TABLE M.8 Emissions of Methane from Coal Mining

Reference	Methane (ft^3) per Ton of Coal	Type of Estimate
Emissions from all mining[a]		
Ehhalt and Schmidt (1978)	87- 306[b]	Global average
Okken and Kram (1989)	247-309[c]	Global average
Okken and Kram (1989)	211	U. S. average
ICF Incorporated (1990)	400	U. S. average[d]
ICF Incorporated (1990)	492	World average[d]
Emissions from underground mines[e]		
Deul and Kim (USBOM) (1988)	25-3,100	Range from over 100 U. S. mines
Deul and Kim (USBOM) (1988)	833	1975 Average; 60 U. S. mines
Curl (IEA) (1978)	160-1,300	From a German study
Methane content of coal[f]		
Deul and Kim (USBOM) (1988)	416-705	Anthracite
Deul and Kim (USBOM) (1988)	192-544	Low and medium-volatile bituminous
Deul and Kim (USBOM) (1988)	64-288	High-volatile bituminous
Deul and Kim (USBOM) (1988)	64-224	Shallow, high-volatile bituminous
Deul and Kim (USBOM) (1988)	200	Estimate of average U. S. gas content
This report	180[g]	Estimate of average U. S. gas content

[a] Total methane emitted as result of coal mining in the world or in the United States divided by total coal production in the world or in the United States.

[b] I divided their estimate of global methane emissions from coal by 1975 worldwide coal production, including lignite and brown coal, assuming a methane density of 20 g/SCF. Ehhalt and Schmidt (1978) cite a 1972 report, which in turn cites a 1963 paper (see text).

[c] Based on 1979 coal production.

[d] Based on ICF's estimate of 368 BCF of methane emitted in the United States and 2,494 BCF of methane emitted globally, and EIA data on United States and world coal production.

[e] Cubic feet of methane emitted from an underground mine per ton of coal produced from that underground mine. The U. S. Bureau of Mines (USBOM) emission rates (Deul and Kim, 1988) are measured at the exhaust fan of the mine. Note that this is not the same sort of ratio as in footnote a. Here the ratio applies only to a specific mine or set of mines, whereas in footnote a, global or regional averages are used. The estimates of emissions to the atmosphere include (or ought to include) emissions from surface as well as underground mines. A global average could be calculated from per-mine values if values were obtained for every kind of mine, and the values were then weighted by global production from that kind of mine.

TABLE M.8 (Cont.)

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[f] Cubic feet (ft^3) of methane contained in a sample ton of coal. This is further removed from a global emission estimate. Total methane emissions are greater than the amount of methane contained in a piece of mined coal; emissions also come from the face of the wall and the cracks in the wall from which the coal was recovered.

[g] I have multiplied the typical methane content of each class of coal by the share of that class of 1987 total coal production in the United States and summed over all classes. The data I used are as follows:

Class of Coal	% Fixed Carbon, Moisture and Ash-Free	Methane (ft^3) per ton	Share of 1987 U. S. Production (%)
Anthracite	94.1	560	0.3
Low-volatile bituminous	80.7	450	2.7
Medium-volatile bituminous	73.5	450	5.8
High-volatile A bituminous	60.5	190	35.7
High-volatile B & C bituminous and subbituminous	~55	130	46.8
Lignite	51.6	120	8.6

The fixed carbon percentages are from EIA, *Coal Data: A Reference* (1989), and are representative of each coal class. The production shares are from EIA, *Coal Production - 1987* (1988). The ft^3-methane/ton methane contents are from Deul and Kim (1988). I have picked typical values (not averages) from Deul and Kim from their ranges (for the direct determination method) for anthracite, low-volatile bituminous, medium-volatile bituminous, and high-volatile bituminous coals. I have assumed that the methane content of subbituminous coal is equal to that of high-volatile bituminous coal because the composition of the two kinds of coal are very similar. I have assumed that lignite contains less methane than does high-volatile bituminous coal because lignite has somewhat less fixed carbon (and methane content is related to fixed carbon).

Note: USBOM = U. S. Bureau of Mines

Thus, one would expect the national-average emission into the air, per ton of coal mined, to be considerably greater than the national-average methane content of coal because of the abandonment of mines with depleted coal. The data of Table M.8 confirm this. Compare, for example, my estimate of the average methane content of coal, 180 ft^3/ton, with a recent, comprehensive estimate of methane emissions from coal mines, 380 ft^3/ton (ICF, 1990).

The draft ICF (1990) report estimates emissions from coal mining and use for every state in the United States as a function of the amount of coal mined, the mining technique, and the methane content of the coal. The approach to estimating the methane content of the coal is similar to the one used here but more disaggregated, and the study takes the additional step of relating methane content to emissions.

M.5.3 Fate of Methane Emissions from Coal Mines

Most, but not quite all, of the methane released from coal enters the atmosphere as methane. Methane may be mixed with air and vented; collected, drained, and flared; or collected, drained, and used as a fuel. Most methane simply is mixed with air and ventilated to the surface. The vented methane is too dilute to be used as a fuel: by law, the vented gas must contain less than 1% methane (Kim, 1990), which is too little to burn. Okken and Kram (1989) state that about 10% of the emitted methane is captured and used as fuel, but this seems high: underground bituminous mines reported consuming 500 million ft^3 of natural, manufactured, and mixed gas in 1987 (U. S. Department of Commerce, U. S. Bureau of the Census, *1987 Census of Mineral Industries, Coal Mining,* 1989), which is much less than 1% of the roughly 300 BCF that, according to Table M.8, would have been emitted by underground bituminous mines. (Of course, it is possible that mines do not report use of coal-bed gas.) Deul and Kim's (1988) discussion of methane recovery indicates that presently only a miniscule fraction of coal-bed gas is put to work. The ICF (1990) report states that only in Alabama do some mining operations use captured methane. Thus, it is safe to assume that virtually all methane is vented to the atmosphere.

The recovery of methane from coal beds in advance of mining, for use as a fuel, will probably reduce methane emissions from mining, but not by as much as might be imagined. This is because the beds from which gas will be recovered are either unmineable (because the coal is too deep, as in the San Juan Basin of Colorado) or so gassy that they would not have been mined had the gas not been removed. In the latter case, the recovery of methane in advance of mining may reduce the methane content to a typical level and, hence, will not really lower the overall emission rate.

Based on this, I assume that U. S. average emissions from mines are 380 ft^3/ton of coal mined. I then assume that in the year 2000, 5% of the coal-bed emissions will be sold as fuel and 5% will be flared. The estimate of methane emissions per ton of coal produced is converted to emissions per ton of coal available to end users by multiplying by the ratio of production to available consumption (Appendix F).

M.5.4 Composition of Coal-Bed Methane and Its Relevance to Global Warming

Coal-bed gas contains small amounts of ethane and CO_2, in addition to methane. These greenhouse gases have different warming potentials per gram; consequently, the precise overall warming effect of vented coal-bed gas depends on the proportions of these individual compounds.

Deul and Kim (1988) show the composition of coal gas from seven coal beds. Coal-bed gas generally contains a bit more methane and much less of the higher hydrocarbons than does natural gas. It contains nitrogen and CO_2, but no sulfur compounds. My assumptions are shown in Table C.3.

In the calculation of CO_2-equivalent greenhouse gas emissions from vented coal-bed gas, separate warming factors are assigned to each species (warming factors are derived in

Appendix O). I assume that all carbon in the small amount of flared coal-bed gas oxidizes completely to CO_2.

M.6 Methane Emissions from Other Combustion Sources (Trains, Ships, Pipeline Compressors, Electricity Plants, Etc.)

For trains, ships, pipeline compressors, tractors, engines, electricity-generating plants, wood boilers, and several other sources, I have used AP-42 (Volumes I and II) methane emission factors (Table A.2; methane emissions from power plants are shown in Table D.4). For cases in which AP-42 provided estimates of THC emissions but not methane emissions, I multiplied the THC emission factor by an estimate of the methane proportion of THC emissions, based on known proportions for the same fuel in similar applications.

The South Coast Air Quality Management District (1990) and the Texas Air Control Board (1990) report total annual methane emissions from specific petroleum refineries in California and Texas. These emission values, divided by the total output of the refineries (in 10^6 Btu) (EIA, *Petroleum Supply Annual*, various years), results in 0.24-2.4 g-methane/10^6 Btu-output, with most values falling toward the lower end of the range. This total emission factor is consistent with the methane contribution from fuel combustion alone, which is about 0.1 g/10^6 Btu output. (One would expect that fuel combustion would produce only a small part of the total refinery methane emissions.) Thus, methane emissions from refinery process areas, excluding boilers, typically are around 0.4 g/10^6 Btu.

There are few data on methane emissions from facilities producing only methanol. Data on methane emissions from plants producing methanol and other products (Texas Air Control Board, 1990), combined with data on the production capacity of methanol facilities (U. S. Department of Commerce, Office of Chemicals and Allied Products, *A Competitive Assessment of the U. S. Methanol Industry*, 1985), indicate that methane emissions may be on the order of 1-10 g/10^6 Btu of methanol. However, it is not clear how methane 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 petroleum refineries process crude oil.

For wood boilers, I use AP-42's methane emission factor. The wood-boiler emission factors apply to the burning of lignin to provide steam in a wood-to-ethanol plant. (However, data in Dahlberg et al. [1988] indicate that methane emissions from the combustion of wood chips are almost 100 times higher than those from the combustion of fossil fuels — 300 ppmv in effluent gas versus 5 ppmv.)

For methane emissions from corn-to-ethanol plants, coal-to-methanol plants, and wood gasification, see the notes to Table A.2.

M.7 Other Sources of Methane

M.7.1 Methane from Road Asphalt

Sackett and Barber (1988) estimate that about 15% of the carbon in newly produced asphalt is converted to methane by photochemical and thermal processes. It is not clear if they mean that every year 15% of the carbon is converted, or if total conversion over the lifetime of asphalt can be represented as 15% in one year (the year the asphalt was produced). It seems as though they mean the latter. Assuming this, this methane source can be allocated to highway vehicles, in grams per mile, via the following procedure. According to the 1987 *Census of Manufactures* (U. S. Department of Commerce; industries 2911, 2951, an 2952), 85% of the asphalt produced by refineries in 1987 was paving grade, and the rest was roofing grade or unclassified. In 1987, U. S. refineries produced 158 million barrels of asphalt and road oil (EIA, *Petroleum Supply Annual,* Volume 1, 1988; this is essentially all asphalt). On the basis of these two facts, I estimate that road-grade asphalt produced in 1987 contained 18.7 Tg of carbon. Applying Sackett and Barbers' (1988) 15% factor gives 2.8 Tg of methane in 1987 from the action of sunlight on the new road asphalt. Multiplying this by a 20:1 methane-to-CO_2 mass conversion factor results in 55 Tg of CO_2-equivalent gases, which, when allocated to the 1.92 trillion miles traveled by all vehicles in 1987 (Federal Highway Administration, *Highway Statistics 1987*, 1988), results in 28 g/mi CO_2-equivalents — about 9% of the tailpipe emissions from a 30-mpg car.

This is not an insignificant source, compared with the other transportation-related greenhouse-gas sources. However, road use, or vehicle-miles-traveled (VMT), is not a function of the type of fuel being used. In other words, there is no plausible relationship between production of road grade asphalt and the choice of transportation energy. Therefore, consideration of this source of greenhouse gases would merely add a single, but not small, g/mi increment to *all* the alternatives. Moreover, I am not aware of low-greenhouse-gas-emitting alternatives to asphalt that could be used on a wide scale. For these reasons, I do not consider this source further. (Note again, though, that this source is relatively large and should be included when comparing automobile transportation with mass transit.)

M.7.2 Methane from Nuclear Reactors

Pressurized-water nuclear reactors produce about 900 Ci/yr of a reduced form of methane, $^{14}CH_4$ (Whalen et al., 1989). However, at 4.6 Ci/g of carbon-14, the total methane production is infinitesimal.

M.7.3 Methane from Liquefied Natural Gas

The use of LNG probably results in additional emissions of methane, but these are tiny and can be ignored. About 2.5% of the loaded LNG boils off during a 15-day trip, but this is used by the tanker as fuel (Chem Systems, 1988). More LNG boils off during off-loading at the terminal, but this is presumably reliquefied or used as fuel. It does not seem likely that more than about

0.2% of the cargo is vented. The EIA (EIA, *Annual Outlook for Oil and Gas 1990*, 1990) projects that the United States will import less than 1 TCF of LNG in the year 2010. The 2 BCF of boiloff from this LNG would be about 1% of total venting and flaring in the United States, and less than 5% of the *lower* estimates of leakage from natural gas systems. Obviously, this small number is lost in the uncertainty surrounding venting and flaring and natural gas leaks.

M.8 Greenhouse Credit for Gas Sources Used as Fuel

Methane leaks naturally from natural gas formations, coal mines, and landfills. Leakage from landfills alone may account for 10-20% of global methane emissions (Bingemer and Crutzen, 1987). If one does nothing about these sources, they will continue to contribute to global warming. On the other hand, if one taps these gas sources and puts them to work as fuel, one would be getting "free" energy from the standpoint of global warming: since the gas would have been vented anyway, burning it as a fuel and deriving work would be useful and would not contribute additional CO_2, relative to the counterfactual. In fact, by putting the gas to work, one would not only get useful work but would actually reduce emissions of greenhouse gases, since burning the gas produces CO_2 and work, whereas venting it releases methane, with no work provided. Hence, one can argue that any gas energy, now and in the future, mined from gas that would have been vented should be assigned a net greenhouse credit. Thomas Gold (1988) has advanced this sort of argument.

