	9.4	16.4	16.4
NO_x (NO_2)	43.4	40.8 / 40.8	37.2	69.5	58.9	58.9
PM	1.3	4.9 / 3.3	9.2	2.0	8.2	1.3
PM_{10}	1.3	4.9 / 3.3	9.2	n.r.	4.1	n.r.
$PM_{2.5}$	n.r.	4.9 / 3.3	n.r.	n.r.	n.r.	n.r.

C. EMISSION FACTORS ASSUMED IN THIS ANALYSIS (G/10^6 BTU)[p]

Pollutant	Natural gas	LPG	Fuel oil
CH_4	1.2	1.7	5.9
N_2O	1.0	1.0	1.0
Total NMOCs	2.3	2.5	2.3
CO	8.7	9.7	16.4
NO_x (NO_2)	43.4	37.2	58.9
SO_x	sulfur content[q]	sulfur content[q]	sulfur content[q]
PM	0.5	0.6	2.0
PM_{10}	0.5	0.6	1.1
$PM_{2.5}$	0.5	0.6	0.8

n.r. = not reported; OC = organic compound; NMOC = nonmethane organic compound.

[a] In the fifth edition of AP-42, before supplement D, the EPA (1995) reported CO, NO_x, PM, TOC, and CH_4 emission factors for residential furnaces, as well as for utility boilers and industrial boilers. In supplement D to the fifth edition, the EPA reported CO and NO_x emission factors for residential furnaces (and for utility and industrial boilers), but reported all other emission factors only for the generic category “NG combustion,” apparently because emissions other than CO and NO_x do not vary much with the heat input or type of combustion. In this table, the value before the slash is the pre-supplement D estimate specifically for residential combustion, and the value after the slash is the supplement-D estimate for residential combustion (in the case of CO and NO_x) or generic combustion (all other pollutants).

[b] Emission factors for commercial boilers with a heat input capacity between 0.3 and 10 million BTU/hr. AP-42 (EPA, 1995) does not report emission factors for residential-size furnaces burning LPG.

[c] Emission factors for residential furnaces.

[d] The EPA reports a value of 0.64 for low-NO_x burners..

[e] The EPA cites as its source a study of greenhouse-gas emissions in Norway.

[f] Citing a 1989 and a 1991 source, the EPA (1995, AP-42, prior to supplement E) reports a factor of 0.05 lbs / 10^3 gal (0.15 g / 10^6-BTU) for residential furnaces, and 0.11 lbs / 10^3 gal (0.33 g / 10^6-BTU) for utility boilers. The same factors are reported in Supplement E. Data reviewed in Delucchi and Lipman (1997) indicate that utility boilers have "direct" emissions in the range of 0.3 to 1.0 g / 10^6-BTU, or even higher. It is possible that controlled boilers emit more, and that some additional N_2O forms indirectly, in the atmosphere, from other combustion products.

It is not clear why residential furnaces should emit less N_2O than do utility boilers, especially since N_2O can increase with decreasing combustion temperature (Delucchi and Lipman, 1997), and residential furnaces presumably operate at much lower temperatures than do utility boilers.

[g] EPA (1990) reports VOCs; EPA (1995) reports NMOCs (or TOC and CH_4, from which NMOCs can be estimated).

[h] EPA (1995) reports TOC emissions of 0.5 and CH_4 emissions of 0.2 lbs / 10^3 gal.

[i] Total PM, equal to "filterable PM" plus "condensable PM". According to EPA, all PM from natural-gas combustion is $PM_{1.0}$ or less.

[j] The EPA implies that this is "filterable" PM only, and does not include "condensable" PM. In the case of natural gas combustion, condensable PM is three times filterable PM (EPA, 1995, p. 1.4-6).

[k] Filterable PM emissions from new residential furnaces. According to EPA, pre-1970s furnaces emit on the order of 3.0 lbs / 10^3 gal. Data for larger boilers indicate that condensable PM is a bit smaller than filterable PM; hence, total PM (filterable plus condensable) from residential furnaces probably is less than 1.0 lb / 10^3 / gal.

[l] EPA (AP-42) (1995) notes that "for natural gas, a fuel with similar combustion characteristics, all PM is less than 10 µm..." (p. 1.5-3).

[m] The EPA (AP-42) (1995) does not report a size distribution of PM from residential furnaces, but it does report the distribution from commercial boilers burning distillate fuel: 55% is PM_{10}, and 42% is $PM_{2.5}$.

[n] The EPA's (AP-42) (1995) rating of the quality of the emission-factor estimate. An "A" rating indicates high quality. See EPA (1995) for details.

[o] The PM emission factor is rated “B”, and the N_2O emission factor is rated “E”.

[p] Source: NMOC, CO, and NO_x estimates are from EPA (1990). CH_4 and PM estimates based on EPA (1995). N_2O based on Delucchi and Lipman (1997). See the text for more discussion.

[q] The sulfur content of natural gas and LPG is negligible. The EIA believes that #2 distillate fuel oil used for home heating has a sulfur content of about 0.3% by weight (Cogan, 1999). However, EIA data on the supply and disposition of low-sulfur distillate fuel in 1997 suggest that some of the fuel used for heating and cooking has a sulfur content of less than 0.05% (consumption in billions of gallons):

Sector	< 0.05% S by wt.	Total	Data source
Highway vehicles	28.6	28.6	EIA *FOKS 1997* (1998)
Commercial	1.0	3.3	“
Industrial	0.6	2.3	“
Residential	} 3.9	6.5	“
Off-highway, farm, other		12.0	“
All sectors	34.1	52.7	EIA, *PSA 1997* (1998)

(FOKS = Fuel Oil and Kerosene Sales; PSA = Petroleum Supply Annual). Assuming that the residential sector consumed 1 billion of low-sulfur distillate, I estimate that in 1997, about 20% of the distillate consumed in the residential and commercial sectors was low-sulfur (S = 0.03% by weight), and 80% was high sulfur (S = 0.3% by weight). I then assume that the share of low-sulfur increases by 1.5 absolute percentage points per year.

TABLE 35. ENERGY USE, LEAKS, AND BOIL-OFF ASSOCIATED WITH COMPRESSION OR LIQUEFATION OF GASEOUS FUELS

	CNG	LNG	CH_2	LH_2
Vehicular storage pressure (psi)[a]	3,000	n.a.	6,000	n.a.
BTUs process energy per BTU fuel, small scale, at refueling site (BTU_{PE}/BTU_{FO})[b]	0.022	0.200	0.069	0.310
BTUs process energy per BTU fuel, large scale, at central site (BTU_{PE}/BTU_{FO})[b]	n.a.	0.150	n.a.	0.260
Fraction of process energy from electricity[c]	1.00	0.00	electricity only	
Fuel leakage or boil-off, per fuel transfer, in a base year (% of output to consumers) (FLR)[d]	0.04%	2.00%	0.08%	4.00%
Number of transfers, small scale, at refueling site	1	1	1	1
Number of transfers, large scale, at central site[e]	n.a.	3	n.a.	3
Base year of fuel-leakage or boil-off (T_B)	1992	1992	1992	1992
The annual % change in leakage or boil-off (?FL)[f]	-1.0%	-3.5%	-1.0%	-4.0%
Of fuel boiled-off, the fraction reliquefied (FLR)[g]	n.a.	50%	n.a.	50%
Calculated BTUs energy per BTU of fuel to vehicles, small scale, at refueling site (BTU_{PE}/BTU_{FM})[h]	0.022	0.203	0.070	0.337
Calculated process BTUs per BTU fuel to vehicles, large scale at central site (BTU_{PE}/BTU_{FM})[h]	n.a.	0.187	n.a.	0.278

CNG = compressed natural gas; LNG = ;liquefied natural gas; CH_2 = compressed hydrogen; LH_2 = liquefied hydrogen; .a. = not applicable.

[a] My assumptions. For hydrogen, see the discussion in the text.

[b] Estimated or assumed as follows:

CNG: I have changed the estimate of the compression energy requirement for CNG from 0.05 BTU-electric/BTU-CNG to 0.022, on the basis of calculations of compressor-work requirements using engineering equations, and actual energy-use data from CNG stations operated by PG&E (1993) (cf. Table 3 of DeLuchi, 1991). Cost data reported in Powars et al. (1994) indicate that typical requirements are at least 0.02 BTU-electric/BTU-CNG.

LNG: DeLuchi (1991) and Powars et al. (1994) assume 0.20 for small-scale liquefaction at the site of refueling. Powars et al. (1994) assume that centralized liquefiers can be powered entirely by the pressure-reduction available at city gate, where the natural gas comes in at about 1000 psi. (Nimocks [1995] notes that in the U. S. 2 facilities do produce LNG from the pressure drop at the city gate.) However, with only a 1000 psi pressure drop, it must take something on the order of 20 SCF gas throughput to liquefy 1 SCF. If there is a competing use for the pressure-drop work, there might not be enough total pressure-drop-work available to liquefy an appreciable amount of gas. I have no idea how much pressure-drop work is available at city-gates throughout the U. S, and simply assume that the additional energy required for liquefaction at centralized plants on average is 0.15 BTU / BTU-LNG.

CH_2: Calculated with Eq. 101; see the discussion in the text.

LH_2: Appendix L of DeLuchi (1993) assumes that large-scale hydrogen liquefaction requires 0.26 BTU for every BTU produced, and Syed et al. (1998) confirm this figure. Berry (1996) states that small-scale liquefiers have been built, and indicates that they require at least 10% more energy (in relative terms) than do large-scale liquefiers. I assume 0.31 for small-scale liquefaction at the site of refueling.

[c] The model allows the user to specify whether natural gas liquefiers and compressors run off of electricity or off natural gas. The table shows the assumed share of electricity in the base case (the remainder being the share of natural gas). Hydrogen liquefiers and compressors are assumed to use electricity only.

[d] My assumptions. See the discussion in the text. The boil-off figures for LNG and LH_2 include gas that boils-off but is eventually re-liquefied rather than vented to the atmosphere.

[e] Plant to truck, truck to refueling station, station to vehicle.

[f] My assumptions, chosen to result in what appear to me to be reasonable long-run loss rates.

[g] My assumptions.

[h] Calculated with Eq. 100. See the discussion in the text.

