

TABLE Q.1 Estimated Emissions of CFCs from Mobile Air Conditioners

10 ⁹ g	Total Emissions		VMT by Cars with Air-Conditioning ^a (10 ⁹ mi)	Estimated Emissions ^b (g/mi)
	Region/Year	Reference		
46.0	USA/1986	Ford ^c	1,200	0.038
31.5	USA/1985	EIA ^d	1,200	0.026
4.278	California/1990	CARB ^e	170	0.025
-- Calculation based on emissions from a single car --				0.031 ^f

^a Rough estimates are based on total vehicle-miles-traveled (VMT) data from *Highway Statistics* and vehicle air conditioner use data in Hammerle et al. (1988). I assume greater use of air-conditioning in California than nationally.

^b Total grams from the first column, divided by total miles from the fourth column.

^c Hammerle et al. (1988).

^d EIA, *Potential Costs of Restricting Chlorofluorocarbon Use*, 1989 (includes manufacturing losses).

^e CARB, *Review and Evaluation of Programs for the Reduction of Chlorofluorocarbon Emissions from Motor Vehicle Air Conditioning Systems, Volume 1, Executive Summary*, 1990 (includes manufacturing losses).

^f According to Ford (Shiller, 1989), the amount of CFCs in a vehicle's air-conditioning system has declined from about 4 lb (prior to concern over CFCs and ozone) to a little under 3 lb in response to planned phase-outs of CFCs. I have assumed 2.6 lb. The CFC charge is changed out about twice over the life of an LDV, so that a total of three full charges are released (the last charge is released when the vehicle is scrapped). This amounts to 3,400 g over the average 108,000-mi life of an LDV, or 31.5 mg/mi. (Emissions from vehicle manufacturing and in-field system charging are small and can be ignored.)

estimate is consistent with estimates of aggregate CFC releases from the U.S. and California fleets (Table Q.1).

Vehicles without air-conditioning, of course, should be counted at 0 mg/mi. I assume that HDVs use the same amount of CFC-12 per mile as do LDVs.

Using a CFC-to-CO₂ conversion factor of 7,300 on a mass basis (Table 8), 31.5 mg/mi of CFC-12 results in 230 g/mi CO₂ equivalent. This is about 33% of the CO₂ equivalent of all other greenhouse gas emissions from the tailpipe of gasoline vehicles and more than 75% of the tailpipe CO₂-equivalent emissions from methanol and natural gas vehicles. Thus, replacing CFC-12 with a nongreenhouse gas would reduce the tailpipe and CFC greenhouse contribution of light-duty highway vehicles by a very large amount.

According to Ford (Hammerle et al., 1988), the automotive industry plans to eliminate CFC-12 as soon as possible. Ford states that existing CFC-12 facilities are being modified to produce a CFC-12 substitute, the fluorocarbon FC-134a (C₂H₂F₄). DuPont has a pilot plant producing developmental volumes of FC-134a for toxicity testing and market development (*Oil and Gas Journal*, September 5, 1989). This fluorocarbon has two environmental advantages over CFC-12: it does not contain chlorine and, hence, should not deplete stratospheric ozone; and, even though it has an infrared absorption band similar to that of CFC-12, it has a lifetime of only a few years (compared to over 100 years for CFC-12) and should not be as potent a greenhouse gas. Toxicology assessments of FC-134a should be completed by 1990, at which time replacement of CFC-12 could begin. Vehicle air-conditioning systems may have to be redesigned to accommodate FC-134a because it is somewhat less efficient at cooling than is CFC-12 (Hammerle et al., 1988). Ford also mentions the development of processes using water vapor rather than CFCs to produce foam for automotive applications.

Q.2 Ozone

Q.2.1 Background

Tropospheric ozone is a greenhouse gas, as well as a regulated oxidant pollutant. An important source of tropospheric ozone in the lower atmosphere in urban areas is the complex series of photochemical reactions involving reactive hydrocarbons (RHCs or NMHCs), nitrogen oxide (NO_x), and carbon monoxide (CO) emissions from highway vehicles and other fossil fuel combustion (Bolle et al., 1986).