Currently, very little of the methane emitted from these sources is collected and used as a fuel. However, these sources are likely to be used more in the future, although their total contribution will still be small compared with that from the use of fossil natural gas (*The Potential of Renewable Energy*, 1990).

Appendix N:

Emissions of Nitrous Oxide from Vehicles, Power Plants, and Other Sources

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N.1 Nitrous Oxide as a Greenhouse Gas

Nitrous oxide (N_2O) is a potent greenhouse gas with a radiative forcing of over 100 times that of carbon dioxide (CO_2) per molecule, and a lifetime of more than 100 years (see Appendix O). N_2O also is involved in stratospheric ozone chemistry (Kramlich et al., 1988; Ryan and Strivastava, 1989). However, global emissions of N_2O emissions are small compared with global emissions of methane (CH_4) and CO_2 and, hence, the contribution of N_2O to future warming is expected to be relatively small (*Climate Change*, 1990).

The concentration of tropospheric N_2O has been increasing at about 0.2%-0.3%/yr over the last 40 years or so, from a preindustrial concentration of about 0.285 parts per million volume (ppmv) (Weiss, 1981; Bolle et al., 1986; Khalil and Rasmussen (in Dickinson and Cicerone, 1986); Pearman et al., 1986; Watson et al., 1990). The primary anthropogenic sources of N_2O are oceans and soils (Bowden and Bormann, 1986; Watson et al., 1990). The combustion of fossil fuels appears to contribute only a minor amount (and less than previously thought, as explained below), although uncertainties remain. The main sink for N_2O is photolysis in the stratosphere (Watson et al., 1990).

N.2 Nitrous Oxide Emissions from Vehicles

N_2O is emitted directly from vehicles (Table N.1). The N_2O emissions from catalyst-equipped gasoline-powered light-duty vehicles (LDVs) depend significantly on the type and temperature of the catalyst rather than total nitrogen oxide (NO_x) levels or fuel nitrogen content. Gasoline contains relatively little nitrogen, and so fuel NO_x and fuel N_2O emissions from autos are low. The high temperatures and pressures of the internal combustion engine (ICE) are sufficient to form NO_x thermally, but are evidently inefficient for production of N_2O (Hao et al., 1987). Thus, cars without catalytic converters produce essentially no net N_2O (Table N.1). On the other hand, Weiss and Craig (1976) predict N_2O exhaust concentrations of up to 400 ppmv from autos with platinum reduction catalysts only, according to the reactions:

$$6NO + 4NH_3 \text{ --> } 5N_2O + 6H_2O$$

and

$$2NO + H_2 \text{ --> } N_2O + H_2O$$

TABLE N.1 Nitrous Oxide Emissions from Gasoline, Diesel, and Methanol Vehicles

Type of Vehicle	Emission Control Equipment	Test	Odometer (mi)	Emissions		NO_x/N_2O[a]	Reference
				ppmv	mg/mi		
Gasoline LDV							
NS	No catalytic	NS	NS	0.1-0.2[b]	0	NS	Pierotti and Rasmussen (1976)
1977 AMC Pacer	No catalytic converter; air pump	FTP-75	NS		5[c]	400	Urban and Garbe (1979)
1977 Volvo	Lean operation; no O_2 sensor	FTP-75	200		0	--	Bradow and Stump (1977)
1978 Oldsmobile Cutlass	Oxygen catalytic converter and EGR	FIP-75	NS		18[c]	45	Urban and Garbe (1979)
1978 Chevrolet Malibu	Oxygen catalytic coverter and air pump	FTP-75	NS		8[c]	86	Urban and Garbe (1979)
1978 Chevrolet Malibu	Oxygen catalytic converter and no air pump	FTP-75	NS		66[c]	18	Urban and Garbe (1979)
1978 Ford Granada	Oxygen catalytic converter and air pump	FTP-75	NS		34[c]	58	Urban and Garbe (1979)
1978 Ford Mustang	Oxygen catalytic converter and no air pump	FTP-75	NS		43[c]	38	Urban and Garbe (1979)
NS	Unspecified catalytic converter	NS	NS	up to 1.0	up to 21[d]	NS	Pierotti and Rasmussen (1976)
NS	Oxygen and reduction catalytic converter	NS	NS	up to 10	up to 300[d]	NS	Weiss and Craig (1976)
1978 Oldsmobile Cutlass V-8	Oxygen catalytic converter	FTP-75	NS	0.27-0.77[e]	23[e]	NS	None; EPA unpublished data used

TABLE N.1 (Cont.)

Type of Vehicle	Emission Control Equipment	Test	Odometer (mi)	Emissions		NO_x/N_2O[a]	Reference
				ppmv	mg/mi		
1977 Oldsmobile Cutlass V-8	Oxygen catalytic converter	FTP-75	NS	0.42-1.68[e]	47[e]	NS	EPA unpublished
1978 Ford Pinto	3-way catalytic converter and EGR	FTP-75	New		34-35[g]	19-26	Smith and Black (1980)
1978 Pontiac Sunbird	3-way catalytic converter and EGR	FTP-75	New		32-48[g]	20-40	Smith and Black (1980)
1978 Saab	3-way catalytic converter and no EGR	FTP-75	New		16-35[g]	10-34	Smith and Black (1980)
4 1978-1980 passenger cars	3-way catalytic converter and 2-way oxygen catalytic converter	FTP-75	New		63-101[h]	8-24	Braddock (1981)
2.2 L Renault	3-way catalytic converter; EFI	FTP-75	NS		50-55	NS	Prigent and DeSoete (1989)[i]
Diesel LDV							
1.9 L Citroen	NS	FTP-75	NS		50-58	NS	Prigent and DeSoete (1989)[i]
Flexible-Fuel Vehicle							
With gasoline or indolene							
Ford Crown Victoria	3-way catalytic converter	FTP-75	12,500		13	58	Ford (1988)[j]

TABLE N.1 (Cont.)

Type of Vehicle	Emission Control Equipment	Test	Odometer (mi)	Emissions		NO_x/N_2O[a]	Reference
				ppmv	mg/mi		
Ford Escort	3-way catalytic converter	FTP-75	4,000		116	5	Ford (1988)[j]
With 85% methanol/ 15% gasoline							
Ford Crown Victoria	No catalytic converter	FTP-75	3,000-24,000		2	758	Ford (1988)[j]
Ford Crown Victoria	3-way catalytic converter	FTP-75	16,000		16	52	Ford (1988)[j]
Ford Escort	No catalytic converter	FTP-75	NS		-7	-370	Ford (1988)[j]
Ford Escort	3-way catalytic converter	FTP-75	4,000		105	4	Ford (1988)[j]
<u>Gasoline HDV</u>							
1979 Ford	NS	Chassis HDTC	11,000		96	133	Dietzmann et al. (1981)
1979 International Harvester	NS	Chassis HDTC	15,000		48	272	Dietzmann et al. (1981)
<u>Diesel HDV</u>							
1977 DDC-stroke	NS	Chassis HDTC	60,000		68-85[k]	494-571	Dietzmann et al. (1980)
1979 Caterpillar 4-stroke	NS	Chassis HDTC	7,000		22-35[k]	545-785	Dietzmann et al. (1980)
1979 Mack 4-stroke	NS	Chassis HDTC	69,000		50-58[k]	500-565	Dietzmann et al. (1980)
1979 Cummins 4-stroke	NS	Chassis HDTC	26,000		35-47[k]	517-773	Dietzmann et al. (1980)

[a] This is the ratio of g/mi emissions (not the ratio of ppm), except as noted.

TABLE N.1 (Cont.)

[b] This is gross concentration; the background concentration of 0.3 ppm has not been deducted. Hence, in this case, uncontrolled vehicles actually consume net N_2O.

[c] Figures shown are reported averages of several tests with emission control systems functioning properly. Some test results were higher or lower than averages. Disabling EGR increased N_2O emissions by a factor of 1.5-3.0. (Prigent and De Soete [1989] also found that N_2O emissions increase when EGR is disconnected.) Other malfunctions are relatively unimportant.

[d] I subtract the background concentration of 0.3 ppmv. The authors report only ppmv. For an order of magnitude estimate of mg/mi, I use the ppmv - to - mg/mi conversion factor implied by the EPA (unpublished) data. This conversion factor is only approximate because it depends on the mass flow rate of exhaust gases, which depends on efficiency and engine size.

[e] ppmv is the range reported for three bags in the FTP cold start test (mg/mi as reported in the reference). Similar emission rates are reported for other test cycles.

[f] With the oxygen sensor disconnected, the N_2O emissions disappeared.

[g] Low end is at 0 mi; high end is 15,000 mi. The N_2O did not vary appreciably with the type of gasoline. Emissions were consistently higher in NYCC and lower in HFET than in FTP.

[h] Results at normal FTP ambient temperature (~78°F). At 55-60°F, one vehicle showed a substantial increase in N_2O emissions; another showed a substantial decrease; and two were unaffected.

[i] SO_2 content of the gas is reported to be 12 ppm for undiluted samples and 10 times less for diluted samples. This is well below what appears to be the concentration that actuates artifactual N_2O formation.

[j] Ford reported detailed speciated FTIR data for two flexible-fuel Escorts and two flexible-fuel Crown Victorias, at different methanol/gasoline mixtures, and with different catalysts and catalyst ages. The Escort was tested on indolene; the Crown Victoria was tested on gasoline. A more detailed summary of the Ford test results is given in Table M.3.

[k] Emissions vary with the quality of diesel fuel used.

Note: EPA testing protocol requires that all emissions be corrected for background concentration (i.e., the ambient concentration must be subtracted from the total measured emissions). I therefore have assumed that all reported emissions are net of background, unless it is clear that they are not (as in Weiss and Craig, 1976; Pierotti & Rasmussen, 1976).
NS = not specified. EGR = exhaust gas recirculation. EFI = electronic fuel injection. FTP = Federal Test Procedure. chassis HDTC = 1983 Heavy Duty Transient Cycle, chassis version.
DDC = Detroit Diesel Corporation. NYCC = New York City Cycle. HFET = Highway Fuel Economy Test. FTIR = Fourier Transform Infrared (spectroscopy).

Seinfeld (in Pierotti and Rasmussen, 1976) suggests that N_2O production in 3-way catalysts (those that oxidize hydrocarbon [HC] and carbon monoxide [CO] and reduce nitric oxide [NO]) may occur by a different route:

$$NH_3 + 2O_2 \dashrightarrow N_2O + 3H_2O$$

Alternatively, as suggested by Soete (Ryan and Srivastava, 1989; see also Prigent and De Soete, 1989), the NO can be reduced by CO rather than hydrogen (H_2):

$$2NO + CO \dashrightarrow N_2O + CO_2$$

Prigent and De Soete (1989) conclude that the N_2O is formed at relatively low catalyst temperatures; at higher temperatures, the N_2O itself is destroyed. They show a graph in which N_2O formation, via reduction by CO over a metal catalyst, peaks at 120 ppmv at 500 K and drops to zero at 400 and 650 K. Prigent (in Ryan and Srivastava, 1989) shows N_2O formation across a 3-way platinum-rhodium catalyst peaking at 180 ppmv at about a 635 K catalyst inlet temperature (which implies a slightly cooler temperature across the catalyst). The dependence of N_2O formation on temperature results in higher emissions from a cold-start test than a hot-start test, and relatively high emissions during catalyst light-off (Prigent and De Soete, 1989). This situation has important implications, one of which is that electrically heated catalysts, which will probably be used in the future to reduce cold-start HC emissions, will probably reduce N_2O emissions, too.

The data of Table N.1 agree with the theory discussed above. As shown in Table N.1, N_2O emissions from cars with 3-way catalysts are uniformly higher than from cars without catalysts, although still comparatively low. In further support of this, Lindskog (in Ryan and Srivastava, 1989) has found that after a cold start the concentration of N_2O in the exhaust pipe, measured after the catalytic converter, is two times higher than the concentration measured before the catalytic converter. The N_2O emissions increase slightly with the age of the catalyst (although the trend certainly is not strong; [Table M.3]), and substantially if EGR is disconnected (Table N.1, note c). Colder ambient temperatures may increase or decrease N_2O emissions (Table N.1, note h). However, lower *catalyst* temperatures almost certainly increase N_2O emissions (Ryan and Srivastava, 1989). This discrepancy is possible because, once a vehicle is warmed up, the temperature of the catalyst is not directly related to the air temperature; it is determined by the temperature of the exhaust gases, which are determined by engine load and combustion conditions. The type of drive cycle appears to influence N_2O as well (Table N.1, g).

Table N.1 shows that emissions from heavy-duty diesel vehicles (HDDVs) are similar to emissions from passenger cars equipped with catalytic converters, but that emissions from gasoline-powered heavy trucks may be higher.

N.2.1 Nitrous Oxide Emissions from Vehicles Powered by Alternative Fuels

The preceding analysis indicates that N_2O emissions from vehicles powered by alternative fuels will depend primarily on the operating temperature and composition of catalysts optimized for the particular fuel. Such catalysts are being developed for methanol but not for natural gas, ethanol, liquefied petroleum gas (LPG), or hydrogen. Until such optimized vehicles are developed and tested for N_2O, the existing alternative-fuel data base will have to suffice.