TABLE 36. EMISSION REDUCTION FACTORS FOR CONTROLLED EMISSIONS FROM EACH POLLUTANT SOURCE

	Train	**Ship**	**Scraper**	**Loader**	**Off-road trucks**	**Farm tractor**	
	diesel	*fuel oil*	*diesel*	*diesel*	*diesel*	*gasoline*	*diesel*
NO_X	0.65	0.65	0.65	0.65	0.65	0.65	0.65
SO_X	1.00	1.00	1.00	1.00	1.00	1.00	1.00
PM	0.10	0.10	0.10	0.10	0.10	0.10	0.10

	Industrial boilers						
	coal	*petrol. coke*	*NG*	*refinery gas*	*crude oil*	*fuel oil*	*LPG*
NO_X	0.50	0.50	0.50	0.50	0.50	0.50	0.50
SO_X	0.10	0.10	0.10	0.10	0.10	0.10	0.10
PM	0.05	0.05	0.05	0.05	0.05	0.05	0.05

	Large engine	**Industrial engine**		**Building heaters**		**NG compressor**	
	diesel	*gasoline*	*diesel*	*NG*	*diesel*	*NG turbine*	*NG engine*
NO_X	0.65	0.65	0.65	0.50	0.50	0.50	0.50
SO_X	0.10	0.10	0.10	0.10	0.10	0.10	0.10
PM	0.10	0.10	0.10	0.05	0.05	0.05	0.05

Equal to the ratio of controlled emissions to uncontrolled emissions, for each emissions source. Control factors based on data in EIA, *Electric Power Annual 1995, Volume II* (1996); EPA, *National Air Pollutant Emissions Trends, Procedures Document* (1998); EPA *Compilation of Air Pollutant Emission Factors, Vol. I, Stationary Sources*, AP-42 (1995); DeLuchi et al. (1992). Factors for CO, CH_4, NMOCs, and N_2O are 1.0 -- i.e., no emission reduction due to controls.

TABLE 37. EMISSION FACTORS FOR COMPONENTS OF REFINERY GAS, RELATIVE TO FACTORS FOR NATURAL GAS, IN INDUSTRIAL BOILERS

	CH_4	LPG[a]	H_2S(b)	H_2(b)
Energy share-->	*0.329*	*0.641*	*0.005*	*0.025*
CH_4	1.0	0.1[d]	0.0	0.0
N_20	1.0	1.0	1.0	1.0
Exhaust OCs excluding HCHO	calculated as NMOCs + CH4 - HCHO - evaporative OCs			
Evaporative OCs	0.0	0.0	0.0	0.0
HCHO	1.0	1.0	0.0	0.0
Total NMOCs	1.0	2.0[d]	0.0	0.0
CO	1.0	1.0	0.0	0.0
NO_X (NO_2)	1.0	1.5[e]	1.0	1.0
SO_X (SO_2)	emissions calculated on the basis of the sulfur content of the refinery gas			
PM	1.0	1.0	0.0	0.0
PM_{10}	1.0	1.0	0.0	0.0
$PM_{2.5}$	1.0	1.0	0.0	0.0

[a] Propane, butane, and pentanes-plus.

[b] Hydrogen and hydrogen sulfide do not contain carbon and so cannot produce CH_4, HCHO, NMOCs, or organic particulate matter.

[c] The contribution of each constituent to the total energy of the refinery gas.

[d] Because LPG does not contain methane, it will emit less methane and more non-methane compounds than will natural gas, which is mainly methane.

[e] The EPA (1995, AP-42) states that NO_X emissions from propane/butane mixtures are 50% higher than NO_X emissions from natural gas.

TABLE 38. U. S. FEDERAL EMISSION STANDARDS FOR NON-ROAD COMPRESSION IGNITION (DIESEL) ENGINES (G/BHP-HOUR)

A. Tiers 1, 2, and 3

Engine power	Tier	Model yr.	NMHC+NO_X	CO	PM
25 < hp < 50	1	1999	7.1	4.1	0.60
	2	2004	5.6	4.1	0.45
50 < hp < 100	1	1998	6.9 (NO_X only)	none	none
	2	2004	5.6	3.7	0.30
	3	2008	3.5	3.7	deferred[a]
100 < hp < 175	1	1997	6.9 (NO_X only)	none	none
	2	2003	4.9	3.7	0.22
	3	2007	3.0	3.7	deferred[a]
175 < hp < 300	1	1996	6.9 NO_X; 1.0 HC	8.5	0.40
	2	2003	4.9	2.6	0.15
	3	2006	3.0	2.6	deferred[a]

Source: EPA (1997a) for Tier 1; Federal Register (1998) for Tiers 2 and 3. See also Table 1 in EPA (2002b).

[a] EPA deferred until its 2001 feasibility review the question of setting a Tier 3 PM standard, because it believed that by then it would have a better understanding of the technical possibilities (Federal Register, 1998). See part B of this table.

B. Proposed Tier 4 standards

Engine power	Model year	NMHC	NOx	PM
hp < 25	2008	none	none	0.30
25 < hp < 75	2008	3.5 NMHC + NOx (50-75 hp only)		0.22
	2013	3.5 NMHC + NOx (25-50 hp)		0.02
75 < hp < 175	2012	0.14 (50% of production)	0.30 (50% of production)	0.01
	2014	0.14 (100% of production)	0.30 (100% of production)	
175 < hp < 750	2011	0.14 (50% of production)	0.30 (50% of production)	0.01
	2014	0.14 (100% of production)	0.30 (100% of production)	
hp > 750	2011	0.14 (50% of production)	0.30 (50% of production)	0.01[a]
	2014	0.14 (100% of production)	0.30 (100% of production)	

Source: Federal Register (2003). Minor changes to CO standards also are proposed.

[a] 50% of a manufacturer's production must meet this standard in this year. In 2014, 100% of production must meet the standard.

TABLE 39. IN-USE EMISSION FACTORS AND FUEL CONSUMPTION FOR DIESEL-POWERED NONROAD FORKLIFT ENGINES, ASSUMED IN THIS STUDY

Engine hp[a]	Model yr.	Tier	BSFC[b]	Emission factors (g/bhp-hr)[c]					
			lb/hp-hr (%)	*CH_4*[d]	*NMHC*[d]	*CO*	*NO_x*	*PM*[e]	*N_2O*[f]
50 - 100 hp	pre-1988[g]	0	0.481 (0.268)	0.06 / 0.07	1.75 / 1.84	6.06 / 10.0	14.00 / 8.00	1.60 / 1.60	0.022
	1988 - 1997	0	0.481 (0.268)	0.08	2.09	8.06	8.55	1.47	0.022
	1998 - 2003	1	0.471 (0.274)	0.06	1.47	2.31	7.11	1.47	0.022
	2004 - 2007	2	0.462 (0.280)	0.05	0.83	2.31	5.36	0.30	0.022
	2008 - 2012	3	0.453 (0.285)	0.04	0.40	2.31	3.40	0.22	0.022
	2013+	4	0.444 (0.291)	0.04	0.40	2.31	3.40	0.02	0.022
100 - 175 hp	pre-1988[g]	0	0.433 (0.298)	0.06 / 0.07	1.75 / 1.84	6.06 / 10.0	14.00 / 8.00	1.60 / 1.60	0.022
	1988 - 1996	0	0.433 (0.298)	0.06	1.43	6.24	8.63	0.82	0.022
	1997 - 2002	1	0.424 (0.304)	0.05	0.83	2.31	7.11	0.82	0.022
	2003 - 2006	2	0.416 (0.311)	0.05	0.60	2.31	4.63	0.22	0.022
	2007 - 2013	3	0.408 (0.317)	0.05	0.40	2.31	2.88	0.10	0.022
	2013+	4	0.399 (0.323)	0.04	0.15	2.31	0.28	0.01	0.022

[a] I provide estimates for engines in the 50 to 175 hp range because the EPA's final Regulatory Impact Analysis of its new standards for nonroad diesel engines indicates that most standard diesel forklifts are in the range of 50 to 100 hp, and most rough-terrain diesel forklifts in the range of 50 hp to 175 hp (EPA, 1998b, pp. 8, 11, and 12, Table 2-5). When I run the LEM, I use the estimates for the 50 to 100 hp class.

[b] Brake-specific fuel consumption, in pounds of fuel per brake horsepower hour. The values shown in parentheses are my estimate of the corresponding thermal efficiency (higher heating value [HHV]), calculated assuming that diesel fuel contains 3192 g/gal and 138,700 BTU/gal. The estimates of the bsfc of pre-1988 engines are from Beardsley and Lindhjem (1998b), and are equal to the steady-state bsfc (in their Table1) multiplied by an in-use adjustment factor of 1.18 (in their Table C-2.). I assume that beginning with Tier 1, the bsfc decreases by 2% for each Tier.

[c] My estimates of emissions of CO, NO_X, and HCs at Tiers 0 to 3 are based on Beardsley and Lindhjem (1998b). The emission factors for 1988 and later model years are equal to the steady-state emission factors shown in their Table 1 multiplied by the "in-use adjustment factors" for backhoes and loaders shown in their Table C-2. (Table C-2 has adjustment factors for three general categories of engines. Table C-4 shows which of the three general categories applies to each specific type of engine. According to their Table C-4 , the adjustment factors for backhoes and loaders apply to forklifts.) The in-use adjustment factors are: 2.19 for HC, 2.31 for CO, 1.03 for NO_X, and 2.04 for PM.

My estimates of emissions of CO, NO_X, and HCs at Tier 4 are based on the Tier 4 standards shown in Table 38. My estimates of emissions of other pollutants are explained in other footnotes to this table.

In the LEM, SO_X emissions are calculated on the basis of the sulfur content of the fuel. EPA does the same, except in more detail: they deduct sulfur emitted as PM or in unburned fuel before they calculate SO_2 (Beardsley and Lindhjem, 1998b).

[d] Beardsley and Lindhjem (1998b) and the NEVES (EPA, 1991) show only a total "HC" emission factor. On the basis of emissions tests for diesel motor vehicles, summarized in Delucchi and Lipman (1997) and Lipman and Delucchi (2002) I assume that for pre-1988 and 1988-1997 vehicles,3.5% of the HC is CH_4, and 96.5% is NMHC. However, I assume that the CH_4 percentage increases as total HC emissions decrease for Tiers 1, 2, 3, and 4.

The EPA does have a document that estimates the methane fraction of total HC emissions from nonoad engines (EPA, 2003). However, EPA states that its estimates were based on a few tests of *highway* vehicles. Because Lipman and Delucchi (2002) provide a comprehensive overview of methane emissions from conventional and alternative-fuel highway vehicles, we base our estimates here directly on their work.

The NEVES (EPA, 1991) data, which I use for the pre-1988 emissions, show exhaust, crankcase, evaporative, and refueling HC emissions, and aldehydes. I added all of the NEVES HC emission factors, plus aldehydes, to get a total pre-1988 HC factor, then divided total HC into CH_4 and NMHC.