The concentration of ozone near the earth's surface appears to be increasing, but there is no clear trend in the upper troposphere, where ozone is most effective as a greenhouse gas (Watson et al., 1990). In the mid- to high-latitudes of the Northern Hemisphere, an increase has been observed in the last two decades, particularly in the summer (Bolle et al., 1986), due at least in part to emissions of ozone precursors from airplanes, industry, and vehicles. Wang et al. (1985) and Campbell (1986) estimate that a doubling of the present tropospheric distribution of ozone would

raise the surface temperature by 0.9°C. Ramanathan et al. (1985) predict an 0.1°C increase as a result of a uniform 15% increase in tropospheric ozone from 0 to 9 km.

An absolute increase in ozone in the upper troposphere causes a much greater increase in the surface temperature than do ozone increases in other regions of the troposphere (Wang et al., 1985; Shine et al., 1990). Wang et al. (1985) imply that aircraft NO_x emissions are instrumental in elevating ozone levels. Similarly, Liu et al. (in Ramanathan et al., 1985) argue that most photochemical production of ozone occurs above the boundary layer, in the free troposphere, and that ground-level sources of NO_x are not important in ozone formation in the troposphere at an altitude of 2-8 km. If these claims are true, emissions of ozone precursors from vehicles would contribute little to greenhouse warming. On the other hand, Crutzen and Gidel (1983) claim that the upward transfer of photochemically active species from the polluted boundary layer to the free troposphere (via fast exchange episodes such as cyclones, thunderstorms, fronts, and intertropical convergence) was underestimated by the models and could contribute significantly to ozone formation and, ultimately, climate change.

The atmospheric chemistry and activity of ozone is complex (Crutzen and Gidel, 1983; Ramanathan et al., 1985; Wang et al., 1985; Bolle et al., 1986; Dickinson and Cicerone, 1986; Penner, 1990; Watson et al., 1990). Much remains to be learned about the production and destruction of ozone (all of the trace gases considered here, and others, are involved); the movement of ozone vertically and latitudinally within the troposphere, and between the stratosphere and the troposphere; the relationship between changes in the vertical and latitudinal distribution of ozone and the surface temperature (ozone, unlike the other trace gases, is not uniformly mixed in the troposphere); and the feedback effect on temperature of increasing temperature, changes in precipitation, and concentrations of other trace gases.

Q.2.2 Ozone from Vehicles

Ozone formation in the boundary layer due to vehicular emissions depends in part on the quantity and composition of hydrocarbon (HC) emissions (see Seinfeld, 1986 and 1989, for a discussion of the atmospheric chemistry). Enough is known about emissions of ozone precursors from petroleum-fueled vehicles and their atmospheric chemistry to support an estimate of the contribution of gasoline and diesel vehicles to ozone formation in urban areas in the lower troposphere. Less is known about the effect of methanol and electric vehicles (EVs), and next to nothing is known about the effect of natural gas, hydrogen, ethanol, and liquefied petroleum gas (LPG) vehicles.

Models of single- and multi-day ozone (smog) episodes have shown that methanol substitution for gasoline would reduce peak *urban* ozone levels (see Sperling and DeLuchi, 1991, for a review). The effect of methanol substitution on global climate would depend on the extent to which any reduction in ground-level urban ozone corresponded to a reduction in total global average ozone concentration in the middle and upper troposphere, and on the temperature effect of changes in the vertical distribution of ozone.