Tests by Ford (1988) on "flexible-fuel" Escorts and Crown Victorias showed that N_2O emissions were roughly the same for cars running on gasoline, 50% methanol, 85% methanol, and 100% methanol, and that these emissions did not change appreciably with the age of the catalyst (Tables M.3 and N.1); the one exception was the Escort with a new catalyst and 100% methanol, which had significantly lower emissions. The Escort emitted about 120 mg/mi, and the Crown Victoria about 15 mg/mi. Based on these results, which show no difference in N_2O emissions between methanol-powered and gasoline-powered vehicles, and the lack of any reason to believe that N_2O emissions from optimized methanol vehicles should be much higher or lower than N_2O from gasoline, I assume the level of emissions is the same for methanol LDVs and gasoline LDVs and the same for methanol heavy-duty vehicles (HDVs) and diesel HDVs.

I also assume parity between natural gas, ethanol, LPG, and petroleum vehicles, although there are no supporting data. I assume that all HDVs emit at the same level, too. However, I assume that hydrogen LDVs emit no N_2O because hydrogen vehicles probably will not have catalytic converters.

As discussed above, N_2O formation is related to the temperature across the catalyst. Different fuel and engine combinations will produce different exhaust gas temperatures and, hence, different N_2O emission rates. One way to project N_2O emissions, then; is to estimate the temperature of exhaust gas as a function of fuel type, engine design, engine operation, and catalyst location, and then estimate N_2O emissions as a function of temperature. (This method still does not account for the effect of different catalyst materials.)

My base-case N_2O grams-per-mile assumptions, based on the data and analysis of Table N.1, and assuming the use of electrically heated catalysts to reduce cold-start emissions, are shown in Table B.2. The data of Table N.1 indicate that most vehicles emit 30-100 mg/mi of N_2O. The average for very low-mileage vehicles with a 3-way catalyst is about 60 mg/mi. The lifetime average for these vehicles may be slightly higher, as indicated weakly by the data of Table M.3, and the notes to Table N.1. The failure of emission controls can increase or decrease N_2O emissions: disabling EGR increases N_2O, but disabling the oxygen sensor eliminates N_2O (Table N.1). However, the use of an electrically heated catalyst, or a close-coupled catalyst to reduce cold-start emissions of HC and CO will probably reduce N_2O emissions from the levels of Table N.1.

It appears, then, that the lifetime, in-use N_2O emission rate for vehicles meeting the new Clean Air Act standards for HC and CO will be about the same as the N_2O emission rate of Table N.1. I assume 60 mg/mi.

N.2.2 Nitrogen Oxide and Nitrous Oxide

One way to get around a lack of N_2O emissions data for alternative fuels is to assume a generic relationship between NO_x and N_2O. (The NO_x emissions are almost always known.)

However, there is no theoretical or empirical basis for assuming that N_2O emissions are directly proportional to NO_x emissions. The data of Table N.1 show clearly that the amount of N_2O emissions and the proportion of NO_x to N_2O can encompass a range that extends over an order of magnitude. For any particular vehicle, a change in operating conditions that increases NO_x emissions may decrease N_2O emissions. Indeed, the high temperatures that favor NO_x formation may destroy N_2O emissions. In regard to N_2O emissions from power plants, Sloan and Laird (1990) write that "the observed correlation between N_2O and NO_x was . . . caused wholly by artifact production" (p. 1205). For these reasons, I do not use an NO_x:N_2O ratio, and instead assume that N_2O emissions per mile are the same for LDVs with the same pollution control equipment, regardless of the fuel used. This assumption undoubtedly will be shown to be incorrect as more data become available, but for now, it appears to be the only supportable assumption.

N.2.3 Artifactual Nitrous Oxide Formation

As discussed below, researchers recently have discovered that N_2O may form in the containers of the gas samples waiting to be analyzed. However, this artifactual formation probably does not occur with the analysis of automobile exhaust gases for four reasons. First, a high concentration of sulfur is necessary to promote the formation of N_2O, and gasoline does not contain enough sulfur to generate the threshold sulfur concentration required. The threshold appears to be about 600-1,000 ppm; in the gasoline-vehicle tests by Prigent and De Soete (1989), the sulfur dioxide (SO_2) concentration was two or three orders of magnitude lower. Second, if the N_2O observed in vehicle tests was primarily a result of artifactual formation, and not of gas reactions across a catalytic converter, there probably would not be a significant difference between N_2O production with and without catalytic converters. Third, the production of N_2O across 3-way catalysts can be explained (see above). Fourth, Prigent and De Soete (1989) also measured N_2O on-line (i.e., without storing the samples), and the results were consistent with the results of the vehicle/chassis dynometer tests.

N.3 Nitrous Oxide Emissions from Stationary Sources

Power plants also emit N_2O. Table N.2 shows emissions of nitric oxide (NO) and N_2O as a function of type of fuel, type of combustion, combustion temperature, sulfur content, and sampling protocol.

Early tests, especially those by Hao et al. (1987), indicated that coal-fired power plants emitted relatively large amounts of N_2O. Oil-fired plants were found to emit less than coal-fired plants, but still a lot, and gas-fired plants were found to emit much less than coal-fired plants. Hao et al. (1987) found that 14% of the fuel nitrogen was converted to N_2O and that the concentration of N_2O in the sample (measured in parts per million volume) was about 29% the of NO_x concentration. With coal-fired plants, the parts-per-million-volume concentration ranged into the hundreds (Table N.2).

TABLE N.2 Measurements of Nitrous Oxide Emissions (ppm)

Combustor Type	Fuel	Sample Collection Method[a]	Temperature (K)	Sulfur Content of Fuel (%) or Flue Gas (ppm)	NO (ppm)	N_2 (ppm)	Reference
On-line sampling							
Six utility boilers	Coal	On-line gas chromatograph	NA	NS	~400	0-4	Clayton et al. (1989)
Tunnel furnace	Coal	On-line gas chromatograph	NA	380-1,450 ppm	~600	2.2-4.2	Linak et al. (1989)
Small boiler	Gas	On-line gas chromatograph	NA	0 ppm	62	0	Linak et al. (1989)
Tunnel furnace	#2 fuel oil	On-line gas chromatograph	NA	236 ppm	189	1.3	Linak et al. (1989)
Once-through	#6 fuel oil	On-line gas chromatograph	NA	3% in oil	220	0	Ryan et al. (1989)
Drum boiler	Bituminous coal	On-line gas chromatograph	NA	6% in coal	386	3.3	Ryan et al. (1989)
Conventional boiler	Coal?	On-line non-dispersive i.r.	NA	1,200 ppm	NA	6-16[b]	Sloan and Laird (1990)
Low-NO_x boiler	Coal?	On-line non-dispersive i.r.	NA	1,200 ppm	NA	12-20[b]	Sloan and Laird (1990)
Opposed-fired boiler	Coal	On-line non-dispersive i.r.	NA	NS	662-982	2-11	Kokkinos (1990)
Tangentially fired boiler	Coal	On-line non-dispersive i.r.	NA	NS	325-498	1	Kokkinos (1990)
Integrated-gasification combined cycle (IGCC)	Coal	On-line non-dispersive i.r.	NA	NS	19	1	Kokkinos (1990)
Front-fired boiler	Oil	On-line non-dispersive i.r.	NA	NS	268	1	Kokkinos (1990)
Front-fired boiler	Gas	On-line non-dispersive i.r.	NA	NS	120	2	Kokkinos (1990)
Combustion turbine	Gas	On-line non-dispersive i.r.	NA	NS	19	4	Kokkinos (1990)

TABLE N.2 (Cont.)

Combustor Type	Fuel	Sample Collection Method[a]	Temperature (K)	Sulfur Content of Fuel (%) or Flue Gas (ppm)	NO (ppm)	N_2 (ppm)	Reference
Effect of sulfur and/or temperature							
Fixed-bed (laboratory)	Graphite	On-line gas chromatograph	1,000	0	1, 000	4	Soete (1989)
Fixed-bed (laboratory)	Graphite	On-line gas chromatograph	1,275	0	1,000	0	Soete (1989)
Fixed-bed (laboratory)	Graphite	On-line gas chromatograph	1,275	NS	700	4	Soete (1989)
Entrained flow reactor	Peat	On-line infrared spectrometer	1,000	0.14% S peat	710	~380[c]	Aho and Rantanen (1989)
Entrained flow reactor	Peat	On-line infrared spectrometer	1,123	0.14% S peat	870	~125[c]	Aho and Rantanen (1989)
Fluid-bed versus high-temperature combustion							
CFBC	Coal	Aluminum bags, stored	NA	NS	130	96	Ryan et al. (1989)
Grate combustion	Coal	Aluminum bags, stored	NA	NS	150	20	Ryan et al. (1989)
CFBC	Coal	Aluminum bags, stored	NA	NS	NA	79-165	Ryan et al. (1989)
Tangentially fired	Coal	Aluminum bags, stored	NA	NS	NA	3- 38	Ryan et al. (1989)
FBC	Coal	On-line infrared spectrometer	NA	NS	NA	100	Ryan et al. (1989)
Coal combustor (CFBC)	Coal	On-line infrared spectrometer	NA	NS	2,000	30	Ryan et al. (1989)
CFBC	Coke	Aluminum bags, stored dry	1,180	NS	32	124[d]	Aamand and Andersson (1989)

TABLE N.2 (Cont.)

Combustor Type	Fuel	Sample Collection Method[a]	Temper- ature (K)	Sulfur Content of Fuel (%) or Flue Gas (ppm)	NO (ppm)	N$_2$ (ppm)	Reference
CFBC	Coke	Aluminum bags, stored dry	1,073	NS	32	170[d]	Aamand and Andersson (1989)
CFBC	NS	On-line non-dispersive i.r.	NA	NS		20-100	Kokkinos (1990)
CFBC	Coal	NS	NA	NS	80-315	NA	Makansi (1991)
Emission controls							
With SCR	Coal	Stainless steel, stored	NA	NS	NA	3-6[d]	Ryan et al. (1989)
Low-NO$_x$ boiler	Coal?	On-line non-dispersive i.r.	NA	1,200 ppm	NA	12-20[e]	Sloan and Laird (1990)
Front-fired boiler with urea	Oil	On-line non-dispersive i.r.	NA	NS	117	23	Kokkinos (1990)
On-line versus stored sampling:							
Tangentially fired	Pulverized coal	On-line gas chromatograph	NA	~1,200 ppm	385	4.5	Ryan et al. (1989)
Wall-fired	Pulverized coal	PVC bag, stored	NA	~1,200 ppm	700	32	Ryan et al. (1989)
Down-fired	Pulverized coal	On-line gas chromatograph	NA	NS	600	<10	Ryan et al. (1989)
Down-fired	Pulverized coal	Stainless steel, stored	NA	NS	150-900	10-220	Ryan et al. (1989)
Conventional wall-fired	Coal?	On-line non-dispersive i.r.	NA	1,200 ppm	~700	11-17[b]	Sloan and Laird (1990)
Conventional wall-fired	Coal?	Plastic bag, 2-3 days, wet	NA	1,200 ppm	~700	53-121[b]	Sloan and Laird (1990)

TABLE N.2 (Cont.)

Combustor Type	Fuel	Sample Collection Method[a]	Temperature (K)	Sulfur Content of Fuel (%) or Flue Gas (ppm)	NO (ppm)	N_2 (ppm)	Reference
Conventional wall-fired	Coal?	Plastic bag· 8-17 days, wet	NA	1,200 ppm	~700	114-211[b]	Sloan and Laird (1990)
Conventional wall-fired	Coal?	Plastic bag, 3-15 days, dry	NA	1,200 ppm	-700	12-18[b]	Sloan and Laird (1990)
Early tests (May have artifactual N_20 formation):							
Utility boiler	Coal	Glass syringe, stored	NA	NS	~450	156-204	Hao et al. (1987)
Utility boiler	Oil	Glass syringe, stored	NA	NS	~170	13- 81	Hao et al. (1987)
Utility boiler	Gas	Glass syringe, stored	NA	NS	~ 60	2- 10	Hao et al. (1987)
Utility boiler	Oil	Pyrex flask, stored	~1,700	NS		25	Weiss and Craig (1976)
Utility boiler	Coal	Pyrex flask, stored	~1,770	NS		25.8	Weiss and Craig (1976)
Utility boiler	Coal	Stainless steel, stored	NA	NS	NA	33-38	Pierotti and Rasmussen (1976)
Utility boiler	Gas	Stainless steel, stored	NA	NS	NA	3-11	Pierotti and Rasmussen (1976)

[a] I assume that gas samples that were not analyzed on-line (i.e., immediately, as they were being produced) were stored.

[b] Measured at 3% oxygen, dry.

[c] ppm was reported at 1,073 K per 10% CO_2. They also reported the percent of fuel nitrogen converted to N_2O at 1,000, 1,073, and 1,123 K. I estimate ppm at 1,000 and 1,123 K by assuming that the ratio of ppmv concentrations at two temperatures is equal to the ratio of fuel conversion percentages at the same two temperatures.

[d] Results reported as 73 mg-N_2O/MJ-coke and 100 mg/MJ, with a conversion factor to ppm of 1.7.

[e] The concentration was 10-25 ppmv before the SCR device.

Notes: CFBC = circulating fluidized-bed combustor. NA = not available. NS = not specified. i.r. = infrared analyzer.
See Ryan and Srivastava (1989) for a similar table.

Coal(?) = Sloan and Laird (1990) do not specifically state which fuel was used, but test conditions and results indicate that the fuel was coal.

N.3.1 Artifactual Formation of Nitrous Oxide

However, more recent research shows that these earlier, high N_2O values obtained for power plants are the result of artifactual N_2O formation. In 1988, Muzio and Kramlich (1988) reported that "the storage of moist combustion products containing SO_2 and NO for periods as short as 2 hours can lead to the formation of several hundreds of parts per million [ppm] of N_2O in the sample containers where none originally existed" (p. 1369). Drying the stored flue gas reduced, but did not eliminate, this artifactual in-sample N_2O formation; removing the sulfur eliminated it. The presence of sulfur in the sample flue gas increased the concentration of N_2O by two orders of magnitude, from less than 5 ppm to several hundred ppm. The effects of moisture and sulfur are illustrated in Table N.2.