[e] The PM emission factors assumed in the EPA's NONROAD model do not account for the Tier 2 PM standards (EPA, 998b; Beardsley and Lindhjem, 1998b) (there is no Tier 1 PM standard for the size classes shown; see Table 38 of this report). EPA assumes that emissions remain at the level estimated for the uncontrolled 1988 - 1996/7 model year (Beardsley and Lindhjem, 1998b). This does not seem reasonable, so instead, I assume that in-use emissions from engines subject to the Tier 1 to 4 standards are exactly equal to the Tier standard (Table 38).

[f] Based on the emission rates estimated for diesel highway vehicles (Delucchi and Lipman, 1997).

[g] Table 1 of Beardsley and Lindhjem (1998b) states that emissions from pre-1988 engines "vary by application," and are given in the NEVES (EPA, 1991). Therefore, I have shown the NEVES emission factors for forklifts (EPA, 1991, Table 2.07). The factors given in the NEVES and used here already have been adjusted for "in-use transient" effects.

The NEVES study gives values for "forklifts" and for "rough terrain forklifts". I show emission factors for both: the first value, before the slash (/) is for forklifts, and the value after the slash is for rough-terrain forklifts.

TABLE 40. IN-USE EMISSION LIFETIME AVERAGE FACTORS AND FUEL CONSUMPTION FOR SPARK-IGNITION NONROAD FORKLIFT ENGINES, ASSUMED IN THIS STUDY

Fuel	**Model year**[a]	**Engine efficiency**[c]	**Emission factors (g/bhp-hr)**[b]						
			CH_4	*Exh. NMHC*	*Evap.*	*CO*	*NO_x*	*PM*	*N_2O*
RFG	pre-2004	0.254	0.30	10.0	3.01	280	6.0	0.100	0.024
	2004-2006	0.262	0.07	1.0	1.51	40	2.0	0.050	0.088
	2007+	0.270	0.06	0.8	0.45	4.0	1.2	0.035	0.088
LPG	pre-2004	0.254	0.30	3.0	calc.	60	12	0.025	0.024
	2004-2006	0.275	0.07	0.50	calc.	24	1.8	0.013	0.088
	2007+	0.283	0.06	0.40	calc.	2.4	1.1	0.009	0.088
CNG	pre-2004	0.254	3.00	1.5	calc.	60	12	0.020	0.024
	2004-2006	0.275	0.85	0.25	calc.	24	1.8	0.010	0.066
	2007+	0.283	0.75	0.20	calc.	2.4	1.1	0.007	0.66

RFG = reformulated gasoline; LPG = liquefied petroleum gases; CNG = compressed natural gas; Exh. = exhaust emissions; Evap. = evaporative NMHC emissions.

[a] See the discussion in the text. I assume that 2004 and later model-year engines will have 3-way catalytic converters with closed-loop emission controls and electronic fuel injection.

[b] Emission factors are estimated as follows:

CH_4: My assumptions are based on data on emissions frrom highway vehicles. Partly on the basis of data discussed in Delucchi and Lipman (1997) and Lipman and Delucchi (2002), I assume that gasoline and LPG vehicles with a 3-way catalyst (model year 2004 and later) emit about as much methane as do diesel vehicles, but that gasoline and LPG vehicles without a 3-way catalyst (pre-2004) emit considerably more than do diesel vehicles. CNG vehicles emit about an order of magnitude more than do gasoline and LPG vehicles (Delucchi and Lipman, 1997). I assumed also that the 3-way catalyst, introduced with the 2004 model year, is more effective at controlling NMHC than CH_4, with the result that CH_4 is a larger percentage of total HC for the post-2004 engines than for the pre-2004 engines.

Exhaust NMHC, CO, and NO_x: I assume that the lifetime average emission factors for large SI engines, implied by the NONROAD model (Stout, 1999a), and the average in-use emission factors specifically for 4-stroke SI forklifts, in the NEVES (EPA, 1991), apply to pre-2004 RFG and LPG engines. (Both sets of emission factors are shown in the text.). I use my judgment in weighing the generic but recent NONROAD lifetime average emission factors with the forklift-specific but older NEVES emission factors.

I assume that post-2004 model-year RFG engines are subject to the new Federal standards discussed in th etext, and estimate emissions with respect to these standards. On the basis of results of tests of automobiles (Delucchi, 1999), I assume that LPG forklifts that use the same post-2004 control technology as gasoline forklifts will have 50% of the NMHC emissions, 90% of the NO_X emissions, and 60% of the CO emissions of post-2004 gasoline forklifts. (See also McGlinchey and Jaques, 2003).

For CNG, in all cases, I again follow the results of automotive tests (Delucchi, 1999) and assume that CNG engines emit the same NO_X and CO but half the NMHC of LPG engines.

<u>Evaporative NMHC:</u> On the basis of the information discussed in the text, I assume 2.0 g/day/gallon-tank-capacity for diurnal, hot soak, resting, and running losses, for large uncontrolled engines. Assuming for forklifts an 18-gallon tank, 1800 hours per year, a maximum power of 75 hp, and an average usage of 30% of maximum power (EPA, 1991, 2002e; Stout, 1999a; Lindhjem and Beardsley, 1998), this 2.0 g/day/gallon-tank-capacity corresponds to 0.32 g/bhp-hr. For crankcase and refueling emissions I adopt the NEVES estimate of 2.69 g/bh-hr. The total is thus 3 g/bhp-hr. I assume that this applies to pre-2004 gasoline engines. I assume that 2004 to 2006 MY gasoline engines emit 50% less, and that post-2007 MY engines emit 85% less.

Evaporative or leakage emissions from CNG and LPG forklifts are calculated (“calc.” in the table) assuming the same leakage percentages as for light-duty highway vehicles (see the section on motor-vehicle emissions in this report). This generally results in lower “evaporative” emissions from CNG and LNG forklifts than from gasoline forklifts. This is consistent with the discussion in Delucchi (1999) and the Federal Register (2002). (The NEVES assumed the same factor or LPG as for gasoline [EPA, 1991, Table 2.07], but this does not seem reasonable.)

<u>PM:</u> For pre-2004 gasoline engines, I assume the NONROAD factors implied at mid-life (see the text). I assume that 2004-2006 model-year gasoline engines have 50% lower emissions, and that post 2007 engines have 65% lower emissions. For LPG and CNG, I apply the LPG/gasoline or CNG/gasoline PM emission ratios implied by tests on automobiles (Delucchi,1999).

<u>N_2O:</u> My assumptions are based on data on emissions from highway vehicles. From the available tests on diesel HDVs, gasoline LDVs with a catalyst, and gasoline LDVs without a catalyst (Lipman and Delucchi, 2002; Delucchi and Lipman, 1997), I infer that gasoline engines without a catalyst emit about the same as diesel engines (maybe slightly more), and gasoline engines with a 3-way catalytic converter emit several times more than do diesel engines. (As discussed in Delucchi and Lipman [1997], the 3-way catalyst causes N_2O emissions to increase.) The limited data on N_2O emissions from AFVs with 3-way catalysts suggest that LPG vehicles emit about the same as, and CNG vehicles slightly less than, do gasoline vehicle (Delucchi and Lipman, 1997).

$\underline{SO_X:}$ SO_X emissions are not shown because they are calculated in the model on the basis of the sulfur content of the fuel.

[c] The NEVES study (EPA, 1991, Table I-15, p. I-52) reports the specific fuel consumption of gasoline SI forklifts to be 0.08 gallons / hp-hr, which indicates a thermal efficiency (brake-hp / fuel-hp) of about 25% (HHV). (The NONROAD model assumes that LPG and CNG industrial engines -- which presumably include forklift engines -- consume 0.70 g / hp-hr, but this must be a misprint.) The current version of NONROAD assumes 0.605 lbs / bh-hr for large gasoline 4-stroke SI engines, and 0.507 lbs / bhp-hr for LPG (EPA, 2002h). The figure for gasoline corresponds to about 0.10 gallons / bhp-hr or only 20% efficiency. I adopt the NEVES figure for forklifts, and assume that it applies to pre-2004 engines. I assume that post-2004 engines are 3% more efficient, and that post 2007 engines are a further 3% more efficient (in relative terms).

Delucchi (1999) assumes that 1995 model-year LPG and CNG automotive engines have the same efficiency as their gasoline counterparts, but that future LPG and CNG engines could be as much as 20% more thermally efficient than their gasoline counterparts. Given that forklift engines are similar to automotive engines (see the brief discussion in the text), I assume that pre-2004 LPG and CNG forklift engines have the same efficiency as their gasoline counterparts, but that post-2004 LPG and CNG forklift engines are 5% more efficient than their gasoline counterparts. Note that fuel consumption estimates used in NONROAD (cited above) imply that LPG engines are about 10% more efficient than gasoline engines.

TABLE 41. EMISSION FACTORS FOR BLACK CARBON AND ORGANIC-MATTER AEROSOLS

Fuel	PM2.5	BC2.5	OM2.5	Notes
Conventional gasoline	0.85	0.30	0.50	PART5 manual (EPA, 1995) and Radian (1990) estimate PM2.5 = 0.89 for gasoline; Durbin et al. (1999) measured PM2.5 = 0.92 for highest-emitting model years, and 0.74 for lowest-emitting model years; B&B best estimate is BC2.5 = 0.27 and OC2.5 = 0.48; Bond et al. (2003) assume BC1.0 = 0.34 and OC1.0 = 0.36; Jacobson (2002) assumes OM = 1.3 OC for all f.f combustion.
Reformulated gasoline	0.85	0.30	0.50	See conventional gasoline.
Diesel vehicles and other except heating	0.93	0.60	0.27	PART5 manual (EPA, 1995) and Radian (1990) estimate PM2.5 = 0.92 for diesel trucks; Durbin et al. (1999) measured PM2.5 = 0.95 for LD diesel vehicles; B&B best estimate is BC2.5= 0.48 and OC2.5 = 0.29; Bond et al. (2003) assume BC1.0 = 0.66 and OC1.0 = 0.21; Jacobson (2002) assumes OM = 1.3 OC for all f.f. combustion.
Diesel -- heating	0.90	0.50	0.30	Assume similar to diesel vehicles.
Fischer-Tropsch diesel	0.93	0.60	0.27	Assume same as petroleum diesel for vehicles.
Soydiesel	0.93	0.60	0.27	Assume same as petroleum diesel for vehicles.
Heavy fuel oil, crude oil, engine oil	caclulated (result is about 0.55)	0.08	0.05	PM2.5 calculated on basis of AP-42 emission factors for fuel-oil utility boilers; B&B cite estimate of BC2.5 = 0.074 and OC2.5 = 0.045 for fuel oil; Bond et al. (2003) assume BC1.0 = 0.08 and OC1.0 = 0.03 for heavy fuel oil; Jacobson (2002) assumes OM = 1.3 OC for all f.f. combustion.
Methanol (pure)	0.95	0.10	0.60	Assume similar to but slightly higher than NG, because single-carbon compound, but a liquid.
Generic solvents, paint	0.00	0.00	0.00	Assume zero PM emissions from this source.
CNG or LNG for LDVs	1.00	0.07	0.65	Assume same as processed NG, below.
Processed NG, raw NG, refinery gas, coalbed gas (combustion emissions)	calculated (result is 1.00)	0.07	0.65	PM2.5 calculated on basis of AP-42 emission factors for NG utility boilers; B&B cite estimate of BC2.5 = 0.067 and OC2.5 = 0.85 for natural gas appliances; Bond et al. (2003) assume BC1.0 = 0.06 and OC1.0 = 0.50 for all NG combustion; Jacobson (2002) assumes OM = 1.3 OC for all f.f. combustion.
Natural gas --heating	1.00	0.07	0.65	Assume same as processed NG, above.
Natural gas as feedstock for H_2, FTD, and MeOH process-area emissions	0.85	0.10	0.70	Assume same as processed NG, above.