Emissions from optimized natural gas vehicles (NGVs) have not been completely characterized, and there is no model of ozone formation from NGV emissions. Qualitative analysis of what little is known suggests that NGVs might contribute slightly less to ozone formation in urban areas than methanol. Methane, the primary constituent of HC exhaust from NGVs, is essentially nonreactive and does not contribute to ozone formation in urban areas. The reactive HC emissions from NGVs appear to be, on the whole, less reactive than those from gasoline and methanol vehicles, although there are few data. This suggests that the use of natural gas would result in a moderate reduction in ozone formation in the boundary layer in urban areas, compared to gasoline, and a small reduction compared to methanol (see Sperling and DeLuchi, 1991, for a summary). But, as mentioned with regard to methanol, one must then determine the change in total (urban + rural + ocean) ozone, along with the change in the vertical distribution and global concentration of ozone resulting from changes in ozone in the boundary layer, and then estimate the associated temperature effects.

For at least two reasons, the use of EVs also could reduce ozone formation in the boundary layer in urban areas. First, power plants produce fewer nonmethane hydrocarbons (NMHCs) per mile of vehicle travel than gasoline and diesel vehicles, regardless of the fuel used at the power plant or the type of plant (Wang et al., 1990). Second, most of the emissions attributable to the use of EVs would occur at night, assuming that batteries were recharged overnight, and could be dispersed by nighttime air movement aloft before the irradiation necessary for ozone formation was most intense. Consistent with these expectations, models of ozone formation due to substitution of EVs for gasoline vehicles have shown very large decreases in ozone formation per EV or mile of travel (Hempel et al., 1989; Dowlatabadi et al., 1990).

However, while nighttime emissions may reduce smog formation in urban areas, they may not reduce ozone formation in the free troposphere because the boundary layer descends at night, and emissions from high stacks may be more likely to be carried to the free troposphere than are ground-level daytime emissions. And, since ozone is more effective as a greenhouse gas in the free troposphere than in the boundary layer, the use of EVs may not reduce the greenhouse effect of ozone as much as it reduces the concentration of ozone in urban areas.

A difficulty in applying the results of ozone-formation models to global warming is that the models calculate ozone concentration in urban areas, whereas in global warming we are interested in the global *average* ozone. Emissions of ozone precursors may leave an urban area and enhance ozone formation outside the area. A fuel that produces slowly reacting NMHC may reduce urban ozone, but may increase exurban ozone, because most of the photooxidation and ozone formation will occur after the NMHCs have left the urban area.

Q.2.3 Estimation of Ozone-Caused Warming from Vehicles

A formal modeling of the relationship between emissions of NMHC, NO_x , and CO and the concentration of ozone, and between ground-level ozone and global warming, is beyond the scope of this report and, in fact, would be quite complex. The IPCC (Shine et al., 1990) has provided very preliminary estimates of the global warming potential of ozone precursors, NMHC, NO_x , and CO, using relatively simple models of tropospheric chemistry. I use their values here. (I have

adjusted the NMHC factor to an NMHC-C factor; see Appendix O for further details.) Data on emission rates of NMHC, CO, and NO_x are from the U.S. Environmental Protection Agency's (EPA's) MOBILE4, EPA's AP-42, and studies of emissions from alternative-fuel vehicles relative to gasoline- and diesel-fuel vehicles (see Appendix B for more details).

Q.3 Water Vapor

The combustion of hydrogen-containing fuels -- gasoline, diesel fuel, methanol, and natural gas -- forms water vapor (H₂O). Water is much more abundant in the atmosphere than CO₂ and controls about 10 times more infrared radiation flux (Campbell, 1986; Dickinson and Cicerone, 1986; Mooney et al., 1987). However, the climatic effect of increased water vapor emissions is not easily determined because water vapor emissions can contribute to both global heating and cooling (Ramanathan, 1988), depending on the effect of the added vapor on global cloud cover, humidity, and precipitation. On one hand, water as vapor and clouds absorbs solar radiation and absorbs and emits thermal radiation (Dickinson and Cicerone, 1986). On the other hand, clouds reflect about 60% of the shortwave radiation they receive from the sun, causing net cooling at the surface. Very recent satellite research into the earth's radiation budget indicates that, on balance, reflection of shortwave radiation wins out, and clouds at present cause a net cooling of the earth (Ramanathan et al., 1989). However, realistic cloud effects have not yet been incorporated into greenhouse models (Kerr, 1989). Modeling the feedback effects of the warming on cloud formation will be especially difficult (Ramanathan et al., 1989).