Importantly, there appears to be an "activation" level of SO_2. Below 600 ppm of SO_2 in the furnace, the concentration of N_2O in the sample remained quite low, about 15 ppm or less; at 1,000 ppm of SO_2, the concentration of N_2O jumped to about 260 ppm. This suggests that a low-sulfur fuel will not produce enough SO_2 to cause the artifactual, in-sample formation of N_2O. Gasoline is only 0.03% sulfur by weight, compared to about 1% for coal; hence, as argued above, it probably does not contain enough sulfur to cause artifactual formation of N_2O.

The discovery of the artifactual formation of N_2O and of the implication that direct N_2O emissions may not be as high as previously thought is actually consistent with expectations, based on combustion kinetics and experiments, about how much N_2O should be formed. In fact, explaining the earlier, higher levels was proving to be something of a problem, since N_2O formed early in the flame zone should be destroyed rapidly before it reaches the exhaust (Lyon et al., 1989). According to Kramlich et al. (1988), the first major EPA-sponsored workshop on N_2O concluded "with an unsolved mystery: it was possible to explain the fact that . . . combustion systems could produce N_2O . . . [but not] that N_2O could survive the combustion environment which produced it and be present in the discharged flue gases" (p. 4-4). The second workshop did not clear up the mystery; the participants concluded that the then-current data base on N_2O emissions was not good enough to support estimates of global sources and sinks of N_2O, and that it was not reasonable to estimate N_2O as a function of NO_x emissions.

It thus appears that previous estimates of N_2O emissions may be high by at least an order of magnitude. (Compare the results of on-line sampling, with which there is no artifactual N_2O formation, to the results of the earlier sampling, and note the direct comparisons of on-line versus stored sampling [Table N.2].) In further support of this conclusion, I note that Aho and Rantanen (1989) reported that under normal conditions in large combustors, N_2O is less than 2% of the sum of NO and NO_2 concentration — an order of magnitude less than the Hao et al. (1987) finding of 29%. (On the other hand, they found that, with the low-temperature combustion of peat, the conversion of fuel nitrogen (N) to N_2O could exceed the conversion to NO. The issue of low-temperature combustion is addressed below.)

N.3.2 Unresolved Issues in Nitrous Oxide Emission Estimation

There are a few qualifications to the recent finding that stationary-source combustion may produce little N_2O. First, direct N_2O emissions from low-temperature combustion, such as in fluidized-bed combustors and catalytic crackers, may still be substantial. Second, N_2O may be formed in the atmosphere indirectly from direct NO emissions, possibly by the same mechanism as in artifactual in-sample formation.

Another, less important, unresolved issue is the role of emission control systems in N_2O formations. These three issues are taken up next.

1. Modeling and experimental results indicate that fluidized-bed combustors, which operate at lower temperatures than do utility boilers and so produce less NO_x, may produce considerably more N_2O than boilers (Ryan and Srivastava, 1989; Aamand and Andersson, 1989; Kokkinos, 1990; Makansi, 1991). Fluidized-bed combustion (FBC) is used at one stage in the production of gasoline: when the fluidized-bed catalytic cracking unit (FCCU), which breaks the large hydrocarbon molecules of crude into the smaller molecules of gasoline, becomes coated with coke residue from the crude, the coke is burned off the catalysts by fluidized-bed combustion (called, in this case, regeneration). Hence, this step in the refining process may produce nontrivial amounts of N_2O (Lyon et al., 1989). Indeed, the few tests on fluidized-bed combustion using on-line sampling, or other processes that are free of artifactual N_2O formation, indicate substantially higher N_2O emissions than higher-temperature combustion processes (Table N.2).

2. It is conceivable that the reaction responsible for artifactual N_2O formation,

   $$2NO + SO_2 \xrightarrow{H_2O} N_2O + SO_3$$

   could occur in the plumes of power plants and produce N_2O. However, Sloan and Laird (1990) point out that the concentration of the reactants in plumes is much less than in the flue gas samples that generate artifactual formation (and the reaction in the samples is already quite slow) so that in plumes in the atmosphere, the reaction likely will be much to slow too generate much N_2O. Kokkinos (1990) agrees.

   Other artifactual formation routes may exist. Recently, Soete (1989) demonstrated that with combustion in the presence of sulfur (in the form of $CaSO_4$, as might result from emission control devices), the concentration of N_2O dropped with increasing temperature up to about 1,200 K, then increased to a maximum of about 4 ppm at 1,275 K, and then declined again. This demonstration of N_2O formation at high temperatures may reestablish the possibility of significant N_2O formation from the combustion of fossil fuels.

Soete (1989) also suggests that in fluidized-bed combustors, N_2O may be formed from the reduction of NO and the oxidation of reduced sulfur compounds, via a path similar to that suspected of generating artifactual N_2O at room temperature in samples with sulfur and NO:

$$2NO + CaSO_r \dashrightarrow N_2O + CaSO_4$$

The particular sulfur compound Soete investigated, $CaSO_4$, is reduced (and thus "prepared," in a sense, for oxidation by NO) by CO and carbon (C), above 1,200 K. Above about 1,400 K, N_2O is destroyed. With other sulfur compounds or different concentrations of oxygen, N_2O formation as a function of temperature may be different.

Nevertheless, in the absence of specific evidence to the contrary, I assume that real stack gas becomes too dilute too quickly to permit significant formation of N_2O via reactions involving sulfur species.

3. The effect of NO_x control on N_2O emissions from stationary sources is unclear. NO_x control technologies may increase N_2O emissions (Lyon et al., 1989), but perhaps not by much. Linak et al. (1989) reported that combustion modifications to control NO_x emissions from coal-fired units caused only a slight (less than 5 ppm) increase in N_2O concentration. Sloan and Laird (1990) found no significant difference in N_2O emissions between low-NO_x and conventional burners (Table N.2), and Aamand and Andersson (1989) found that injecting ammonia into the combustion zone of circulating fluidized-bed combustors did not affect N_2O emissions. In fact, in one experiment, the use of selective catalytic reduction (SCR) actually reduced the N_2O concentration (compared with the level before the SCR; Table N.2). However, urea injection into an oil-fired boiler may have increased N_2O emissions (Kokkinos, 1990; Table N.2).

Given these preliminary findings, I assume that future emission controls on stationary sources will not dramatically affect N_2O emissions.

N.3.3 My Assumptions

Direct N_2O emissions from high-temperature combustors appear to be low. I assume that N_2O emissions from fossil-fuel boilers are in the upper-middle end of the range of recent estimates from on-line sampling. (I use the upper-middle end to give at least a small weight to the possibility of secondary N_2O formation in post-stack gases and the possibility of increased elevated emissions from plants with emission controls.) My assumptions, in grams per million Btu, are shown in Table D.4.

I assume that petroleum refineries have higher emissions (per unit output), primarily due to the use of fluidized-bed combustion (Table A.2).

To convert the emission values measured in parts per million volume, of Table N.2 to grams per 10^6 Btu of fuel, I calculated grams of N_2O per 10^6 Btu of fuel from data in several references that gave values in parts per million volume, then compared the two. The results are shown in Table N.3. The conversion between parts per million volume and grams per 10^6 Btu depends on the CO_2 concentration of the flue gas, which in turn depends on the fuel used. However, for simplicity, I assume that parts per million volume converts to grams per 10^6 Btu numerically at a 1:1 ratio for all fuels, as is roughly indicated by the results of the Table N.3.

N.4 Nitrous Oxide Emissions from Other Combustion Sources

There are no data on N_2O from trains, ships, natural gas pipeline compressors, farm equipment, loaders, petroleum refineries, well equipment, methanol plants, etc. (The AP-42 report on emissions does not give N_2O values.) Consequently, for most sources, I assume that N_2O emissions depend only on the kind of fuel burned, so that, for example, the N_2O emission rate, in grams per 10^6 Btu, from a ship burning residual fuel oil is the same as the known rate from a power plant burning residual fuel oil. (I use the residual fuel factors for diesel as well as residual fuel.)

As discussed above, I assume a relatively high N_2O emission rate for petroleum refineries due to the low-temperature FCCUs. I assume there will be N_2O emissions from methanol plants because the catalysts used to promote methanol synthesis operate at temperatures lower than those found in utility boilers (Appendix J), and, as discussed above, low temperatures -- at least in FBC units and across automobile catalysts -- seem to produce more N_2O emissions. However, low temperature is not a sufficient condition for high N_2O emissions, and, moreover, there are no data on emissions from methanol facilities. Therefore, to check the importance of these assumptions, I also have zeroed out N_2O emissions from all sources for which there are no real data. The overall results are not appreciably different.

For N_2O emissions from coal-to-methanol plants, corn-to-ethanol plants, wood boilers, and wood gasification, see the notes to Table A.2. (These are basically "guesstimates.")

N.5 Nitrous Oxide from the Corona Discharge from High-Voltage Electricity Transmission Lines

A corona discharge can occur as a result of electric fields set up by high-voltage transmission lines. The strength of these electric fields is proportional to the charge density, which tends to be high on isolated conducting surfaces with small radii of curvature, such as power lines. The electric fields will attract ions in the air with charges opposite those of the conductor. The attracted ions may be accelerated enough to produce more ions upon collision with air molecules, which makes the air still more conducting and increases the corona discharge. This process can form atmospheric trace species, such as N_2O.

TABLE N.3 Converting N_2O from Parts per Million Volume to Grams per Million

Reference	Fuel	ppmv	$g/10^6$ Btu
Aho and Rantanen (1989)[a]	Peat	70-250	75-263
Weiss and Craig (1976)[b]	Coal	25.8	27.2
Pierroti and Rasmussen (1976)[c]	Coal	33	46
Hao et al. (1987)[d]	Coal	156-204	88
Hao et al. (1987)[d]	Oil	13-81	27
Dahlberg et al. (1988)[e]	Coal	100	74
Dahlberg et al. (1988)[e]	Oil	30	16
Aamand and Andersson (1989)[f]	Coke	1.7	1.05

[a] Calculated from their data on heating values, nitrogen content, and conversion of nitrogen to N_2O for 10% CO_2 and 1,100 K.

[b] Using their data, I first calculated 0.03 MT-N_2O/quad.

[c] My calculation is based on their data.

[d] Based on NO_x/N_2O ratio and NO_x emission factors.

[e] As shown in the reference. They also found that CO emissions were less than 200 ppmv, and that CH_4 emissions were 5 ppmv for all fuels except wood chips and firewood; for wood chips and firewood CO emissions were 2,000 ppmv or greater, and CH_4 emissions were 3,000 ppmv.

[f] As stated in the reference, at 6% oxygen.

Note: I use 20×10^6 Btu/ton of coal, 6.2×10^6 Btu/barrel of fuel oil, and 1,020 Btu/SCF of gas for all calculations as rough estimates of worldwide averages.

Hill et al. (1984) calculated that in 1980, the corona energy loss from power transmission lines in the United States probably created 2.3×10^{10} g of N_2O and at most created 4.4×10^{11} g of N_2O. In 1980, U.S. electric utilities delivered $2{,}094 \times 10^9$ kWh of electricity (EIA, *Annual Energy Review 1989*, 1990). Using an N_2O-to-CO_2 conversion factor of 290 (Table 8), the N_2O formed from the corona discharge most likely amounted to 3.2 g of CO_2 equivalents per kWh of delivered fuel, and at most amounted to 61 g of CO_2 equivalents per kWh-delivered fuel. The most likely estimate of 3.2 g/kWh is less than 1% of the emissions from the fossil-fuel power-plant cycle (Table D.4) and only about 5% of emissions from the nuclear-fuel cycle.

The calculations of Hill et al. (1984) are a first-cut, and they point out that this source needs to be better quantified. Fortunately, their most-likely estimate is so small that it does not matter if it is included in the total of greenhouse-gas emissions from electricity generation and use. (I have included it here.) However, the maximum estimate of 61 g/kWh is of the same order of magnitude as emissions from the nuclear-fuel cycle and, hence, cannot be ignored. Therefore, I have included an electric-vehicle scenario analysis using the high-end estimate for N_2O from a corona discharge.

N.6 Nitrous Oxide and Nitrogen Oxide Emissions from Fertilizer

As estimated in Appendix K, a considerable amount of fertilizer is used to grow corn. A small amount of the nitrogen in the fertilizer is released to the atmosphere as N_2O and NO_x (the rest ends up in the crop, in the soil, in water, in microorganisms, or in the air as gaseous nitrogen [N_2]). The net amount of N_2O and NO_x released from corn fields depends on the amount and type of fertilizer, soil and weather conditions, agricultural practices, and more (Eichner, 1990; Anderson and Levine, 1987; Conrad et al., 1983). (The contribution of N_2O from fertilizer to greenhouse gas emissions from the corn-to-ethanol process was first estimated in an unpublished work by Stefan Unasch of Acurex Corporation, Mountain View, California.)

N.6.1 Nitrous Oxide Emissions

Recently, Eichner (1990) reviewed and summarized most of the available data on N_2O emissions from soils, including studies of N_2O emissions specifically from corn fields. The corn studies found that about 0-1.8% (average of about 1.3%) of the nitrogen in the fertilizer had evolved to the atmosphere as nitrogen in N_2O during the study period. Mosier et al. (1986) found that 1.5% of the nitrogen in fertilizer applied to corn was lost as nitrogen in N_2O (compared to 0.4% for barley) and cited an earlier study that estimated a 1.3% loss rate (0.6% for barley). (These two studies were covered in Eichner [1990], but she did not report the amount of fertilizer nitrogen evolved to N_2O.) Anderson and Levine (1987) calculated that 1.2% of the fertilizer applied to a corn plot in April and May was lost as nitrogen in N_2O in June, July, and August (the corn was harvested in September).