Coal, utility boiler	calculated (result is about 0.22)	0.05	0.05	PM2.5 calculated on basis of AP-42 emission factors for coal utility boilers; B&B best estimate is BC2.5 = 0.051 and OC2.5 = 0.069 for utility and industrial coal combustion; Bond et al. (2003) assume BC1.0 = 0.006 and OC1.0 = 0.00 for pulverizedhard coal, but say emissions can be higher with poor mixing; Jacobson (2002) assumes OM = 1.3 OC for all f.f. combustion.
Coal, industrial boiler	calculated (result is about 0.06)	0.05	0.05	PM2.5 calculated on basis of AP-42 emission factors for coal industrial boilers; for rest, see "coal, utility boiler."
Coke combustion	0.45	0.50	0.35	Assume coke combustion PM is smaller than coal combustion PM. Bond et al. (2003) assume BC1.0 = 0.48 and OC1.0 = 0.34 for coke ovens.
Biomass fuel combustion	calculated (result is 0.78)	0.05	0.28	PM2.5 calculated on basis of AP-42 emission factors for coal utility boilers; B&B cite an estimate that BC2.5 = 0.031 and OC2.5 = 0.071; Bond et al. (2003) assume PM1.0 = 0.86, BC1.0 = 0.05, and OC1.0 = 0.20 for wood-fired boilers; OM is larger than OC.
Biomass burning (agricultural residues)	0.50	0.05	0.40	Assume aerosols from residue burning generally are larger than aerosols from bio-fuel combustion (because of poorer combustion, lower quality fuel); Jacobson (2002, 2003) assumes that OM:BC= 8:1 for biomass burning.
Ethanol (pure)	0.85	0.15	0.65	Assume similar to but slightly higher than NG.
LPG	1.00	0.10	0.60	Bond et al. (2003) assume emissions from LPG are similar to emissions from kerosense. However, emissions from propane combustion typically are more similar to emissions from methane combustion.
Hydro, nuclear, solar	0.00	0.00	0.00	No PM emissions from these sources.
Refinery process areas (mainly FBCC)	0.45	0.01	0.01	Radian (1990) reports PM2.5 is about 0.45 for catalytic crackers, and that EC (similar to BC2.5) and OC2.5 are less than 0.01.
Ethanol production (corn or wood) process area	0.60	0.10	0.40	No data. My assumptions.
Methanol production (coal or wood) process area	0.60	0.10	0.40	No data. My assumptions.
Syncrude from coal process area	0.50	0.25	0.55	No data. My assumptions.

Steel coking from coal	0.50	0.10	0.10	AP-42 shows PM2.5 from coke manufacturing covers a wide range, depending on process stage and controls. No other data (but see "steels (process areas)" below.)
Biodiesel process area	0.60	0.10	0.40	No data. My assumptions.
Vehicle assembly process area	0.50	0.10	0.10	No data. My assumptions.
Steels (process areas)	0.50	0.10	0.10	Radian (1990) shows PM2.5 = 0.27 for steel sinter plant and iron-ore dust sinter, 0.39 for open hearth furnace, 0.65 for oxygen furnace, and 0.75 for steel electric arc furnace (for recycled steel). No data on BC or OC.
Plastics (process areas)	0.50	0.15	0.25	No data. My assumptions.
Aluminum (process areas)	0.40	0.05	0.20	Radian shows PM2.5 = 0.40, EC = 0.03, and OC = 0.04 – 0.28 for aluminum processing and aluminum reduction potline. AP-42 showss PM2.5 is 0.17 to 0.40 for various processes.
Generic materials (process areas)	0.50	0.15	0.25	No data. My assumptions.

PM2.5 = mass emssions of $PM_{2.5}$ divided by mass emissions of PM; PM1.0 = mass emissions of $PM_{1.0}$ divided by mass emissions of PM_{10}; BC2.5 = mass emssions of $BC_{2.5}$ divided by mass emissions of $PM_{2.5}$; OM2.5 = mass emssions of $OM_{2.5}$ divided by mass emissions of $PM_{2.5}$; OC2.5 = mass emssions of $OC_{2.5}$ divided by mass emissions of $PM_{2.5}$; BC1.0 = mass emssions of $BC_{1.0}$ divided by mass emissions of $PM_{1.0}$; OC1.0 = mass emssions of $OC_{1.0}$ divided by mass emissions of $PM_{1.0}$; B&B = Battye and Boyer (2002); BC = black carbon; EC = elemental carbon; OC = organic carbon; OM = organic matter; FTD = Fischer-Tropsch diesel; LPG = liquefied petroleum gases; LDV = light-duty vehicle;NG = natural gas; CNG = compressed natural gas; LNG = liquefied natural gas; H_2 = hydrogen; FBCC = fluidized bed catalytic combustion; MeOH = methanol.

TABLE 50. CALCULATED WEIGHT, EFFICIENCY, AND EMISSIONS OF VEHICLES

SPREADSHEET TABLES NOT AVAILABLE IN THIS VERSION

TABLE 51A. ENERGY INTENSITY OF FUELCYCLES: BTUS OF PROCESS ENERGY CONSUMED PER NET BTU OF FUEL TO END USERS

SPREADSHEET TABLE NOT AVAILABLE IN THIS VERSION

TABLE 51B. ENERGY CONSUPMTION OF FUELCYCLES: BTUS OF PROCESS ENERGY CONSUMED PER MILE OF TRAVEL BY VEHICLES (U. S. 2015)

Fuel -->	**CG**	**Hwy diesel**	**F-T Diesel**	**SD100**	**LPG**	**LPG**	**CNG**
Feedstock ---->	*oil*	*oil*	*NG*	*soy*	*NG*	*oil*	*NG*
End use	4,841	24,600	24,600	27,087	4,599	4,599	4,645
Fuel dispensing	9	45	43	56	8	8	103
Fuel distribution, storage	37	189	310	265	33	33	212
Fuel production	732	1,901	39,301	10,521	120	241	81
Feedstock transmission	53	304	431	481	0	55	0
Feedstock recovery	237	1,363	1,523	5,369	118	248	114
Ag. chemical manufacture	0	0	0	1,690	0	0	0
Total	*5,908*	*28,403*	*66,208*	*45,470*	*4,879*	*5,185*	*5,156*

Fuel -->	**CH2**	**M100**	**M100**	**M100**	**SCG**	**E100**	**E100**
Feedstock ---->	*NG*	*NG*	*coal*	*wood*	*wood*	*wood/grass*	*corn*
End use	4,145	4,396	4,396	4,396	4,645	4,394	4,394
Fuel dispensing	354	16	16	16	103	12	12
Fuel distribution, storage	0	116	89	76	191	64	58
Fuel production	995	2,129	1,950	3,321	1,741	4,192	2,198
Feedstock transmission	224	71	48	77	65	76	121
Feedstock recovery	209	253	45	136	115	160	364
Ag. chemical manufacture	0	0	0	38	32	322	614
Total	*5,927*	*6,981*	*6,544*	*8,061*	*6,893*	*9,221*	*7,760*

Note: diesel, F-T diesel, and SD100 are used in HDVs; all other fuels are used in LDVs.

TABLE 52A. TYPE OF PROCESS ENERGY USED AT EACH STAGE OF THE FUELCYCLE: FEEDSTOCKS

SPREADSHEET TABLE NOT AVAILABLE IN THIS VERSION

TABLE 52B. TYPE OF PROCESS ENERGY USED AT EACH STAGE OF THE FUELCYCLE: FUELS

SPREADSHEET TABLE NOT AVAILABLE IN THIS VERSION

TABLE 53A. EFFICIENCY OF ELECTRICITY GENERATION, BY FUEL TYPE (U.S. 2015)

Coal	Oil	Gas boiler	Gas turbine	Nuclear	MeOH	H2	Biomass
0.328	*0.346*	*0.481*	*0.481*	*n.a.*	*0.481*	*0.481*	*0.264*

TABLE 53B. SOURCE OF ELECTRICITY, BY TYPE OF GENERATING PLANT, FOR VARIOUS PROCESSES (YEAR 2015 EXCEPT AS INDICATED)

	Coal	Oil	Gas boiler	Gas turbine	Nuclear	MeOH	H2	Biomass	Hydro	Other
Recharging EVs	0.636	0.200	0.006	0.146	0.011	0.000	0.000	0.000	0.000	0.000
Oil refining and NGL plants	0.282	0.086	0.237	0.082	0.203	0.000	0.000	0.005	0.094	0.011
LDV manufacture	0.343	0.065	0.074	0.052	0.266	0.000	0.000	0.008	0.178	0.014
HDV manufacture	0.490	0.062	0.051	0.021	0.249	0.000	0.000	0.005	0.110	0.013
Uranium enrichment	0.660	0.008	0.012	0.008	0.241	0.000	0.000	0.000	0.070	0.002
Starch/ethanol plant	0.719	0.009	0.031	0.010	0.203	0.000	0.000	0.010	0.010	0.007
Wood/ethanol plant, displaced power	0.496	0.007	0.166	0.137	0.095	0.000	0.000	0.018	0.073	0.007
Aluminum production	0.338	0.003	0.041	0.022	0.033	0.000	0.000	0.001	0.562	0.001
Compression of natural gas	0.496	0.007	0.166	0.137	0.095	0.000	0.000	0.018	0.073	0.007
Compression or liquefaction of hydrogen	0.496	0.007	0.166	0.137	0.095	0.000	0.000	0.018	0.073	0.007
Water electrolysis	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	1.000	0.000
Rail transit	0.496	0.007	0.166	0.137	0.095	0.000	0.000	0.018	0.073	0.007
National average power	0.496	0.007	0.166	0.137	0.095	0.000	0.000	0.018	0.073	0.007

Values for oil refining and NG processing, auto manufacture, uranium enrichment, and corn-to-ethanol production same as in DeLuchi (1991). See the text for an explanation of the changes for recharging EVs and generic power.