In any event, my calculations indicate that water vapor emissions from current fossil fuel use (the direct contribution, not accounting for any feedback effects on evaporation) are negligible -- about 0.05% of the average amount of water in the atmosphere and 0.0013% of current global evaporation (DeLuchi, Sperling, and Johnston [1987]). I therefore ignore water vapor emissions from highway vehicles.

With solar-electrolytic hydrogen, there is no net emission of water because the hydrogen is made from water and then emitted as water. There will be changes in the distribution and form of water -- the hydrogen will be made from liquid water in one location and then emitted as water vapor in a different location -- but the total amount of water vapor emitted will be quite small compared to global evaporation (DeLuchi, et al., *A Comparative Analysis of Future Transportation Fuels*, 1987).

Appendix R:
Scenarios for Europe and Japan

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R.1 Introduction

The base-case input parameters are meant to represent the United States in 2000. To obtain emission results for countries other than the United States, important input variables in the model must be changed to represent the pertinent country or region. To model Japan and Europe, I changed the major input variables pertaining to (1) emissions from electricity generation, (2) emissions from petroleum refining, (3) tailpipe and evaporative emissions from gasoline and diesel-fuel vehicles, (4) tailpipe and evaporative emissions from alternative-fuel vehicles, (5) methane emissions from coal mining, and (6) fuel and feedstock transportation. These changes are discussed next. To model Canada, I assumed that all the base-case United States data apply, except the national average electricity mix.

R.2 Greenhouse Gas Emissions from Electricity Generation in Europe and Japan

To calculate greenhouse gas emissions from the generation of electricity used for a particular process (e.g., petroleum refining), one must know in what countries the process is done, the mix of fuels used to generate the electricity and the efficiency of electricity generation in those countries, and the emission factors for power plants in those countries. (This method follows the method used for the U. S. base case; see Appendix D.) In this section, these data are developed for the European and Japanese emission scenarios.

R.2.1 Geographic Distribution of Electricity-Consuming Processes Related to Fuel Use and Vehicle Use in Europe and Japan

Table R.1 shows the distribution of petroleum refining, uranium enrichment, and automobile manufacturing done for the countries of the European Economic Community (EEC) in 1988 and 1989. I assume that the distribution will be the same in 2000. To simplify the analysis, I assume that all the uranium nuclear fuel used in Japan is enriched in the United States, but that all the automobiles used in Japan are made in Japan and all the gasoline and diesel fuel used in Japan are produced in Japan.

TABLE R.1 Geographic Distribution of Electricity-Using Processes Pertaining to Use of Vehicles and Fuels in Europe, 1988-1989

Process	Fraction of Each Process that Is Done in the Following Countries						
	U.K.	Japan	Germany	France	Italy	U. S.	Nether- lands
Auto manufacture ^a	0.10	0.05	0.30	0.25	0.15	0.15	0.00
Petroleum refining ^b	0.20	0.00	0.20	0.20	0.20	0.20	0.20
Uranium enrichment ^c	0.00	0.00	0.00	0.85	0.00	0.15	0.00

^a The auto-manufacturing shares are calculated from data on world motor-vehicle production (Motor Vehicle Manufacturer's Association, 1990).

^b The petroleum-refining shares are based on the total petroleum-refining capacity of European countries (IEA, *Energy Statistics of OECD Countries, 1987-1988*, 1990).

^c The uranium-enrichment shares are calculated from data in EIA, *World Nuclear Fuel Cycle Requirements* (1990). (EURODIF, the main enrichment facility in Europe, is in France.)

R.2.2 Electricity Generation Shares by Fuel Type, Electricity Generation, and Distribution Efficiency

Table R.2 shows projections of generation shares by fuel type and of generation and distribution efficiencies for Japan and countries in Europe in 2000. Note that the estimates in Table R.2 are the power mix used for all purposes in each country (the national "average" mix), not the power mix likely to be used for any specific purpose within those countries. Therefore, they are comparable to the United States average or generic power mix in Table 6.