Researchers expect that fertilizers continue to produce N_2O after the study period and, further, that some of the applied fertilizer is carried away by drainage and groundwater and evolves to N_2O off the site, beyond the measuring equipment. Eichner (1990) and the Intergovernmental Panel on Climate Change (IPCC) (Watson et al., 1990) double the measured, on-site N_2O emissions to account for these other avenues of N_2O production. Eichner's doubling is a rough estimate; the IPCC's doubling is based on two studies (Ronen et al., 1988; Conrad et al., 1983). Data in Ronen et al. (1988) indicate that *upper limit* emissions of N_2O from the drainage of fertilized land could be twice as high as the average emission rate measured for corn (1.3%, as above). They do not say what the typical situation would be. Conrad et al. (1983) cite one study of the leaching of nitrogen fertilizer into groundwater and another showing that groundwater may be supersaturated with N_2O, and then they assume that the N_2O emissions from groundwater containing leached nitrogen fertilizer may be equal to the measured field losses.

If these doubling estimates are correct, then about 2.6% of the nitrogen in fertilizer ends up as nitrogen in N_2O in the air. However, it must be emphasized that the accounting for off-site evolution of N_2O from groundwater and drainage water is based on virtually no data and is extremely uncertain.

With these data (and caveats), grams of N_2O from fertilizer per 10^6 Btu of ethanol made available to end users can be calculated as:

$$N2O_f = N2O_e \times F \times N \times 453.65 \times 1.57 \times BG \times LL \times O \times V \times S$$

where:

$N2O_f$ = grams of N_2O from fertilizer per gallon of ethanol produced;

$N2O_e$ = grams of nitrogen evolved as N_2O per gram of nitrogen of fertilizer, on-site and during the sampling period (1.3%, as above);

F = factor to account for N_2O evolved after the sampling periods and off the site (assumed here to be 2);

N = nitrogen application rate, in lb-nitrogen/bushel of corn (about 1.23, according to the data of Table K.1);

453.6 = grams per pound;

1.57 = molecular weight of N_2O/molecular weight of N_2;

BG = bushels of corn/gal of ethanol;

LL = liquid loss factor (Appendix B);

O = own use factor (Appendix A);

V = gal ethanol/10^6 Btu (Appendix C); and

S = the share of N_2O emissions that should be assigned to ethanol (assumed here to be at least 0.8, as explained below).

Taking the calculation two steps further (by dividing $N2O_f$ by miles per gallon of an ethanol vehicle and multiplying by the N_2O-CO_2 conversion factor) reveals that N_2O from fertilizer can be a surprisingly large source of greenhouse gases per mile. This prompts one to examine the uncertainty of the values for the variables in the equation for $N2O_f$.

It is clear that BG and N are known well and are not likely to be in error by more than 10 or 20% (the case of low-input agriculture being treated separately). The biggest uncertainties, of course, lie in the measured N_2O formation rate and especially the enhancement factor (F). $N2O_e$

for corn was quite a bit higher than $N2O_e$ for grains, plants, and grasses; however, it was consistently higher and most of the studies did produce similar results (Eichner, 1990; Mosier et al., 1986; Anderson and Levine, 1987). It is not clear if the higher rate for corn can be explained plausibly, or if it is reasonable to expect that further research will bring the rate for corn closer to the rate for, say, grains. It also is possible that the high evolution rate for corn is due to the type of fertilizer. Nevertheless, current research clearly indicates that about 1.3% of fertilizer nitrogen is converted to N_2O during study periods.

The enhancement factor (F) of 2 is a bit better than a guess, but not much. In a scenario analysis, I consider an enhancement factor (F) of 1.25 instead.

The share factor (S), which accounts for the possibility that the use of the by-products of the corn-to-ethanol process will eliminate some N_2O emissions, is virtually impossible to estimate accurately. However, the factor is likely to be high; that is, it is likely that the by-products will not displace nearly as much fertilizer-derived N_2O emissions as will be produced by the corn farming. To understand this, one must recognize two things: first, that the correct way, in principle, to calculate the share factor (S) is to attribute *all* N_2O emissions from the corn farming to ethanol production and then to deduct N_2O emissions from any crop-growing foregone because of the availability of the ethanol by-products; second, not much N_2O would be so foregone. The first point follows because we are, hypothetically, interested in the effects of an ethanol policy, without which there would be no incremental ethanol production and, in most cases, no incremental N_2O emissions from the soil. In other words, if we choose to make ethanol from corn, there will be N_2O emissions from the fertilizer; if we choose not to, there will be no emissions (except for the case in which ethanol is produced by diverting corn feed to ethanol production and the feed is made up of grass).

The second point follows because only some of the by-products will substitute for other crops (distiller's dried grains and solubles [DDGS] may displace some soybean production, but fusel oil and ammonium sulfate will not displace crops) and because the growing of these displaced crops (such as soybeans) probably does not release as much N_2O as does the growing of corn. For example, total N_2O emissions per acre from corn fields , on average, almost four times higher than total emissions from soybean fields (Eichner, 1990). (These are total emissions during the sample period, not just emissions attributo fertilizer; corn and soybean sites were not necessarily treated similarly, and emissions were not adjusted to reflect crop output.) Further, data in Marland and Turhollow (1990) indicate that the ethanol protein by-product associated with 1 acre of corn will displace only 0.7 acre of soybeans so that, all told, the N_2O-from-soybean credit, relative to a unit emission of N_2O from corn growing, is equal to $1.0 \times 0.7 \times 0.25 = 0.175$. This means that net (after credit) emissions of N_2O from the fertilization of corn fields is about 80% of the gross N_2O emission.

In the base-case analysis, I assign 80% of the N_2O emissions to ethanol production (because the by-products effectively displace 20% of the emissions). In the case in which by-product credits are assigned based on the energy value of all by-products, I implicitly assign only 55% of N_2O emissions to alcohol production.

The upshot of this uncertainty analysis is that N_2O emissions from fertilizer use are likely to be high unless the current data are substantially wrong and stacked against ethanol. In scenario analyses, I consider different values for the share factor (S), the enhancement factor (F), and $N2O_e$.

N.6.2 Nitrogen Oxide

There are few data on NO_x emissions from fertilizer use. Anderson and Levine (1987) found that 0.79% of the fertilizer applied to a corn site was lost as nitrogen in NO. Other researchers also appear to be finding that NO emissions are of the same order of magnitude as N_2O (Guenzi, 1991). However, there is considerable uncertainty at this point and the findings are tentative, to say the least.

Anderson and Levine (1987) did not measure NO_2 but reported on a study in which as much NO_2 as NO was emitted. However, because this is insufficient evidence, I ignore NO_2. Anderson and Levine (1988) also did not measure the NO evolved off the site; however, they did find that NO was not emitted from soil saturated with water, which implies that little would be emitted from groundwater or drainage water containing fertilizer. (I assume no off-site production of NO.)

The calculation of NO emissions per gallon is analogous to the calculation for N_2O.

N.6.3 Wood-to-Fuel Sytems

As discussed in the appendix on biomass, some short-rotation intensive cultivations (SRIC) systems may use fertilizer. This fertilizer will result in the evolution of some N_2O and NO_x, with the magnitude depending on the factors mentioned above. The calculation of this emission is similar to that for corn (above), although some of the terms are defined differently. The biggest difficulty in this calculation is that no one has studied N_2O emissions specifically from SRIC systems. It is likely, though, that emissions are less than from corn fields. Data in Eichner (1990) show that grass and plant fields evolve only about 0.25% of fertilizer nitrogen as N_2O, compared with 1.3% for corn. In the absence of specific data, I assume 0.3% (with an enhancement factor of 2.0). I consider higher values in a scenario analysis. For NO, I use the same factor as for corn.

The share factor (S) for N_2O from SRIC systems should be one, because wood-to-fuel systems do not produce by-products.

Appendix O:

Converting Emissions of Methane, Nitrous Oxide, Carbon Monoxide, Nonmethane Hydrocarbons, and Nitrogen Oxides to the Temperature-Equivalent Amount of Carbon Dioxide

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O.1 Background

One can best understand the combined effect on climate of all greenhouse gases emitted as a result of using alternative fuels if all of the emissions are represented by one measure. I do this here by expressing all greenhouse gas emissions in terms of one of the gases. Specifically, I convert methane (CH_4), carbon monoxide (CO), nitrous oxide (N_2O), nonmethane hydrocarbons (NMHCs), and nitrogen oxide (NO_x) mass emissions into the mass amount of carbon dioxide (CO_2) emissions with the same temperature effect. The "same temperature effect" is defined here in terms of degree-years over a period of time, where one degree-year is an increased surface temperature of 1°C for one year. Thus, CH_4 emissions, for example, are converted to the temperature-equivalent amount of CO_2 emissions by calculating the annual mass emissions of CO_2 that would result in the number of degree-years of warming produced by a given annual CH_4 mass emission rate. The number by which CH_4 (or N_2O, CO, NMHC, or NO_x) is multiplied to give the temperature-equivalent mass of CO_2 emissions is called the equivalency factor (EF), and the result of the conversion is a "CO_2-equivalent" amount of CH_4. Thus, 4 g/yr of CH_4 multiplied by, for example, 20 (the equivalency factor) gives 80 g/yr of CO_2-equivalents. This 80 g/yr can be added directly to actual CO_2 emissions to produce a total, single estimate of CO_2-equivalent greenhouse gas emissions.

In order to convert non-CO_2 greenhouse gas emissions into CO_2 emissions having the equivalent temperature effect, one needs to know, for both gases, the relationship between the equilibrium surface temperature and the equilibrium atmospheric concentration, and the relationship between the increase in the yearly emissions and the increase in the equilibrium atmospheric concentration. One also must consider interactions between gases (for example, CO and CH_4) and the ultimate fate of the gases. (CH_4 ends up being oxidized to CO_2 and water vapor [H_2O] by the hydroxyl radical [$OH^{\cdot}$].) Finally, one must pick a period of time to do the analysis; since one is equating degree-years of warming over a period of time, the equation will depend on the length of time chosen. This choice is important.

O.2 The Calculation of Equivalency Factors

A simple calculation of equivalency factors is presented in DeLuchi et al., *Transportation Fuels and the Greenhouse Effect*, 1987. Okken and Kram (1989) (see also Okken, *Energy Policy*, 1990) and Grubb (1989) also derive equivalency factors. However, the most sophisticated and accurate estimates are in recent papers by Lashof and Ahuja (1990), Rodhe (1990), Wilson (1990), and the Intergovernmental Panel on Climate Change (IPCC) (Shine et al., 1990). All four papers use the same basic method, paralleling that used to develop the ozone (O_3) depletion potential for chloroflourocarbons (CFCs). The basic formula (following Lashof and Ahuja, 1990, and Shine et al., 1990) is:

$$EF_i = \frac{\int_{0-t} a_i(t)\ C_i(t)}{\int_{0-t} a_{co_2}(t)\ C_{co_2}(t)}$$

where:

EF_i = equivalency factor for gas (i), relative to CO_2,

$a_i(t)$ = relative radiative forcing of gas (i) as a function of time, in units of: °C-increase per ppm (parts per million)-increase for gas (i)/°C-increase per ppm-increase for CO_2,

$c(t)$ = function expressing the amount of gas (i) that remains in the atmosphere over time, and

t = length of time (years into the future) considered.

All papers express c(t) as an exponential decay function. In Lashof and Ahuja (1990), this function is:

$$e^{-t/r_i}$$

where:

t = defined as above and

r_i = *average* residence time of gas (i), in years (note that this is the average residence time, not the actual residence time of every unit of emission).

The values used in the papers for these parameters are summarized in Table O.1. For many parameters, the authors use essentially the same values; however, this does not mean that, in general, the method and the data are flawless. In fact, there are several key uncertainties, including the effective lifetime of CO_2 and other greenhouse gases; the relative radiative forcing of CH_4 and other gases; the effect of changing concentrations of all gases on their residence times and radiative forcing; the indirect effect of chemical reactions among gases; and the selection of the appropriate timeperiod for the analysis. These problems are discussed briefly in this section (see Shine et al., 1990, for a further discussion of some of these issues).

TABLE O.1 Comparison of Four Calculations of Greenhouse Gas Equivalency Factors

Reference: Parameter	Wilson (1990)	Lashof & Ahuja (1990)	Rodhe (1990)	*Climate Change* (1990)
r_{co2}	100	230[a]	(120[b])	(120[b])
r_{ch4}	9	14.4[c]	10	10
r_{n2o}	150	160	150	150
r_{o3}	0.2	NS	0.1	~0.1[d]
r_{co}	NS	2.1	NA	~0.2[e]
r_{cfc-12}	NS	120	120	130
a_{ch4}	25	44[f]	25	21
a_{n2o}	250	256	200	206
a_{o3}	2,000	NS	2,000	NS
a_{co}	NS	44	NS	NS
a_{cfc-12}	NS	19,500	15,000	15,800
Results[g]				
EF_{ch4}	10 (18)[h]	10	i	9
EF_{n2o}	375	180		190
EF_{co}		2.2		2
EF_{nmhc}				6
EF_{nox}				14
EF_{cfc12}		3,700		4,500

[a] An "effective" lifetime, calculated by the authors from the behavior of CO_2 in ocean-atmosphere-biosphere-CO_2 models. See text for further discussion.