TABLE 53C. FUELCYCLE CO_2-EQUIVALENT EMISSIONS FROM POWER PLANTS (BEST CEFS) (U. S. 2015)

g/million-BTU-input	*Coal*	*Fuel oil*	*NG/boiler*	*NG/turbine*	*Nuclear**	*Methanol*	*Hydrogen*	*Biomass*	*Hydro**
Aldehydes (as HCHO)	0.051	0.13	n.e.	n.e.	0.00	n.e.	0.00	n.e.	0.00
NMOC	1.40	2.30	3.87	1.92	0.00	4.00	0.00	56.19	0.00
NMOC - C	0.84	1.96	3.10	1.54	0.00	1.60	0.00	33.71	0.00
Ozone-weighted NMOC	0.91	1.15	1.55	0.77	0.00	1.60	0.00	36.52	n.e.
CH4	0.93	0.85	1.02	10.89	0.00	0.70	0.00	0.04	87.90
CO	11.69	15.15	24.01	49.90	0.00	15.00	0.00	0.09	0.00
N2O	0.93	0.33	0.63	2.00	0.00	1.00	1.00	0.02	0.00
NOx as NO2	159.13	44.69	25.12	62.63	0.00	62.76	72.17	52.21	0.00
SOx	222.22	167.06	0.16	0.16	0.00	0.65	0.00	24.45	0.00
PM	20.34	16.73	3.38	19.01	0.00	2.00	0.00	0.40	0.00
PM10	9.33	12.31	3.38	19.01	0.00	n.e.	0.00	0.36	0.00
PM2.5	4.30	9.41	3.38	19.01	0.00	n.e.	0.00	0.31	0.00
g-CO2-eq/MMBTU-input	*Coal*	*Fuel oil*	*NG/boiler*	*NG/turbine*	*Nuclear**	*Methanol*	*Hydrogen*	*Biomass*	*Hydro**
CO2 from combustion	94,940	75,002	53,326	53,220	0	63,782	0	0	1,465
Other gases from combustion	(9,092)	(6,060)	492	1,433	0	649	410	(890)	1,410
Subtotal from combustion	*85,848*	*68,942*	*53,818*	*54,654*	*2,109*	*64,431*	*410*	*(890)*	*2,875*
Upstream fuelcycle emissions	4,154	11,201	9,328	9,328	3,999	26,186	4,011	9,673	0

Total fuelcycle emissions	*Coal*	*Fuel oil*	*NG/boiler*	*NG/turbine*	*Nuclear**	*Methanol*	*Hydrogen*	*Biomass*	*Hydro**
g-CO2-eq/MMBTU-generated	274,165	231,444	131,363	133,101	6,108	188,510	9,198	33,333	2,875
g-CO2-eq/MMBTU-delivered^	298,939	252,503	143,720	145,609	7,573	205,836	10,932	37,166	4,058
g-CO2-eq/kWh-delivered	***1,020***	***862***	***490***	***497***	***26***	***702***	***37***	***127***	***14***
% change vs. coal plant	n.a.	-15.5	-51.9	-51.3	-97.5	-31.1	-96.3	-87.6	-98.6

"Best CEFs" are my best estimates of CO2-equivalency factors, as distinguished from the IPCC GWPs.

*Units in nuclear-power and hydro-power case are g/MMBtu of net power generated.

^Assuming a 92% distribution efficiency in the U. S., and 0.011 g/kWh-N_2P due to the corona discharge from power transmission lines.

TABLE 54. CO_2-EQUIVALENT EMISSIONS PER UNIT OF ENERGY DELIVERED TO END USERS, BY STAGE AND FEEDSTOCK/FUEL COMBINATION (G/10^6-BTU)

SPREADSHEET TABLE NOT AVAILABLE IN THIS VERSION

TABLE 55. EMISSIONS OF INDIVIDUAL POLLUTANTS PER UNIT OF ENERGY DELIVERED TO END USERS, BY STAGE AND FEEDSTOCK/FUEL COMBINATION (G/10^6-BTU)

SPREADSHEET TABLE NOT AVAILABLE IN THIS VERSION

TABLE 56. TOTAL EMISSIONS OVER THE WHOLE UPSTREAM FUELCYCLE, PER UNIT OF ENERGY DELIVERED TO END USERS, BY POLLUTANT AND FEEDSTOCK/FUEL COMBINATION (G/10^6-BTU) (BEST CEFS) (U. S. 2015)

Fuel ------>	**Coal**	**CG**	**RFG**	**Hwy diesel**	**F-T diesel**	**Fuel oil**	**Still gas**	**Coke**	**LPG**	**LPG**	**LPG**	**C**
Feedstock ------>	*coal*	*oil*	*oil*	*oil*	*NG*	*oil*	*oil*	*oil*	*oil*	*NGL57*	*NG*	
Carbon dioxide (CO2)	1,562	18,523	18,643	13,434	20,411	10,807	4,890	7,276	11,029	7,623	5,006	1
Nonmethane organic compounds (NMOCs)	1.0	42.5	35.6	12.4	8.5	10.7	7.0	9.2	19.7	13.2	8.3	
Methane (CH4)	138.7	212.3	209.8	204.5	192.2	123.3	161.0	139.7	189.5	142.7	106.8	3
Carbon monoxide (CO)	9.4	63.5	56.0	58.2	51.6	54.8	39.7	59.2	50.9	36.3	25.0	
Nitrous oxide (N2O)	0.0	1.1	0.6	0.8	1.0	0.5	0.2	0.3	0.5	0.3	0.2	
Nitrogen oxides (NO2)	12.0	73.6	72.6	67.2	77.9	66.9	43.3	64.5	62.2	45.8	33.3	
Sulfur oxides (SOx)	2.8	57.4	48.0	54.9	10.2	49.7	33.9	50.5	51.1	26.3	7.2	
Particulate matter (BC+OM)	0.2	2.1	2.1	1.7	1.5	1.6	1.1	1.6	1.6	1.2	0.9	
HFC-134a (mg)	0.2	0.5	0.5	0.5	0.4	0.5	0.1	0.2	0.4	0.4	0.3	
CO2-equivalent GHG emissions	4,083	23,946	24,246	18,092	26,453	14,036	8,263	10,692	15,172	11,231	8,203	1

Fuel ------>	**Nuclear #**	**CH2**	**CH2**	**MeOH**	**MeOH**	**MeOH**	**EtOH**	**EtOH**	**SCG**	**Biodiesel**	**Gra**
Feedstock ------>	*uranium*	*water*	*NG*	*NG*	*coal*	*wood*	*corn*	*(W0/G100)*	*wood*	*soy*	*gras*
Carbon dioxide (CO2)	4,032	19,635	94,562	23,714	79,831	26,411	94,710	39,488	18,542	165,376	20,5
Nonmethane organic compounds (NMOCs)	0.6	1.0	8.6	15.4	95.2	24.4	242.1	25.8	11.4	174.3	5.1
Methane (CH4)	8.5	141.0	392.8	184.5	225.0	63.5	201.3	85.1	117.0	292.4	31.
Carbon monoxide (CO)	10.5	8.9	63.4	49.1	43.2	157.5	400.0	154.7	460.2	1,588.8	47.
Nitrous oxide (N2O)	0.1	0.6	1.6	1.0	2.2	4.0	65.7	29.6	3.3	283.4	14.
Nitrogen oxides (NO2)	10.0	39.4	125.0	111.5	100.2	161.9	761.4	517.4	120.4	3,270.0	239
Sulfur oxides (SOx)	8.4	33.2	51.3	20.4	55.2	27.7	66.0	4.3	21.1	73.8	4.2
Particulate matter (BC+OM)	0.1	0.7	2.8	1.5	3.6	7.0	16.5	9.8	5.8	72.1	1.5
HFC-134a (mg)	0.1	0.2	1.5	0.8	1.4	4.1	3.2	4.2	2.5	6.3	1.4
CO2-equivalent GHG emissions	4,163	21,847	104,464	29,217	88,575	40,909	147,525	68,126	34,623	390,617	28,4

TABLE 57. GRAM-PER-MILE EMISSIONS BY VEHICLE/FUEL/FEEDSTOCK COMBINATION, AND STAGE OF THE FUELCYCLE

SPREADSHEET TABLE NOT AVAILABLE IN THIS VERSION

TABLE 58. SUMMARY OF PERCENTAGE CHANGES IN CO_2-EQUIVALENT EMISSIONS (BEST CEFs) (U. S. 2015)

LDVs, fossil or nuclear feedstocks (versus 26 mpg LDGV)	*fuelcycle only*	*fuel+materials*
Baseline: Gasoline (CG) made from crude oil (CO2-equivalent g/mi)	*527.6*	*600.0*
Gasoline (RFG-Ox10) made from crude oil	-1.5%	-1.3%
Gasoline (CFG) made from synthetic oil from coal	102.1%	89.8%
Diesel (0.001% S) made from crude oil	6.9%	4.7%
Methanol (M85) made from NG100/C0	-10.4%	-9.1%
Natural gas (CNG) made from NG100	-27.6%	-23.8%
Hydrogen (CH2) made from NG100/N0/H0/So0	-15.5%	-13.0%
LPG (P95/BU5) made from NGL57/LRG43	-25.7%	-22.7%
LDVs, biomass feedstocks (versus 26 mpg LDGV)		
Ethanol (E90 (corn)) made using C36/NG52/B0/EL8	10.9%	9.6%
Ethanol (E90 (W0/G100)) made using C0/NG0/B99/EL0	-37.4%	-32.9%
Methanol (M85 (wood)) made using C0/NG0/B96/EL3	-47.8%	-42.0%
Natural gas (CNG (wood)) made using C0/NG0/B98/EL2	-65.0%	-56.6%
Battery EVs (versus 26 mpg LDGV)		
All recharging from coal-fired power plants	-22.7%	-17.5%
All recharging from oil-fired power plants	-34.7%	-28.1%
All recharging from gas (boiler)-fired power plants	-62.9%	-52.8%
All recharging from nuclear power plants	-98.0%	-83.7%
All recharging from hydropower plants	-99.0%	-84.5%
National average generation mix: C64/F20/NG15/N1/B0/H0	-32.1%	-25.7%
Fuel-cell EVs (versus 26 mpg LDGV)		
Gasoline (RFG-Ox10) made from crude oil	-51.7%	-46.8%
Methanol (M100) made from NG100/C0	-52.9%	-47.7%
Methanol (M100) made from wood	-83.1%	-74.2%
Ethanol (E100 (W0/G100)) made using C0/NG0/B99/EL0	-70.7%	-63.4%
Hydrogen (CH2 (water)) made from N0/H100/So0	-89.7%	-79.7%
Hydrogen (CH2 (NG)) made from F1/NG97/B0/EL2	-60.2%	-53.8%