R.2.3 Emission Factors for Electricity Generation

Emission factors for electricity generation ($\text{g}/10^6$ Btu-fuel input, as per Table D.4) vary from country to country. (See IEA, *Electricity Controls in Electricity Generation and Industry*, 1988, for a summary of emission standards in Europe.) The IEA (1991) projected carbon monoxide (CO) and nitrogen oxides (NO_x) emission factors for electricity generation in Europe and the Pacific in 2005. I have used these projections (to represent 2000), along with the

TABLE R.2 Projected Electricity-Generation Shares by Fuel Type and Projected Electricity-Generation Efficiency in Europe and Japan for the Year 2000

Country	Generation by Type of Fuel ^a				Efficiency of Electricity Generation and Distribution ^b		
	Coal	NG	Oil	Nuclear	Coal	NG	Oil
Canada	0.18	0.03	0.04	0.16	0.34	0.42	0.39
France	0.08	0.01	0.02	0.70	0.33	0.53	0.30
Germany	0.50	0.06	0.04	0.33	0.33	0.33	0.41
Japan	0.19	0.17	0.14	0.38	0.41	0.41	0.42
Sweden	0.03	0.01	0.02	0.43	0.19	0.20	0.24
United Kingdom	0.70	0.01	0.09	0.18	0.32	0.26	0.32
EEC-wide ^c	0.41	0.15	0.07	0.16	0.33	0.36	0.34

^a The generation shares were calculated from each country's projection of its mix of fuels input to electricity generation in the year 2000 (International Energy Agency, *Energy Policies and Programmes of IEA Countries*, 1989), except in the case of France. The value for France is its actual 1987 mix as shown in IEA, *Energy Balances of OECD Countries, 1986-1987* (1989).

^b I calculated the efficiency of electricity generation for each fuel in each country in 1987 on the basis of data in the IEA report, *Energy Balances of OECD Countries, 1986-1987* (1989) (converted from lower to higher heating values), then assumed that electricity generation and distribution will be 5% more efficient (in relative terms) in the year 2000 than it was in 1987.

^c The estimate of EEC-wide generation and distribution efficiency is my own. EEC = European Economic Community.

base-case U. S. emission factors for methane (CH₄), nitrous oxide (N₂O), and nonmethane organic compounds (NMOC) (Table R.3). Note that the IEA's NO_x emission factors are lower than my projections for the United States (Table D.4).

R.3 Emissions from Petroleum Refineries in Europe and Japan

Emissions from refineries in Europe and Japan are a function of the mix of fuels used to generate process heat, the amount of fuel used to produce a particular product, and the emission factors for refinery boilers.

TABLE R.3 Projected Emission Factors for Electricity Generation in Europe and the Pacific, by Fuel Type, in Grams of Pollutant per 10⁶ Btu of Fuel Input^a

Pollutant	Europe				Pacific (Japan)			
	NG	NG	Oil	Coal	NG	NG	Oil	Coal
CO	24	24	6	50	24	24	6	50
NO _x	97	97	96	174	24	24	56	190
CH ₄ , N ₂ O, NMOCs	-- Same as Table D.4 --				-- Same as Table D.4 --			

^a Emission factors are given on a higher-heating-value basis. The CO and NO_x emission factors are IEA (1991) projections for the year 2005. The CH₄, N₂O, and NMOC emission factors are assumed to be the same as projected here for the United States in the year 2000 (Table D.4).

^b I assume that 75% of the natural gas used as a power plant fuel is burned in boilers in the year 2000.

I assume that in 2000, refineries in Europe and Japan will produce low-sulfur diesel fuel, and standard, unleaded, nonreformulated gasoline. I assume standard gasoline but low-sulfur diesel fuel because, according to a comprehensive review of fuel quality and emissions regulations in Europe (McAarragher et al., 1990), there are, at the moment, no hard plans to introduce reformulated gasoline in Europe. However, it appears that low-sulfur diesel fuel will be required by the mid to late 1990s.