[b] This is the time required for the concentration of CO_2 to decline to 1/*e* (37%) of its original value. It is not the effective, weighted-average residence time actually used by the authors in their calculation of equivalency factors.

[c] The value is 9.6 x 1.5; the 1.5 accounts for the increase in the lifetime of CH_4 caused by increasing CH_4 emissions (a sort of congestion effect).

[d] According to the IPCC (Watson et al., 1990), ozone has a lifetime of "at most several weeks" (p. 29).

[e] According to the IPCC (Watson et al., 1990), CO has a lifetime of "2-3 months" (p. 30).

TABLE O.1 (Cont.)

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[f] The value is 25 x 1.7; the 1.7 accounts for the effect of CH_4 on the concentration of ozone and H_2O.

[g] The equivalency factors on a mass basis, integrated to infinity (500 years, in the IPCC *Climate Change* analysis).

[h] Wilson shows a factor of 10 in her Table 4. However, 10, on a mass basis, is not consistent with the factor of 5 she shows for a mole basis (5 x 2.75 [the relative molar masses of CO_2 and CH_4] = 13.75). Further, I recalculated the mass equivalency factors, using her formula, values, and assumptions, and came up with 6.65 (mole basis) and 18.3 (mass basis).

[i] Rodhe gives value for t = 100 yr only. I have not calculated values for t = infinity, using her formula and input values, because her input data, method, and results are, in most cases, close to those of the IPCC (Shine et al., 1990).

Notes: r_i and a_i are as defined in text. The a_i are on a mole basis.
EF_i = equivalency factor (conversion factor) between gas (i) and CO_2 on a mass basis, with zero discount rate and no time cutoff, except as noted.
NS = not specified.

O.2.1 The Effective Lifetime of Carbon Dioxide

The equivalency factors for all greenhouse gases depend quite strongly on the estimated or assumed effective lifetime of CO_2. The longer the effective lifetime of CO_2, the greater its warming effect, and the less the relative warming effect of other gases. (Recall that in calculating the equivalency factor, the cumulative warming effect of CO_2 is in the denominator.) The problem here is that it is more difficult to estimate a lifetime for CO_2 than for the other greenhouse gases because CO_2 behaves differently.

After a *non*-CO_2 greenhouse gas is emitted, it is removed from the atmosphere by chemical and physical processes and never returns. The average lifetime of a non-CO_2 gas can be defined as the ratio of the atmospheric content (in grams) to the removal rate (in g/yr) (Watson et al., 1990). The CO_2, however, is not destroyed in the atmosphere by chemical processes; rather, it cycles between major reservoirs, such as the ocean and the atmosphere. Thus, a CO_2 lifetime defined only by the absolute CO_2 removal rate will considerably understate the actual concentration of CO_2 because removed CO_2 can return to the atmosphere. The effective lifetime of CO_2 depends on how one models this cycling.

After a pulse injection of CO_2 and the corresponding instantaneous increase in the atmospheric concentration of CO_2, the concentration will decline rapidly in the first decades, (then much more slowly over a period of centuries), and eventually reach a new equilibrium concentration greater than the preinjection concentration (Watson et al., 1990). This process can be modeled with ocean-atmosphere models. Given the results of a model, one can calculate a weighted-average residence time.

Lashof and Ahuja (1990) take this approach. Using the results of an ocean-general circulation model, they assume that 9.8% of a pulse CO_2 emission is, in effect, permanently removed from the atmosphere in 1.9 years; 24.9% in 17.3 years; 32.1% in 73.6 years; 21.0% in 362.9 years; and 13.1% in 1,000 years. The result (the sum of the product of the percentages and the lifetimes) is 253 years. The original model used an infinite lifetime instead of 1,000 years because a portion of the CO_2 is never removed, but since this results in an infinite weighted-average residence time for CO_2, Lashof and Ahuja arbitrarily reduced the infinite-lifetime category to 1,000 years. As they note, the calculation of the weighted-overall-average residence time is extremely sensitive to this choice because 13.1% of CO_2 emissions will have this long life, whatever it is assumed to be. If one chooses 5,000 years, the weighted-average residence time is 750 years instead of 230 years; if one chooses 500 years, it is 160 years.

Shine et al. (1990) also represent the behavior of CO_2 using an ocean-atmosphere model, but they use a different model (although, at one point, the IPCC report refers to the model used by Lashof and Ahuja). Shine et al. (1990) state that 50% of a pulse-injection of CO_2 is effectively removed in 50 years, and that 50% of the remainder is removed in 250 years, so that after 250 years, only 25% is left. A portion is never removed. They do not give the weighted-average effective lifetime, and one cannot calculate it from their data because they do not state how they handle the portion of the CO_2 pulse that has an infinite lifetime. Thus, it is not clear if Shine et al. (1990) use a different effective weighted-average lifetime than do Lashof and Ahuja. They do report a lifetime for CO_2 of 120 years, but this is based on a different definition of the lifetime (the amount of time required for the concentration to decline to $1/e$ [37%] of its original value) and is not the effective weighted-average residence time according to the circulation model (and apparently is not the implicit residence time they use to calculate equivalency factors).

Rodhe's (1990) treatment is similar to that of the IPCC, although she uses yet another circulation model. She also reports a $1/e$ lifetime of 120 years. The upshot of this is that the all-important question of the correct effective weighted-average lifetime of CO_2 is still open to debate and could range from 100 to 500 years or more.

There is some uncertainty regarding the current lifetime of other greenhouse gases, due in part to uncertainty about the sources and sinks of the gases. I have not explored the effect of this uncertainty on the calculation of equivalency factors.

0.2.2 The Relative Radiate Forcing of Methane

All of the papers reviewed here assume that a molecule of CH_4 absorbs 20 to 25 times more energy than a molecule of CO_2 (see a_{ch4} in Table O.1). This value is consistent with what generally has been reported in the literature. However, a recent analysis of spectroscopic and satellite data found CH_4 to be only 13.2 times more effective than CO_2 (Gamache and Golomb, 1990). The use of this lower value would reduce EF_{ch4} by a factor of almost 2.

According to Shine et al. (1990), there is uncertainty in the basic spectroscopic data for other greenhouse gases, particularly CFCs.

O.2.3 Future Lifetimes and Radiative Adsorptive Strength

To date, all calculations of equivalency factors use a single average residence time (r_i) for all of the gases. In reality, though, the residence time of a greenhouse gas will change as its own concentration and the concentration of other gases change. For example, as the concentration of CH_4 and CO increase, the lifetime of CH_4 will increase if the production of OH, the primary chemical sink for both species, remains roughly constant. Ideally, one would express the lifetime of CH_4 as a function of the concentration of CH_4 and CO over time. This has not yet been done. Lashof and Ahuja (1990) do make a simple adjustment to account for this effect in the case of CH_4 (see notes to Table O.1).

Similarly, studies to date have assumed constant values for the relative radiative forcings (a[t]), even though the radiative forcing of a gas depends on both its own concentration and the concentration of other gases, which change over time. For example, there is significant overlap between some of the infrared adsorption bands of CH_4 and N_2O (Shine et al., 1990), which means that the relative radiative forcing of these gases in the future may be less than it is at present, as they both fill the same thermal emission window.

O.2.4 The Effect of Indirect Greenhouse Gases and Chemical Interactions

The production, alteration, and destruction of greenhouse gases and greenhouse-gas precursors can produce other greenhouse gases or greenhouse-gas precursors and affect the concentration, distribution, and lifetime of (other) greenhouse gases. The incompletely understood and difficult-to-model interactions between NO_x, CO, NMHCs, OH, O_3, and H_2O typify this sort of problem (see Appendix Q and Shine et al., 1990, for further discussion).

Current calculations of equivalency factors consider only a few such interactions and, generally, only simply. Lashof and Ahuja (1990) explicitly consider the effect of CO emissions on the lifetime of CH_4 (CO and CH_4 "compete" for the same chemical sink); Wilson (1990) and Rodhe (1990) do not. Wilson considers the effect of CH_4 on tropospheric ozone in some detail. The IPCC chapter (Shine et al., 1990) considers the effect of CH_4 on tropospheric ozone and stratospheric H_2O, and the effect of CO, NMHC, and NO_x on tropospheric ozone. All papers account for the fact that the carbon in all organic species ends as CO_2.

O.2.5 The Length of Time to be Considered in the Analysis

Equivalency factors depend greatly on the length of time considered. For virtually all gases, the equivalency factor is much smaller if one considers a 500-year period rather than a 10-year period, because virtually all greenhouse gases have a greater radiative forcing per mole but a shorter residence time than does CO_2. If, in the calculation of the equivalency factor for a particular greenhouse gas, one considers a period of time less than or equal to the life of the greenhouse gas, the equivalency factor will be relatively high because, over the whole period of analysis, the greenhouse gas will be in the atmosphere and causing a greater forcing than CO_2. However, over a much longer period, the greenhouse gas will have been gone from the atmosphere for most of the time (or will have oxidized to CO_2 and water), so its average warming effect will be less over the longer period than over the shorter period. The importance of the time period can be seen in Table O.2, which shows the IPCC's equivalency factors for 20-, 100-, and 500-year periods.

There are two ways to approach the question of the right time period for the analysis: use a time cutoff, with no "discounting" of warming in future years, or use a nonzero discount rate with no time cutoff. Here I argue that it is inappropriate to use a discount rate for this problem, because the resulting equivalency factor is too sensitive to small changes in the rate. I argue that we should express the extent of our concern about future warming through our choice of a time cutoff. (We should not use a cutoff on top of a discount rate because it is an unnecessary and unfamiliar combination.) I will support this argument by calculating equivalency factors using both the discount-rate method and the time-cutoff method and then examining the difference.

Consider Table O.2 and Table O.3, which show equivalency factors as a function of t, the time horizon or point beyond which all greenhouse warming is ignored. These tables show that moving from a long-term horizon of 100-200 years to a short-term horizon of 20-50 years roughly doubles or triples the equivalency factor for CH_4, CO, and NMHC. This relationship is simple and concrete: shorten the time horizon from a century or two to a few decades, and most non-CO_2 greenhouse gases become two or three times as important. This timescale is a human one, and the effect (a doubling or tripling) is easy to understand.

Alternatively, one can apply a discount rate to future warming, disvaluing all future warming relative to current warming. To illustrate, I will use simple linear models of the

TABLE O.2 IPCC Mass Equivalency Factors[a]

Time (yr)	Greenhouse Gas					
	CH_4	N_2O	CO	NMHC	NO_x	CFC-12
20	63	270	7	31	150	7,100
100	21	290	3	11	40	7,300
500	9	190	2	6	14	4,500

[a] Includes indirect effects (e.g., effect of CO, NMHC, and CH_4 on ozone).

Source: Shine et al. (1990).

TABLE O.3 EF_{ch4} as a Function of Time Cutoff

	EF_{ch4}, Using Formula and Data from:			
Time (t)	Wilson[a] (1990)	Lashof and Ahuja (1990)	Rodhe (1990)	Shine et al. (1990)
20 yr				63
50 yr	60	40		
100 yr	41	24	(15) 30[b]	21
200 yr	28	16		
Infinity	18	10		9[c]

[a] I follow Wilson (1990) in assuming that 1 mole of CH_4 produces 0.85 mole of ozone and 0.85 mole of CO_2. However, I also account for the fact that, since the ozone and CO_2 are produced from the decay of CH_4, and CH_4 lasts an average of nine years (according to her) before it decays, the ozone and the CO_2 will be around for nine fewer years to cause warming. This means that if the warming effect of a mole of CH_4 is calculated from $t = 0$ to $t = 100$ years, the warming effect of the 0.85 mole of ozone and CO_2 produced by the CH_4 should be calculated from $t = 9$ to $t = 100$ years.

[b] Rodhe calculates the direct heating effect of CH_4, and then appears to estimate that the indirect effects of CH_4 (formation of ozone and H_2O, and eventual oxidation to CO_2) are equal to the direct effect, so that the total heating effect is about twice the direct effect. In this table, the lower number shown is the calculated direct effect; the upper number is the (assumed) total effect.

[c] Integration to t = 500 yr.

Note: All equivalency factors are on a mass basis.

atmospheric fate of CH_4 and CO_2 emissions. Assume that a mole of CO_2 in effect stays in the atmosphere 200 years and causes a unit increase in temperature every year for the 200 years. Assume that an emission of a mole of CH_4 lasts only 10 years as CH_4, but causes a 25-unit increase per year for those 10 years, after which it is converted to CO_2 and causes a unit increase per year for the next 200 years. (I ignore the effect of CH_4 emissions on ozone and H_2O.) With a zero discount rate, the equivalency factor, on a mass basis, is:

$$2.75 \times (25 \times 10 + 200)/200 = 6.19$$

Now apply a discount rate of *i*. The three separate streams -- CH_4 for 10 years, CO_2 for 200 years, and CO_2-from-CH_4 for 200 years (this last stream starting in year 10) can be viewed as annuities and discounted by the formula:

$$([1 + i]^n - 1)/(i[1 + i]^n)$$

where:

n = number of years and

i = discount rate.

In addition, the 200-year stream of CO_2-from-CH_4, starting in year 10, must be discounted back to year zero by $1/(1 + i)^{10}$. The resulting mass equivalency factors (EF_{ch4}), as a function of interest rate i, are shown in Table O.4.

Note that the equivalency factor changes sharply with small (perhaps meaningless) changes in the interest rate. For example, from a discount rate of 1% to a rate of 3%, the equivalency factor doubles. It is difficult to imagine that many people can meaningfully distinguish between an interest rate of 1% and an interest rate of 3%, yet the difference between these is a factor of 2 in EF_{ch4}.