Blank (Blank) made from Blank	-60.2%	-54.5%

HDVs, fossil or nuclear feedstocks (versus 6 mpg HDDV)	*fuelcycle only*	*fuel+materials*
Baseline: Diesel (0.001% S) made from crude oil (CO2-equiv. g/mi)	*2,746.6*	*2,872.8*
F-T diesel (FTD100) made from NG	-0.3%	-0.3%
Diesel made from synthetic oil from coal	99.5%	95.1%
Gasoline (RFG-Ox10) made from crude oil	21.3%	21.0%
Methanol (M100) made from NG100/C0	-6.9%	-6.6%
Natural gas (CNG) made from NG100	-22.5%	-21.6%
Hydrogen (CH2) made from NG100/N0/H0/So0	-7.6%	-6.9%
LPG (P95/BU5) made from NGL57/LRG43	-21.0%	-20.3%
HDVs, biomass feedstocks (versus 6 mpg HDDV)		
Diesel mix (FTD0/SD0) made using oil, NG, soy	-0.0%	-0.1%
Biodiesel (SD100 (soy)) made using C0/NG80/B0/EL17	180.0%	172.5%
Ethanol (E100 (corn)) made using C36/NG52/B0/EL8	21.5%	20.5%
Ethanol (E100 (W0/G100)) made using C0/NG0/B99/EL0	-39.1%	-37.4%
Methanol (M100 (wood)) made using C0/NG0/B96/EL3	-60.3%	-57.6%
Natural gas (CNG (wood)) made using C0/NG0/B98/EL2	-66.6%	-63.8%
Fuel-cell HD EVs (versus 6 mpg HDDV)		
Gasoline (RFG-Ox10) made from crude oil	-36.2%	-35.1%
Methanol (M100) made from NG100/C0	-37.8%	-36.6%
Methanol (M100) made from wood	-77.6%	-74.7%
Ethanol (E100 (W0/G100)) made using C0/NG0/B99/EL0	-61.2%	-59.0%
Hydrogen (CH2 (water)) made from N0/H100/So0	-86.3%	-82.9%
Hydrogen (CH2 (NG)) made from F1/NG97/B0/EL2	-46.9%	-45.3%
Blank (Blank) made from Blank	-85.3%	-82.0%

Notes:

LDGV = light-duty gasoline vehicle
HDDV = heavy-duty diesel vehicle
CG = conventional gasoline
RFG = reformulated gasoline
Ox = oxygenate (ETBE, MTBE, ethanol, methanol) (volume % in active gasoline)
M = methanol (volume % in fuel for methanol vehicle; remainder is gasoline)
CNG = compressed natural gas
LNG = liquefied natural gas
CH2 = compressed hydrogen
LH2 = liquefied hydrogen
E = ethanol (volume % in fuel for ethanol vehicle; remainder is gasoline)
P = propane (volume % in LPG)
BU = butane (volume % in LPG)
FTD = Fischer-Tropsch dieses (volume % in fuel; remainder is soy diesel or conventional diesel)
SD = soydiesel (volume % in fuel; remainder is petroleum diesel)
NG = natural gas (% as feedstock [methanol, hydrogen, NGVs], or % of electricity generation [EVs],
EL = electricity, % of energy input to fuel production processes
C = coal (% as feedstock [methanol], or % of electricity generation [EVs], or % of energy input to fuel
F = fuel oil (% of electricity generation, % of energy input to fuel production process)
N = nuclear power (% of electricity generation [EVs, hydrogen vehicles])
B = biomass power (% of electricity generation [EVs], or % of energy input to fuel production process
So = solar power (% of electricity generation [EVs, hydrogen vehicles])
H = Hydro power (% of electricity generation [EVs, hydrogen vehicles])
NGL = natural gas liquids (volume % as source of LPG)
LRG = liquid refinery gases (volume % as source of LPG)
S = sulfur
W = wood (trees) (% as feedstock [ethanol])
G = perennial grasses (% as feedstock [ethanol])

Table 61. Results for space heating and water heating (Best CEFs) (U. S. 2015)

Heating source	*g-CO_2-equivalent/ MMBTU-heat-provided*	*% ch*
Natural gas	79,340	
LPG	89,178	
Fuel oil	107,371	
Electricity (average mix: C50/F1/NG30/N10/B2/H7)	201,305	
Electricity (100% natural gas)	147,913	
Electricity (100% fuel oil)	259,870	
Electricity (100% coal)	307,661	

"Best CEFs" are my best estimates of CO2-equivalency factors, as distinguished from the IPCC GWPs.

TABLE 63. UPSTREAM FUELCYCLE EMISSIONS AS A PERCENTAGE OFF END-USE EMISSIONS, BY POLLUTANT AND FEEDSTOCK/FUEL COMBINATION (BEST CEFS) (U. S. 2015)

Fuel ------>	**Coal**	**CG**	**RFG**	**Hwy diesel**	**F-T diesel**	**Fuel oil**	**Still gas**	**Coke**	**LPG**	**LPG**	**LPG**	**CNG**
Feedstock ------>	*coal*	*oil*	*oil*	*oil*	*NG*	*oil*	*oil*	*oil*	*oil*	*NGL57*	*NG*	*NG*
CO2	n.a.	27%	27%	19%	30%	n.a.	n.a.	n.a.	18%	12%	8%	20%
NMOCs	n.a.	30%	35%	19%	16%	n.a.	n.a.	n.a.	59%	39%	25%	52%
CH4	n.a.	2649%	2617%	4896%	5111%	n.a.	n.a.	n.a.	2246%	1692%	1266%	271%
CO	n.a.	5.4%	6.0%	8.4%	11.5%	n.a.	n.a.	n.a.	6.9%	4.9%	3.4%	4.8%
N2O	n.a.	4.9%	2.5%	27.9%	33.1%	n.a.	n.a.	n.a.	2.0%	1.2%	0.6%	1.9%
NO2	n.a.	53%	61%	10%	12%	n.a.	n.a.	n.a.	47%	35%	25%	42%
SO2	n.a.	287%	674%	870%	165%	n.a.	n.a.	n.a.	1057%	544%	149%	439%
PM (BC+OM)	n.a.	41%	41%	7%	7%	n.a.	n.a.	n.a.	118%	88%	65%	144%
HFC-134a (mg)	n.a.	n.e.	n.e.	n.e.	n.e.	n.a.	n.a.	n.a.	n.e.	n.e.	n.e.	n.e.
CO2-equivalent GHGs	n.a.	27%	28%	19%	30%	n.a.	n.a.	n.a.	20%	15%	11%	28%

Fuel ------>	**Nuclear**	**CH2**	**CH2**	**Methanol**	**Methanol**	**Methanol**	**Ethanol**	**Ethanol**	**SCG**	**Biodiesel**	**Grass**	**Wood**
Feedstock ------>	*uranium*	*water*	*NG*	*NG*	*coal*	*wood*	*corn*	*(W0/G100)*	*wood*	*soy*	*grass*	*wood*
CO2	n.a.	1612%	7762%	38%	128%	42%	143%	60%	36%	224%	n.a.	n.a.
NMOCs	n.a.	8%	65%	31%	192%	49%	296%	32%	91%	1107%	n.a.	n.a.
CH4	n.a.	2923%	8141%	4181%	5099%	1439%	1519%	642%	93%	25732%	n.a.	n.a.
CO	n.a.	2.9%	20.5%	6.3%	5.6%	20.3%	52%	20%	63%	507%	n.a.	n.a.
N2O	n.a.	n.a.	n.a.	4.1%	8.9%	15.9%	262%	118%	19%	10624%	n.a.	n.a.
NO2	n.a.	27%	85%	81%	72%	117%	550%	374%	92%	468%	n.a.	n.a.
SO2	n.a.	495%	764%	293%	791%	397%	966%	63%	668%	693%	n.a.	n.a.
PM (BC+OM)	n.a.	46%	191%	66%	163%	314%	742%	440%	548%	603%	n.a.	n.a.
HFC-134a (mg)	n.a.	n.e.	n.e.	n.e.	n.e.	n.e.	n.e.	n.e.	n.e.	n.e.	n.a.	n.a.
CO2-equivalent GHGs	n.a.	438%	2092%	38%	115%	53%	182%	84%	52%	455%	n.a.	n.a.

"Best CEFs" are my best estimates of CO2-equivalency factors, as distinguished from the IPCC GWPs.

TABLE 65. MATERIAL LIFECYCLE EMISSIONS COMPARED WITH VEHICLE END-USE EMISSIONS

SPREADSHEET TABLE NOT AVAILABLE IN THIS VERSION

FIGURE 1. THE SINGLE-SIDED LOGISTIC FUNCTION.