R.3.1 Fuel Use by Refineries

I project the mix of fuels used by refineries in Europe and the Pacific (the Pacific is used here to represent Japan) by using 1988 data from the International Energy Agency (IEA, *Energy Statistics of OECD Countries, 1987-1988, 1990*). The 1988 data are shown in Table R.4. In Table R.4, I have included the IEA's estimates of fuel used by refineries in the United States for comparison with the EIA data used in this report (Table H.4). The IEA data for the United States (Table R.4) are reasonably consistent with the EIA data used here.

I use the "Energy %" results in Table R.4 for 1988 to represent the mix of fuels used by refineries in Europe and the Pacific in 2000. To calculate the energy intensity of refineries in Europe, in Btu of process fuel per Btu of product, additional analysis is required. European refineries probably consume less energy per Btu of gasoline produced than do refineries in the United States because they produce less gasoline per barrel of crude (and the average process

TABLE R.4 Fuel Use by Refineries in Europe and the Pacific, 1988^a

Process Fuel Used at Refinery ^b	EEC ^c		United States		Pacific Region	
	Energy Use ^d	% of Energy ^e	Energy Use ^d	% of Energy ^e	Energy Use ^d	% of Energy ^e
NG + coke gas (10 ⁹ cal)	11,424	3.41	157,365	21.09	2,445	1.83
Refinery gas (10 ³ t)	12,385	47.18	32,951	56.32	7,575	72.29
LPG (10 ³ t)	193	0.68	962	1.51	8	0.07
Coke (10 ³ t)	3,330	8.32	14,478	16.23	419	2.62
Oil (10 ³ t)	11,143	35.11	583	0.82	2,384	18.82
Electricity (gWh)	20,676	5.31	34,966	4.03	6,788	4.37
Total quads used	1.33	100.00	2.96	100.00	0.53	100.00
Total crude + NGL refinery input (quads) ^f	22.94		31.12		7.98	
Ratio of fuel use to total input (%) ^g		5.79		9.52		6.64

^a The data on the amount of each process fuel consumed are from the IEA report, *Energy Statistics of OECD Countries, 1987-1988* (1990). I converted these data from the units shown to quads by using the heating values of Tables C.1 and C.3.

^b t = metric ton (1,000 kilograms) and cal = calorie.

^c EEC = European Economic Community.

^d The amount of fuel used in 1988, in the units indicated.

^e For each fuel, the energy % is equal to the amount of Btu of that fuel used by refineries divided by the total amount of Btu of fuel used by refineries, multiplied by 100.

^f Calculated from the following data on refinery input:

	EEC	United States	Pacific
Crude input (10 ³ t)	524,563	701,269	183,833
NGL input (10 ³ t)	5,075	16,230	564

From IEA, *Energy Statistics of OECD Countries, 1987-1988* (1990).

^g Total quads of fuel used by refineries for process energy, divided by total quads of crude oil and natural gas liquids input to refineries, multiplied by 100.

energy per Btu of gasoline increases as the gasoline yield increases) and, in particular, less unleaded gasoline.

Note that the data of Table R.4, which show that European refineries consume much less energy per barrel of crude input than U. S. refineries, do not by themselves show that European refineries consume less energy per barrel of any individual product because, in Europe, a much larger fraction of the crude input is converted into products, such as diesel and residual fuel, that require relatively little energy to make (EIA, *Energy Statistics of OECD Countries, 1987-1988, 1990*). The low overall energy consumption of European refineries is thus explained, at least in part, by the output mix. However, my own calculations indicate that European refineries also consume less energy per barrel of product, at least for now.

Forthcoming tailpipe-emissions regulations in Europe will require the use of a catalytic converter, which in turn will require the use of unleaded gasoline. This will increase energy requirements. On the other hand, Europeans will likely continue to use much more diesel fuel than do Americans, and this situation will tend to keep energy requirements per Btu of gasoline lower in Europe. I assume 0.12 Btu of process fuel per Btu of gasoline in Europe in 2000. For low-sulfur diesel fuel, I assume the same value as in the United States.