It is likely that people can more naturally represent their balancing of present and future concerns by referring to blocks of time within the realm of human experience than by trying to understand quantitatively how a 1% interest rate treats the future differently than a 3% rate. All of us understand human timescales, but few can attach much meaning to small differences indiscount rates. Because of the difficulty in choosing between small interest rates and the sensitivity of the result to small changes in the rate, I feel it is better to pick a time cutoff (or time horizon) with no interest rate.

TABLE O.4 Methane Equivalency Factor as a Function of the Discount Rate

i (%)	EF_{ch4}
0.0	6.2
0.5	7.9
1	10.0
3	19.7
5	28.2
7	35.2

Notes: *i* = discount rate (see text for explanation).

O.3 Review of Estimates of Equivalency Factors and Assumptions Used in This Analysis

O.3.1 Methane

Wilson's estimate of the equivalency factor for CH_4 appears to be relatively high (Table O.1, note g). Part of the explanation is that Wilson uses a much shorter average residence time for CO_2. Recall from above that a shorter residence time reduces the cumulative warming effect of CO_2 and thereby increases the warming effect of the other gases relative to CO_2. The main reason for Wilson's higher equivalency factor, though, is her assumption of a relatively long lifetime for ozone -- twice as long as that assumed by the IPCC (Shine et al., 1990) and Rodhe (1990). Wilson adds the warming effect of 1 mole of ozone to the direct warming effect of each mole of CH_4 that oxidizes in the troposphere, because the oxidation of 1 mole of CH_4 on balance leads to the formation of 1 mole of ozone. Also, because of the very high relative radiative forcing of ozone (Table O.1), a difference of a factor of 2 in the lifetime makes a big difference in the final result.

The lifetime of tropospheric ozone is probably on the order of a few weeks rather than a few months (Bolle et al., 1986; Wuebbles and Edmonds, 1988; Watson et al., 1990). With this correction to Wilson's calculation, there is reasonable agreement concerning the equivalency factor for CH_4.

In this report, I use values very slightly less than those of the IPCC, in token deference to the recent evidence that the radiative forcing of CH_4 may be less than heretofore thought (see discussion above).

O.3.2 Nitrous Oxide

Three (Rodhe, 1990; Lashof and Ahuja, 1990; and Shine et al., 1990) of the four papers calculate virtually the same equivalency factor for N_2O. (Rodhe's equivalency factor for a 100-year period is 300, very close to the IPCC's equivalency factor of 290, as shown in Table O.2.) This agreement is due in part to N_2O having few indirect effects on climate and, hence, being relatively easy to model.

Wilson's equivalency factor for N_2O is higher because she uses both a relatively short effective lifetime for CO_2 (which reduces the cumulative warming impact of N_2O) and a relatively high molar radiative forcing for N_2O. I note that if Wilson's calculation is redone with the 230-year CO_2 residence time used by Lashof and Ahuja (1990), the result, for an infinite time horizon, is an EF_{n2o} of 160, which is close to the results of the others.

O.3.3 Carbon Monoxide, Nonmethane Hydrocarbon, and Nitrogen Oxide

The IPCC (Shine et al., 1990) and Lashof and Ahuja (1990) calculate the same equivalency factor for CO for the long-term horizon (Table O.1).

Only the IPCC has estimated equivalency factors for the ozone precursors, NMHC and NOx. As discussed in Appendix Q, it is difficult to calculate the warming effect of NMHC and NOx; hence the equivalency factors used here must be viewed as *very* preliminary.

The IPCC gives a generic equivalency factor for NMHCs; it does not give specific factors for individual hydrocarbons and probably does not intend that the NMHC factor apply as such to emissions of alcohols. To adapt the equivalency factor of NMHCs to include alcohols and to enable it to distinguish, crudely, between individual hydrocarbons, I will convert the equivalency factor to a carbon-weight basis. The rationale for this is twofold. First, part of the warming effect of NMHCs and alcohols (indeed, of any organic compound) is due to the eventual oxidation of the carbon to CO2; hence, part of the mass-based warming effect of any organic compound will be a function solely of the carbon content. Second, it appears that, at least for some NMHCs, the higher the carbon number, the higher the ultimate ozone-forming potential. Seinfeld (1989) notes that higher-carbon alkanes consume more nitric oxide (NO), which shifts the NO, atomic oxygen (O), molecular oxygen (O_2), ozone (O_3), and NO_2 equilibrium in favor of O_3. Also, reactivity increases with carbon number within any family of compounds, and reactivity Pmay be related to ozone-forming potential globally (it certainly is related to ozone formation in urban areas).

Accordingly, I convert the IPCC's factor for NMHCs to a carbon (C) basis. I assume that the IPCC based its equivalency factor for NMHCs on the current NMHC mix, which the IPCC says is mostly terpenes, isoprenes, and alkenes (Shine et al., 1990). For this mix, the average carbon-weight content is about 85%. Thus, the values of Table 8 are equal to the IPCC values of Table O.2 divided by 0.85. This gives an equivalency factor based on the carbon content of NMHCs and alcohols.

The most important effect of expressing the equivalency factor of NMHCs on a carbon basis is to exclude the weight of oxygen in methanol and ethanol emissions.

O.3.4 The Base-Case Time Horizon and Equivalency Factors of This Analysis

As we saw above, the equivalency factors are a function of the time horizon. This analysis follows the IPCC (*Climate Change*, 1990), and considers time periods of 20, 100, and 500 years. I use the 100-year time period as the base case. However, I use these horizons only to give a range of results; I do not endorse the use of a short (less than 50- or 100-year) horizon. In fact, there is scant justification for the use of a 20-year time horizon.

The case for a long time horizon is obvious: global warming is a very long-term problem and will affect many generations to come. Several greenhouse gases persist for centuries. Even if net anthropogenic emissions of greenhouse gases are eliminated, the global climate still will be in flux for decades, primarily because of the large thermal inertia of the ocean. And the deep ocean, which stores CO_2, turns over on a time scale of centuries. Therefore, to account for most of the climatic effects of current emissions of greenhouse gases, one must extend the analysis for decades at least; otherwise, the warming effects of some emissions will be ignored.

Of course, one reasonably might attach less importance to the very long-term than to the middle-term. This might be a reason for choosing a 100-year time period over a 500-year period, but it would not seem to justify choosing a 20-year time period, since such a short view would discard a large portion of the period of climate change.

An argument recently advanced in support of using a very-short-term horizon is that we should be concerned with the *rate* of warming as well as with the ultimate increase in equilibrium surface temperature, and that the short-term horizon represents the effect of different gases on the rate of global warming (Wilson, 1990; Shine et al., 1990). The implicit claim is that the greater the 20-year warming factor for a particular greenhouse gas, the greater the rate of warming due to that gas. But I would argue that, while the rate of warming certainly is important (because the ability of biological systems to adapt to change depends on the rate of change), equivalency factors based on a short-term horizon do not tell us anything about the relative contribution of different gases to the rate of global warming.

A 20-year equivalency factor, as defined now, tells us how much radiative forcing a unit emission of a gas causes over 20 years relative to a unit emission of CO_2. It does not tell us how the forcing occurs over the 20 years, or what happens after the 20 years, as a result of the one-time emission of one unit of the gas. (One can calculate these trajectories, but the point is that the equivalency factor itself tells us nothing about them.) More importantly, the overall *global* rate of warming depends on the total quantity and rate of emissions of *all* gases, and the equivalency factors as calculated now consider neither total emissions nor the rate of emissions -- they tell us the relative radiative forcing of a one-time emisssion of a unit of a gas. Thus, not only does the equivalency factor not characterize the warming trajectory due to a unit emission of gas, it does not tell us anything at all about the *global* warming trajectory of a *global* emissions scenario involving particular overall amounts of the gas. So, not only is a short-term equivalency factor not a cardinal measure of the relative contribution of a gas to the global rate of warming (an equivalency factor of 18.6 is not twice as "fast" or "bad" as an equivalency factor of 9.3), it is not even an *ordinal* measure (an equivalency factor of 18.6 does not even indicate a faster or worse overall rate of warming than an equivalency factor of 9.3). The current equivalency factors, then, do not help us evaluate the effect of any particular policy on the rate of global warming.

To best see that equivalency factors cannot be used to evaluate the effects of gases on the rate of warming, one can compare the equivalency factors as calculated now with a conceptually sound measure of the relative contribution of different gases to the rate of global warming. To make a conceptually sound measure, one might calculate the global warming trajectory caused by a specific global emissions scenario, then change hypothetical emissions of one gas only and recalculate the warming trajectory. One would repeat this, changing emissions of different gases one at a time. One then would try to relate the change in emissions of each gas to some aspect of the corresponding change in the warming trajectory, such that the relation represented the relative contribution of the particular gas to the rate of warming. If one is able to represent or define the rate of warming in a way that captures the importance, or "bad" effects, of a "fast" rate of warming, then one has a measure of the relative contribution of emissions of a particular gas to the undesirability of a "fast" rate of warming for the specific emissions scenarios considered. (Note, though, that it will not be easy to define an appropriate rate of warming because we do not understand the specific relationship between warming trajectories and damage to the planet. How will we compare a very fast warming over 10 years to a slower, but still rapid, warming over 20 years? Definitions and weights will likely seem quite arbitrary.)

There are two things to note about this conceptually sound measure of the relative contribution of different gases to the rate of warming. First, it looks nothing like the equivalency factors as calculated now. The correct measure relates changes in global emissions to some aspect of a global warming trajectory and defines and ranks rates of warming; the equivalency factors calculated now involve neither total emissions, nor warming trajectories, nor definitions of warming rates. Second, any particular measure of the relative contribution of a gas to the overall rate of climate change (however defined) probably would hold only for the specific emissions scenario considered. Different baseline emissions scenarios (and there are innumerable emissions scenarios) probably would give quite different results. Thus, one probably could not claim that a rate-of-warming factor calculated for a particular scenario had *general* implication. (Note that the equivalency factors, which represent a gas's relative contribution to equilibrium warming, are also a function of emissions, implicitly, because the relative radiative forcing and the lifetime of the gases are a function of their concentration, which in turn is a function of emissions. However, it is likely that the current equivalency factors are less sensitive to changes in the implicit underlying emissions scenarios than the rate-of-warming factor outlined here would be to changes in the explicit emissions scenario.)

I conclude, therefore, that there is, at present, little justification for the use of a 20-year time horizon in the calculation of equivalency factors.

Appendix P:

Greenhouse Gas Emissions from Making Materials for Vehicles, Power Plants, Pipelines, Ships, Trains, etc., and from Assembling Vehicles

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P.1 Greenhouse Gas Emissions from Assembling Vehicles

It takes energy to assemble vehicles, and the use of most forms of energy produces greenhouse gases. The amount of energy used to assemble vehicles is related to the weight of the vehicle being assembled, and because alternative-fuel vehicles (AFVs) generally will not weigh the same as the gasoline- and diesel-fuel vehicles they replace, the assembly of AFVs will require a different amount of energy than the assembly of similar gasoline and diesel-fuel vehicles and will produce a different amount of greenhouse gases. Therefore, emissions from vehicle assembly should be included in the estimation of the greenhouse effect of alternative transportation fuels.

To calculate emissions of greenhouse gases from vehicle manufacture, it is most convenient to start with the amount of energy required to assemble 1 pound (lb) of vehicle. If one knows the total empty weight of one class of vehicle (e.g., passenger vehicles), one can calculate the total energy required to assemble that particular class of vehicle. With data on the kind of energy, one can calculate emissions of greenhouse gases per vehicle assembled, and with data on the lifetime mileage of that class of vehicle, one can calculate emissions of greenhouse gases per mile for that particular class of vehicle. Emissions per mile for any other class of vehicle of X (e.g., heavy trucks) can be calculated by multiplying emissions per mile for the known class (Y) by the ratio of weights, X:Y, on the assumption that assembly energy is proportional to vehicle weight.

In this section, I calculate assembly energy per lb-vehicle by using current data on total energy consumption by vehicle manufacturers, total amount of each class of vehicle produced, and weight of each class of vehicle. I compare the result with two older estimates of Btu/lb. I then break down the assembly energy into electricity and types of fuel. Finally, with data on lifetime mileage by class of vehicle, I calculate greenhouse gas emissions per mile due to vehicle assembly.

I assume that energy required for assembly is directly proportional to vehicle weight for all vehicles — alternative-fuel vehicles, petroleum-fuel vehicles, electric vehicles (EVs), and internal-combustion-engine vehicles (ICEVs) — regardless of the materials they are composed of. I allow for the possibility that AFVs may have a different number of lifetime vehicle miles traveled (VMT) than the petroleum-fuel vehicles they replace, and so they have a different grams-per-mile (g/mi) emission rate due to vehicle assembly. (A longer lifetime spreads the assembly energy and emissions out over more miles and lowers the g/mi per mile emission rate due to assembly).

I include the weight of alternative-fuel storage systems (e.g., batteries, high-pressure tanks) in the total vehicle weight for the purpose of estimating emissions from vehicle assembly. This implicitly assumes that it takes a battery manufacturer as much energy to assemble a pound of vehicle as it does a vehicle manufacturer to assemble a pound of vehicle (excluding the storage system). This is probably not exactly right, but it is probably a good approximation.