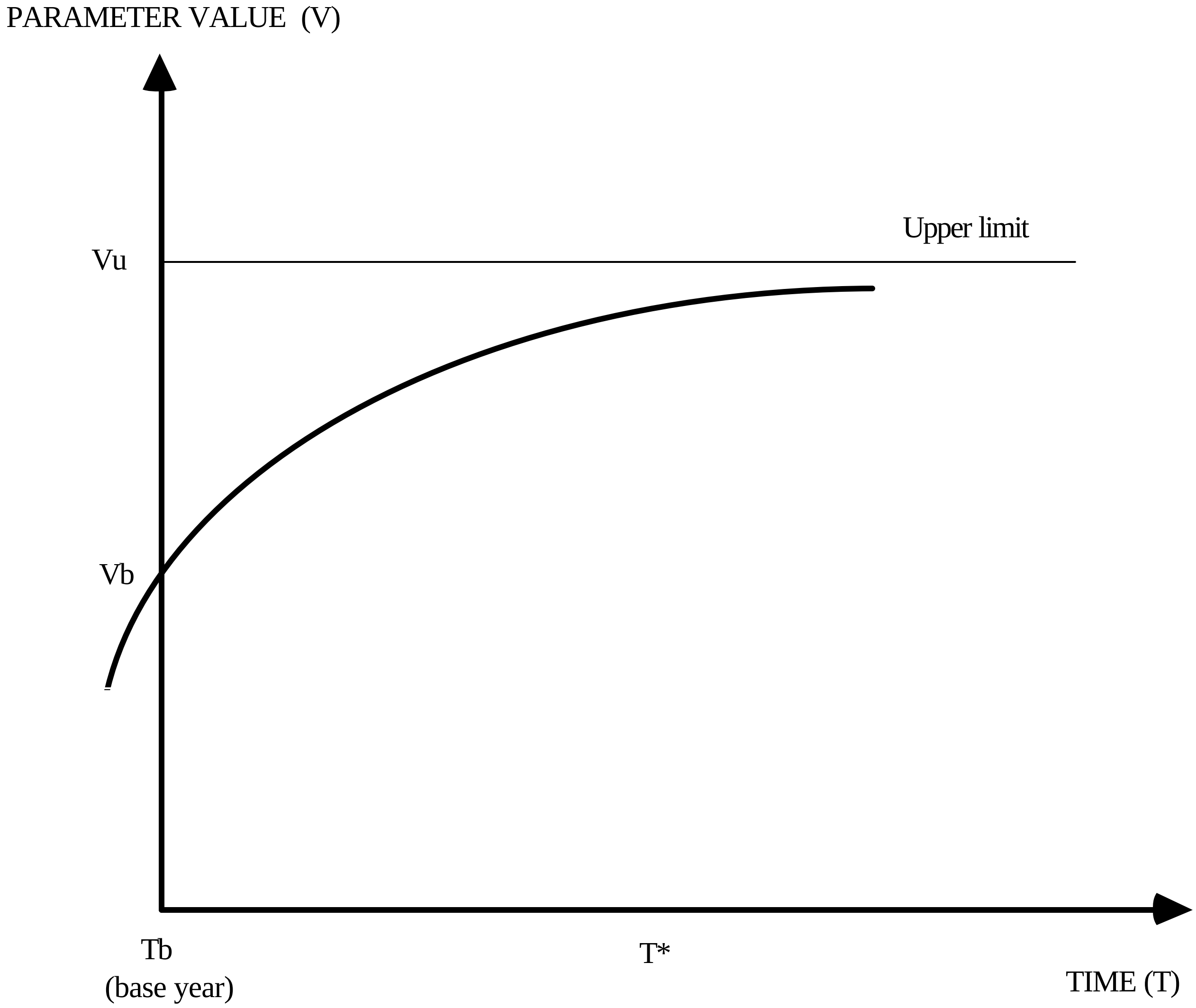


FIGURE 2. THE DOUBLE-SIDED LOGISTIC FUNCTION

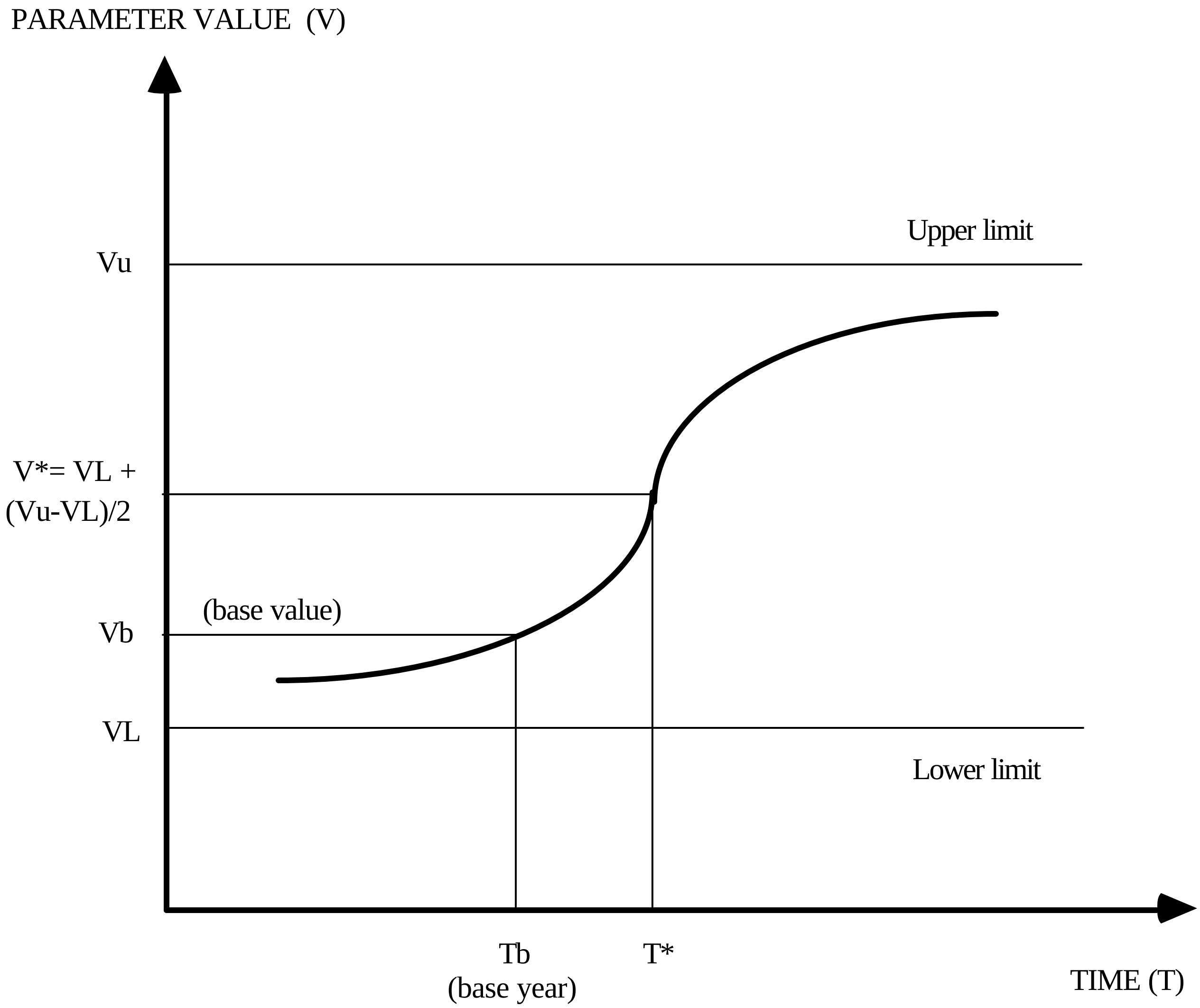


FIGURE 3. THE MARKET DISPLACEMENT EFFECT OF THE CO-PRODUCTS OF PRODUCTION PROCESSES

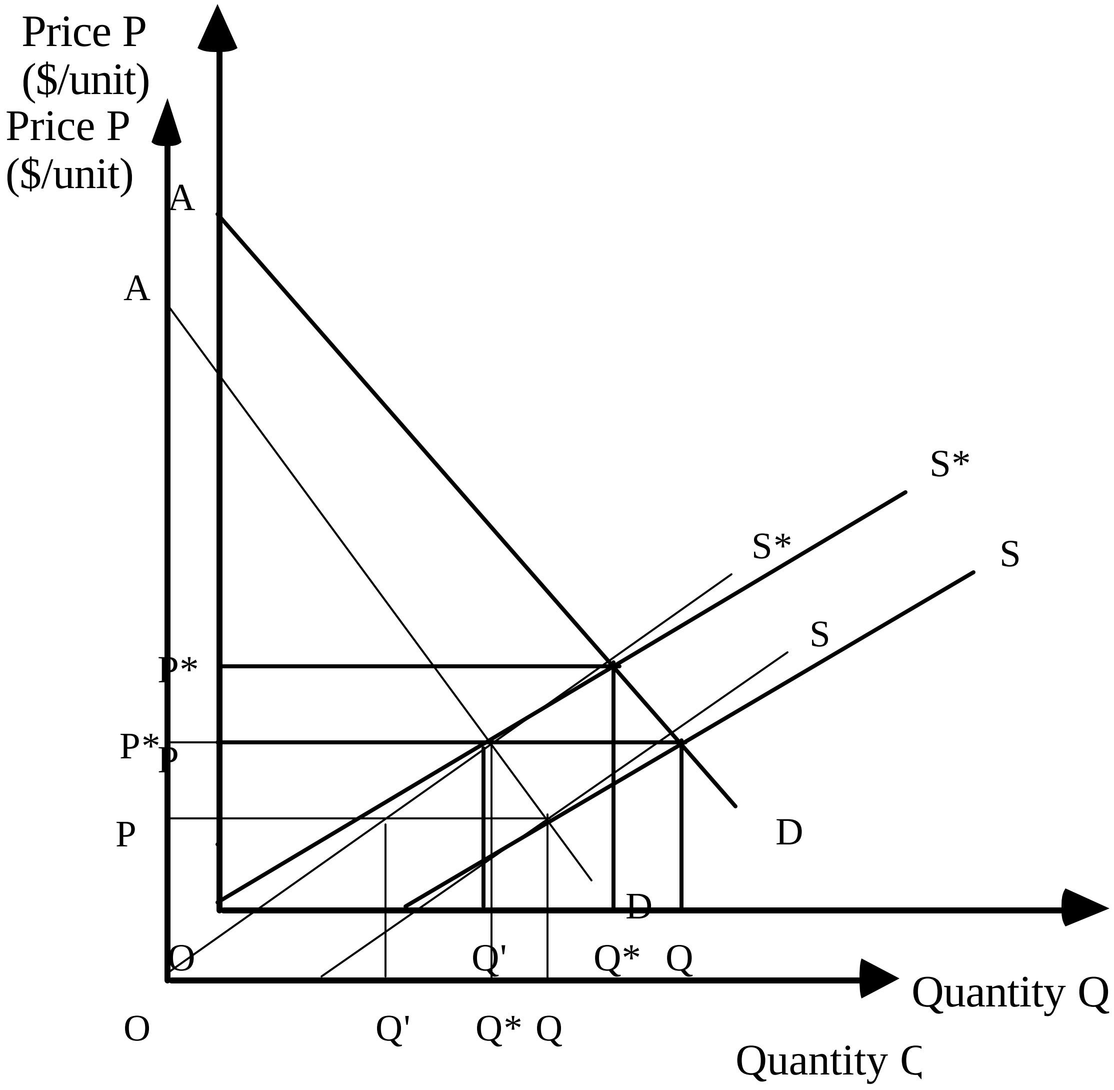


FIGURE 4. THE MARKET DISPLACEMENT EFFECT OF A SHIFT IN DEMAND FOR A PRODUCT