R.3.2 Emission Factors for Refinery Boilers

To calculate emissions from refinery boilers in Japan and Europe, I use the base-case emission factors for refinery boilers in the United States for all pollutants except NO_x (Table A.2). For NO_x , I use projections made by the IEA (1991) for 2005 to represent the year 2000 (Table R.5).

R.4 Emissions from Gasoline Light-Duty Vehicles and Diesel-Fuel Heavy-Duty Vehicles in Europe and Japan

Presently, exhaust emission standards are much more lax in Europe than in North America. However, the European community is finalizing relatively stringent new motor vehicle emissions standards (McAllagher et al., 1990). Considering these standards, the IEA (1991) has used the U. S. Environmental Protection Agency (EPA) MOBILE4 emissions model to project CO , CH_4 , and NO_x emission factors for motor vehicles in Europe in 2005. These projections are shown in Table R.6. Since the IEA (1991) did not project NMOC emissions, I estimated tailpipe and evaporative NMOC emissions in Europe using the EPA's MOBILE4 emission model. I specified the input parameters to MOBILE4 so that it approximately reproduced the IEA's projected emission factors for CO , NO_x , and CH_4 (Table R.6), then used the corresponding MOBILE4 NMOC emission factor. I assume that vehicles in Europe will use three-way catalytic converters, so they will emit about the same amount of N_2O as that projected here for vehicles in the United States in 2000.

TABLE R.5 Projected NO_x Emission Factors for Refinery Boilers in Europe and the Pacific for the Year 2005, in Grams per 10⁶ Btu of Fuel Input^a

Boiler	Europe	Pacific
NG boiler	37.8	13.0
Oil boiler	96.0	0.0

^a Calculated by multiplying the base-case NO_x emission factor for industrial boilers (Table A.2) by the ratio of the IEA (1991) projection of the NO_x emission factor for European and Japanese refinery boilers in 2005 to its projection of the factor for U. S. refinery boilers in 2005. I use the IEA's "Pacific" factors to represent Japan.

TABLE R.6 Projected Emission Factors for Light-Duty Gasoline Vehicles and Heavy-Duty Diesel Vehicles in Europe, in Grams per Mile^a

Pollutant	LDGV	HDDV
CH ₄	0.14	0.16
CO	14.5	8.05
NO _x	1.29	27.40
Tailpipe NMOC	1.00	2.50
Evaporative NMOC	1.65	0.00
N ₂ O	0.06	0.06

^a CH₄, CO, and NO_x emission factors are the IEA (1991) projections for Europe in 2005. Tailpipe and evaporative NMOC emission factors are from my run of MOBILE4, set to reproduce the IEA's projected European emission factors. The N₂O emission factors are from Table B.2.

I assume that Japan will adopt the new emissions standards mandated by the U.S. Clean Air Act (CAA), and that as a consequence, vehicles in Japan will have the same emission factors as projected here for vehicles in the United States in 2000 (Table B.2).

The IEA's (1991) projections for the United States do not account for the new CAA legislation affecting motor vehicles. Thus, if any country in Europe adopts the new U. S. standards, vehicles in those countries will have emission factors lower than those shown here. I did not account for this possibility.

Evaporative emissions from gasoline vehicles are a function of the volatility of gasoline and the extent and effectiveness of emission controls. There are no strict limits on gasoline volatility in Europe or Japan. However, the European community is planning to control evaporative emissions from both vehicles and Stage I fuel restocking at service stations. Some European countries are considering controlling emissions from refueling vehicles. Therefore, to estimate evaporative emissions in Europe in 2000, I reran the EPA's MOBILE4 model and my own model of upstream evaporative emissions, assuming that gasoline has an RVP of 11.5, that emissions from refueling

vehicles are controlled beginning in 1998, and that Stage I refueling is controlled. Because vehicular and upstream evaporative emissions of gasoline are a relatively minor source of greenhouse gases, I have not shown the new European results here.