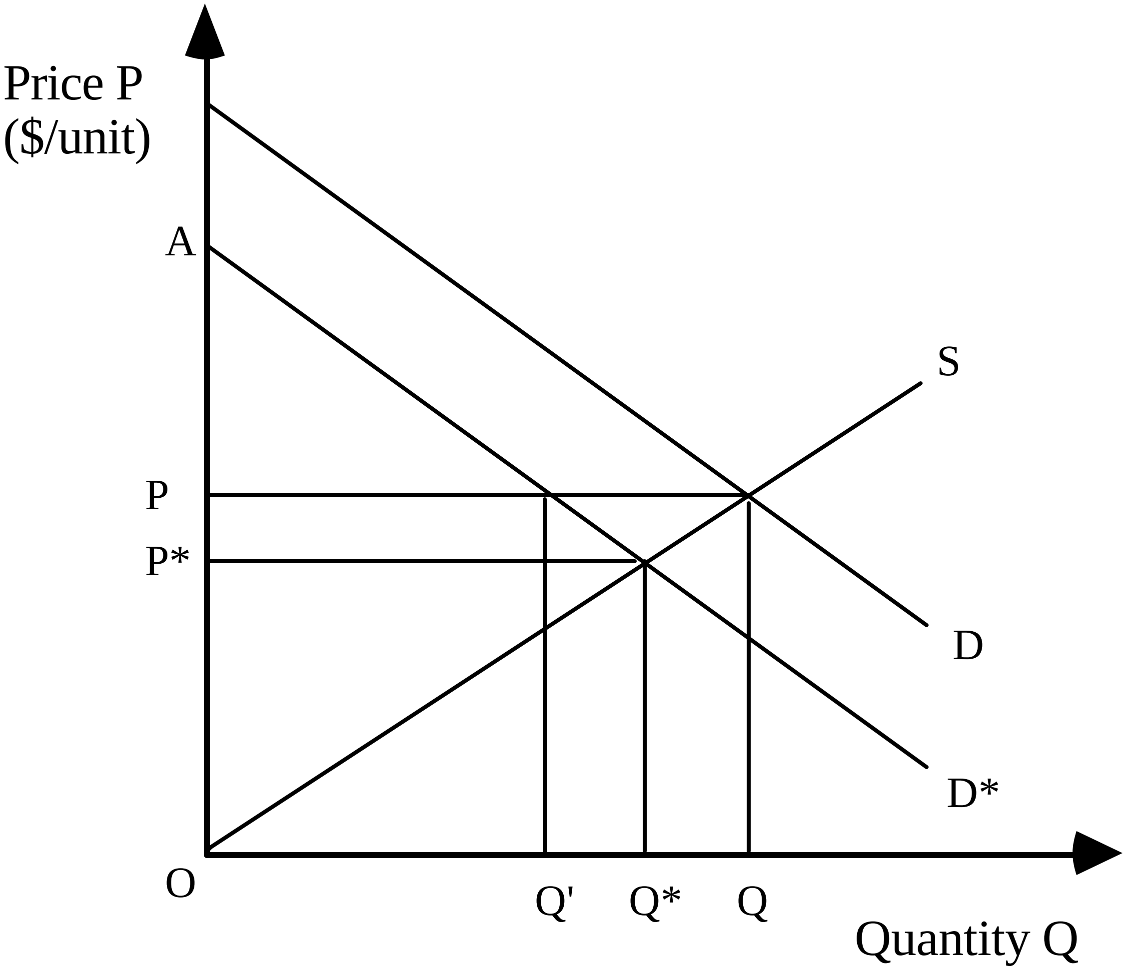


FIGURE 5. FATE OF SYNTHETIC N INPUT AN AGRICULTURAL SYSTEM (PARAMETER VALUES FOR CORN)

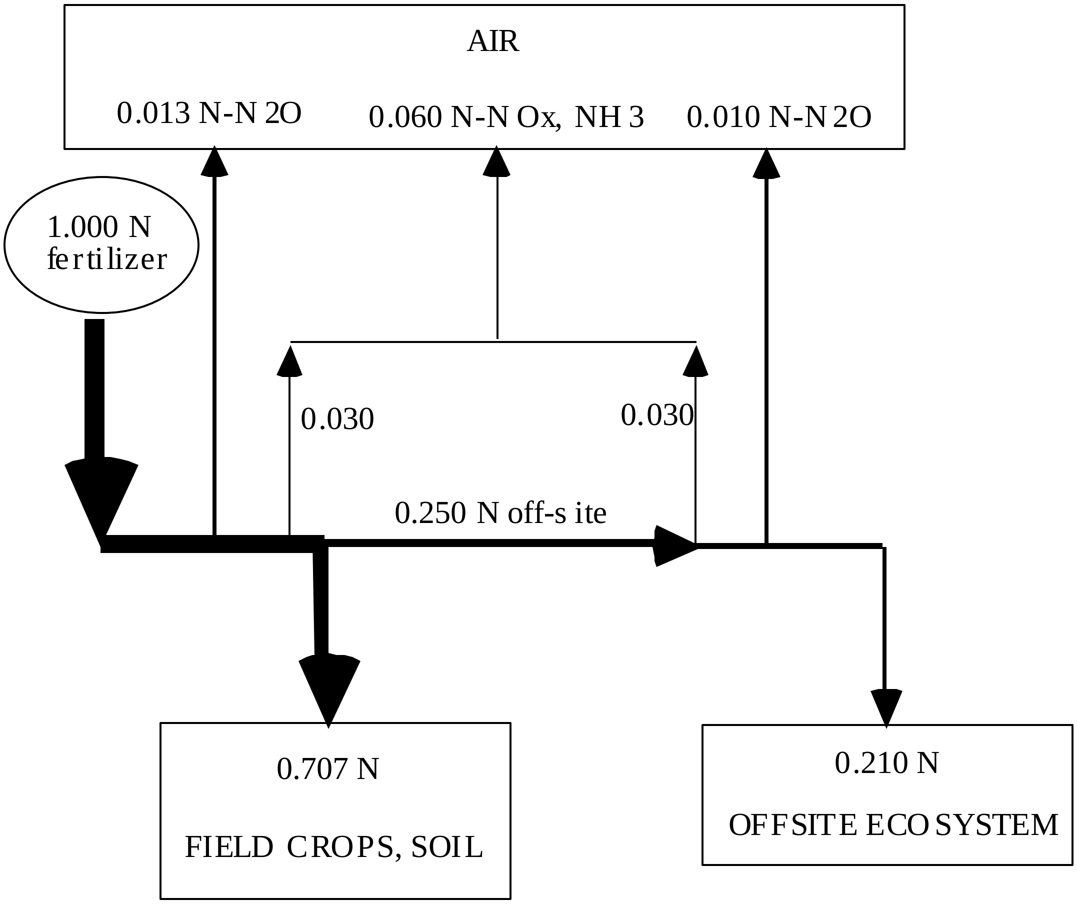
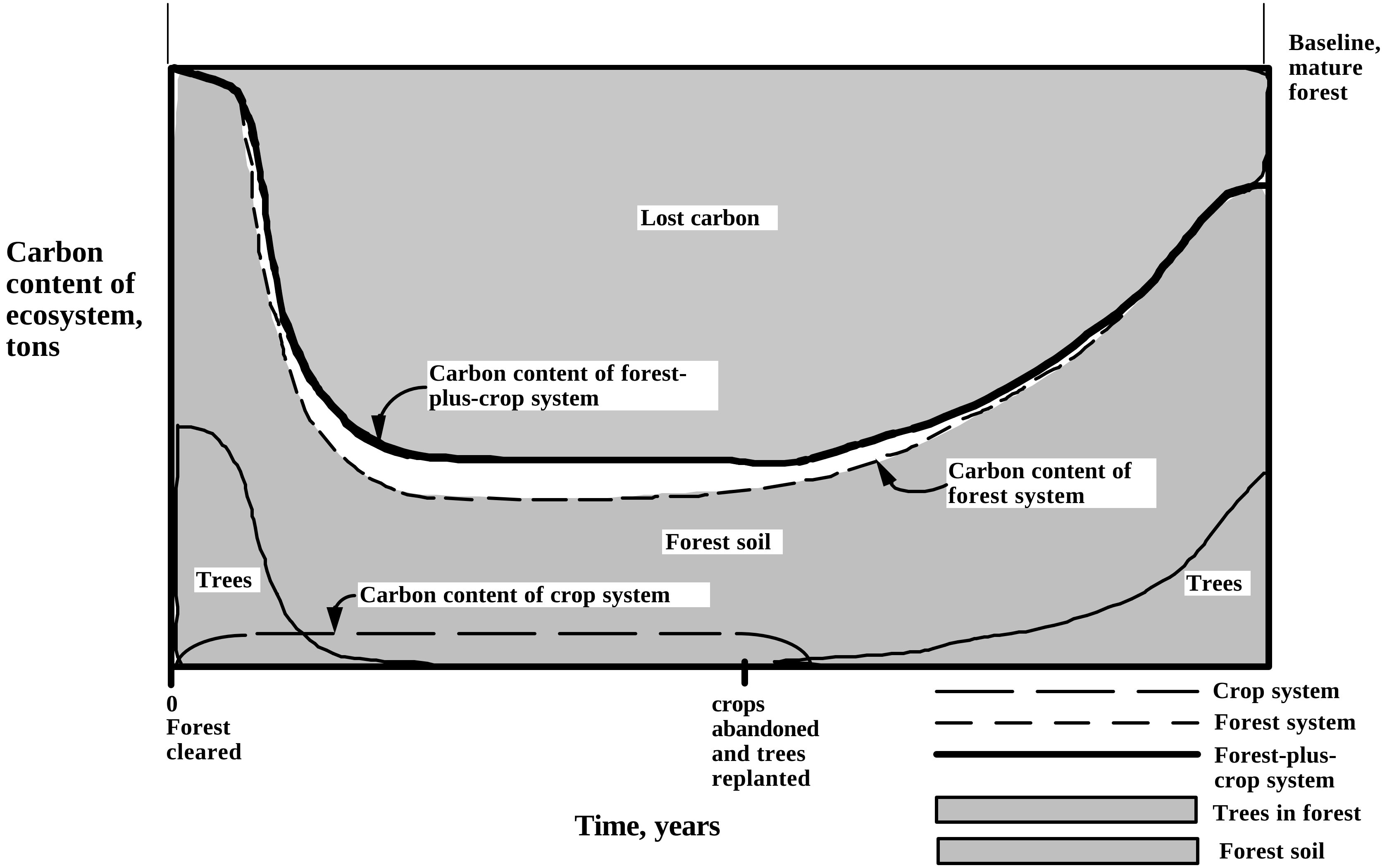


FIGURE 6. CHANGES OVER TIME IN THE CARBON CONTENT OF BIOMASS AND SOIL, DUE TO CHANGES IN LAND USE



Carbon history of replacing forest with crops