R.5 Emissions from Alternative-Fuel Vehicles in Europe and Japan

I establish two cases for alternative-fuel LDVs: a stoichiometric case and a lean-burn case. In the stoichiometric case, I use the base-case relative thermal efficiency values in Table 2. I assume that all alternative-fuel LDVs in Europe and Japan emit the same amount of NO_x as gasoline LDVs in Europe and Japan, and that alcohol-fuel LDVs emit the same amount of CO as the gasoline LDVs, but that gaseous-fuel LDVs emit less CO than gasoline LDVs. I scale NMOC and CH_4 emissions so that the ratio of emissions from alternative-fuel LDVs to emissions from gasoline LDVs is the same as it is in the U.S. base case.

Because the new EEC standards will not be nearly as strict as the new U.S. CAA standards, lean-burn alternative-fuel LDVs might be able to meet the new EEC NO_x standard. In the lean-burn case for alternative-fuel vehicles in the EEC, I assume that their relative thermal efficiency improves by 10 absolute percentage points over the values in Table 2 (e.g., methanol goes from +15% to +25%), and that CO emissions are less than in the stoichiometric case — significantly less in the liquefied petroleum gas and compressed natural gas cases. I assume that NMOC and CH_4 emissions are 25% lower than in the EEC stoichiometric case for alternative-fuel vehicles.

I assume that alternative-fuel HDVs will have lower NO_x emissions than EEC diesel HDVs because the IEA emission projections for diesel HDVs are quite high, and alternative-fuel HDVs can produce lower NO_x emissions when the NO_x standard is not constraining all vehicles to the same emission level. I assume that alternative-fuel HDVs will emit more CO and NMOCs than will diesel HDVs. I calculated evaporative emissions on the basis of the relative volatility and the molecular vapor weight of the alternative fuel (Appendix B), and I estimated tailpipe NMOC emissions relative to NMOC from gasoline vehicles.

For all cases and all countries, I use the N_2O emission factors in Table B.2 because N_2O is formed on a three-way catalyst.

For all countries, I assume the base-case U. S. electric vehicle characteristics.

R.6 Methane Emissions from Coal Mining in Europe and Japan

Most of the coal used in Japan is imported from Australia (EIA, *International Energy Annual*, 1991). The methane emission factor for coal produced in Australia is about 300 ft³/ton (ICF, *Methane Emissions*, 1990). The average for coal produced in and imported to Europe is about 450 ft³/ton-produced (ICF, *Methane Emissions*, 1990). I use these values here for the year 2000 analysis.

R.7 Feedstock Production and Feedstock and Fuel Transport Patterns in Europe and Japan

To model feedstock and fuel transportation patterns in Europe and Japan, I used data on international production and movement of coal, oil, and gas as reported by the EIA (*International Energy Annual*, 1991).

R.7.1 Coal

I assume that Europe produces most of the coal it consumes but that it imports some coal from the United States by dry bulk carriers. I assume that rail transport distances for coal are shorter in Europe than in the United States. For coal use in Japan, I assume that most coal is imported from Australia and that rail transport distances within Japan are very short.

R.7.2 Crude Oil and Petroleum Products

I assume that Europe imports a greater fraction of the total crude oil it uses than does the United States. I assume that pipeline transport distances for both crude oil and petroleum products are shorter in Europe than they are in the United States. I also assume that truck transport is not as long in Europe as in the United States, because Europe is more densely settled than the United States. In Japan, I assume that most oil is imported from the Middle East and that pipeline transport distances within Japan are very short.

R.7.3 Natural Gas

For natural gas (NG) use in Europe, I use the base-case U. S. values for production and transport. Natural gas fields in Western Europe are closer to major consumers than are NG fields in the United States, but Europe also imports a lot of NG from the U.S.S.R., and this NG must be piped a long distance to end users in Europe. In Japan, I assume that all gas is imported as liquefied natural gas from the South Pacific.

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Appendix S:
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