P.1.1 Energy Used by Vehicle Manufacturers

In 1985, the manufacture of transportation equipment (e.g., aircraft, railroad equipment, watercraft, and motor vehicles*) consumed 0.321 quad of primary energy (EIA, *Manufacturing Energy Consumption Survey*, 1988). (This does not include energy used to make the materials used in motor vehicles, which I consider separately, below). Primary energy is the energy value of all fuels and electricity used by the manufacturer. The EIA *Manufacturing Energy Consumption Survey* (MECS) does not report energy consumption by subsector of the transportation equipment manufacturing industry. The U.S. Bureau of the Census's *Annual Survey of Manufactures* (ASM) reports that the manufacture of motor vehicles and motor vehicle equipment consumed 0.068 quad of electricity (Btu-electric) in 1985, but it does not give further data on energy consumption by the motor vehicle manufacturing sector. (The MECS and the ASM estimates of net electricity consumption by manufacturers of transportation equipment are within 5% of each another.)

There are at least two ways to infer energy consumption by motor vehicle manufacturing by using the MECS and ASM data. First, one can assume that the motor vehicle sector's share of total energy consumption in the transportation equipment industry is equal to the dollar value of shipments of motor vehicles, divided by the dollar value of shipments of all transportation equipment. This ratio was 0.61 in 1985 (ASM, 1988).

Second, I note that motor vehicle manufacturers accounted for 58.6% of the total spending for energy by transportation equipment manufacturers, 57.9% of total electricity use by all transportation equipment manufacturers, and 67% of total spending for fuels other than electricity. These statistics are close to the 61% ratio based on total value, and they suggest that motor vehicle manufacture consumed about 60% of the 0.321 quad consumed by all transportation equipment manufacturers. This results in 0.1926 quad.

P.1.2 Curb Weight of Vehicles, by Class

Heavenrich (1988) provides data on the average gross vehicle weight (GVW) and assumed payload of 1985 model year cars and light trucks weighing less than 8,500 lb. With these data,

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* The standard industrial classification (SIC) category "Motor Vehicles and Motor Vehicle Equipment" (SIC 371) includes establishments that manufacture and assemble complete passenger cars, trucks, and buses; establishments that manufacture car and truck chassises and bodies; establishments that manufacture motor vehicle parts and accessories; and establishments that manufacture motor homes and truck trailers. It does not include manufacturers of motorcycles, automotive glass, automotive batteries, tires and tubes, stamped body parts, carburetors, and other minor auto parts. Therefore, my estimate of energy used in vehicle manufacturing excludes energy used to make some parts.

one can calculate the average curb or "empty" weight of passenger vehicles and light trucks: it is simply the GVW less the standard assumed payload (300 lb).

Unfortunately, the same cannot be done for medium and heavy (which together I call "other") trucks, because even if the average GVW could be calculated, there is no standard assumed payload that can be deducted from the GVW to yield the curb weight. Instead, I calculate the average curb weight for other trucks (cab and trailer) from data on the average curb weight and number of trucks, by three weight classes:

$$ACW_o = F_{8500} \times CW_{8500} + F_m \times CW_m + F_h \times CW_h$$

where:

ACW_o = average curb weight of other trucks (medium and heavy trucks),

F_{8500} = the number of trucks between 8,500 and 10,000 lb GVW divided by the number of all trucks weighing more than 8,500 lb (0.14, according to data from the Motor Vehicle Manufacturer's Association [MVMA]),

CW_{8500} = curb weight of trucks between 8,501 and 10,000 lb GVW (I assume 8,950 lb, which is the midpoint GVW of 9,250 minus a 300-lb payload),

F_m = number of medium trucks (GVW 10,001-19,500 lb) divided by the number of all trucks weighing more than 8,500 lb (0.05, according to the MVMA),

CW_m = curb weight of medium trucks (14,000 lb; my assumption),

F_h = number of heavy trucks (GVW of more than 19,500 lb) divided by the number of all trucks weighing more than 8,500 lb GVW (0.81, according to the MVMA), and

CW_h = curb weight of heavy trucks, cab plus trailer (20,000-25,000 lb, according to the manufacturers, dealers, and drivers I spoke with).

The resulting average weights for passenger cars, light trucks, and heavy trucks are shown in Table P.1. These weights are multiplied by the number of vehicles manufactured in each class (discussed next) to produce total weight of manufactured vehicles, which can then be divided into total manufacturing energy to yield Btu/lb.

P.1.3 Number of Vehicles Manufactured (needed to calculate Btu/lb-vehicle)

The MVMA (1987) reports production of passenger cars and production of motor trucks and buses, but does not disaggregate production of motor trucks and buses by gross vehicle weight. However, they disaggregate factory sales of motor trucks and buses by vehicle weight (I need production data because manufacturing energy consumption is related to units produced, not units sold). I have assumed that the (unknown) ratio, number of trucks produced in a given weight class X divided by total production of trucks and buses, is equal to the (reported) number of trucks of weight class X sold by the factory divided by total factory sales of trucks and buses. This seems reasonable, since production very nearly equals factory sales.

One further assumption is needed here. The MVMA factory sales data define light trucks as those that weigh 6,001-10,000 lb GVW, but the U. S. Environmental Protection Agency (EPA) weight data used above define light trucks as those that weigh less than 8,500 lb GVW. Therefore, I need an estimate of the proportion of trucks in the 6,001-lb to 10,000-lb GVW category that weigh more than 8,500 lb. According to *Ward's Automotive Yearbook*, which reports the weight of every light truck made in the United States, few trucks weigh 8,500-10,000 lb GVW compared with the number that weigh less than 8,500 lb. On the basis of *Ward's Yearbook*, I assumed that 10% of the 6,001-lb to 10,000-lb trucks were between 8,500 and 10,000 lb.

P.1.4 Comparison With Other Estimates

The Btu/lb result of the preceding analysis is shown in Table P.1, along with two older estimates. In a thorough analysis, Berry and Fels (1972, 1973) used data from the 1967 *Census of Manufactures* to calculate the energy required to assemble autos and produce the materials used in autos. They calculated a higher energy requirement/pound auto assembly in 1967 than I do for 1987 assembly, which probably results, in large part, from improvements in energy-use efficiency over the 20-year period. Part of the difference, though, results from the fact that Berry and Fels (1972) counted energy for assembly of certain parts that I omitted, such as carburetors and other minor vehicle subsystems.

Hudson's (1982) estimate is between mine and that of Berry and Fels, which is consistent with an assumption of increasing efficiency of manufacture. The rest of the difference between my estimate and Hudson's estimate might result from different definitions of vehicle fabrication and vehicle manufacture. Hudson does not define vehicle fabrication in terms of SIC categories; it may be the same as the large group, SIC 371, "Motor Vehicles and Motor Vehicle Equipment," or it may include just the largest subcomponent, SIC 3711, "Motor Vehicles and Passenger Car Bodies." Given these considerations, I assume a Btu/lb value as shown in Table P.1 .

TABLE P.1 Data Used to Estimate Energy (Btu) Consumed to Assemble a Pound of Vehicle

Vehicle	Curb Weight (lb)[a]	Vehicles Made (10^6)[a]
Passenger car	3,121	8.185
Light truck[b]	3,663	3.035
Other truck[b]	20,625[c]	0.430

0.1926	Calculated quads used to assemble motor vehicles[d]
4,239	Calculated Btu/lb-vehicle[e]
Compare with:	
5,657	Hudson (1982)
6,208	Berry and Fels (1972)
5,000	Used here[f]

[a] See text for discussion and sources.

[b] "Light trucks" weigh less than 8,500 lb gross. "Other trucks" weigh more.

[c] I calculated an average 31,500 GVW for trucks weighing more than 8,500 lb GVW.

[d] From MECS and ASM data. See text for discussion and methods.

[e] 0.1926 quad divided by weight of all vehicles made.

[f] I have used a value higher than calculated from MECS and ASM data because, as discussed in the text, those data do not include all relevant assembly operations.

P.1.5 Breakdown of Total Energy

According to MECS, in 1989, about 31% of the total energy used to make transportation equipment in the Midwest was electricity (as Btu per electric); 8% was oil, 43% was natural gas (NG), and 18% was coal. (I use the distribution for the Midwest region because about two-thirds of all autos are manufactured in the Midwest.) Here, I will assume 35% for electricity rather than the 31% of MECS, because 0.35 × 0.1926 quad of total energy consumed by vehicle manufacturers (see above) equals 0.068 quad electric, which is the figure given in the ASM for the manufacture of motor vehicles. I then adjust the other energy shares proportionately (7% oil, 41% NG, and 17% coal).

P.1.6 Electricity Mix Used by Auto Manufacturing Plants

Emissions of greenhouse gases from the use of electricity to assemble autos depend on the mix of fuels used to generate electricity. This mix is determined by the location of assembly plants and the generating mix of the utilities that serve them. *Ward's Automotive Yearbook* (as reported in *Motor Vehicle Facts and Figures, 1987*) shows auto and truck assembly by city and state. This data, combined with data from the Energy Information Administration (EIA) on the mix of fuels used to generate electricity in every U. S. utility, and data on the service area of every U. S. utility (Electrical World, 1988), gives the breakdown of electricity use by fuel type for auto and truck and assembly in the United States. (I have assumed that auto- or truck-manufacturing energy divided by total manufacturing energy is equal to the weight of all autos produced, or all trucks produced, divided by the weight of all vehicles produced.) Electricity mixes by city and city shares of total motor vehicle manufacturing energy are shown in Table D.1.

P.1.7 Lifetime Mileage of Vehicles

I estimate the average lifetime mileage of petroleum-fuel passenger cars, light trucks, and other trucks as the cumulative mileage at the median age of scrappage for each vehicle. The *Transportation Energy Data Book 9* reported the medium age of scrappage for cars, light trucks (up to 10,000 lb GVW), and heavy trucks (more than 26,000 lb GVW). The California Air Resources Board (1986) shows cumulative mileage as a function of age for cars, light-duty trucks, and heavy gasoline and diesel trucks. (I assumed half of the heavy-duty trucks were diesel and half were gasoline). For this calculation, I ignore the discrepancy between the 8,500-lb and the 10,000-lb definition of a light truck. I also ignore the fact that cumulative mileage at median lifetime is not the same as average lifetime mileage (the number I want), the latter being total miles traveled by all vehicles in a class before scrappage, divided by the number of vehicles. The results for petroleum-fuel heavy-duty vehicles (HDVs) and light-duty vehicles (LDVs) are shown in Table P.2.

In the base case, I assume that petroleum, methanol, ethanol, NG, hydrogen, and liquefied petroleum gas (LPG) light-duty vehicles all have the same lifetime (Table P.2). However, electric

TABLE P.2 Lifetime of Vehicles and Fuel-Storage Systems in Miles

Vehicle	Petroleum	Alcohol	NGVs	EVs	Hydrogen	LPG
Passenger car	108,000	108,000	108,000	153,000[a]	108,000	108,000
Light duty truck	163,000	163,000	163,000	163,000	163,000	163,000
Heavy-duty truck	335,000	335,000	335,000	NA[b]	335,000	335,000
LDV fuel storage	108,000[c]	108,000[c]	150000[d]	104,000[e]	300,000[f]	108,000[g]
HDV fuel storage	335,000[c]	335,000[c]	335,000[d]	NA[b]	600,000[f]	3,350,00[g]

Lifetime for petroleum-fuel vehicles from data in CARB (1986) and *The Transportation Energy Data Book*, edition 9. See text for more details.

[a] Electric vehicles will last longer than ICEVs (DeLuchi, 1989). I assume a 42% increase in life.

[b] EVs will not be used in HD applications.

[c] Tanks for liquid fuels are cheap and durable. They generally last the lifetime of the vehicle but are not valuable enough to salvage at the end of the life of the vehicle. Hence, I assume that liquid-fuel tanks have the same life as the vehicle.

[d] The situation with CNG tanks is complicated. U. S. DOT currently demands that composite tanks be destroyed after only 15 years, but manufacturers guarantee them for much longer than this, and Italy allows steel tanks to last 30 years (see DeLuchi, Sperling, and Johnston, 1987). If the 15-year limit remains, CNG tanks could last for a little longer than one HDV lifetime, and almost 1.5 LDV lifetimes. If the limit is raised, which certainly seems reasonable in light of the manufacturer warranty and the performance of the tanks to date, tanks could last for much longer. However, for tanks to last beyond one vehicle lifetime, they will have to removed from the scrapped first vehicle and reinstalled in another. It is not clear how such large-scale recycling might be restricted or what it might cost.

In the base case, I have assumed conservatively that CNG tanks last 15 years — about one HDV lifetime and almost 1.5 LDV lifetimes. In a scenario analysis, I consider a 30-year life for CNG tanks in both LDVs and HDVs.

For LNG, see the discussion pertaining to LH_2.

[e] An advanced Na/S battery should last for about 1,000 deep cycles, to 80% DOD (DeLuchi, 1989; Hamilton, 1988). Assuming 104 miles per deep cycle (80% of a maximum range of 130 miles, as in Table 2), the battery lifetime is 104,000 miles.

[f] There are several reasons to assume that hydrides will last for several LDV lifetimes. With very pure hydrogen, hydrides can last for hundreds of thousands of miles (DeLuchi, 1989). Hydrides are very expensive — much more expensive than CNG, LPG, or LNG containers — and hence definitely will be worth recycling. Finally, hydrides do not have to be safety-tested as CNG containers do; hence, their reuse probably will not be as subject to potentially restrictive regulation.

In the base case, I assume that hydrides last for 2,500 cycles to 80% of capacity, resulting in 300,000 miles for LDVs and 600,000 miles for HDVs (based on 80% of the ranges of Table 2). I consider a 5,000-cycle life in a scenario analysis.

TABLE P.2 (Cont.)

Little is known about the lifetime of LH_2 dewars or about the feasibility and cost of transferring a container from one vehicle to another. In the base case, I simply assume that the container dies with the vehicle.

[g] I assume that an LPG tank will last as long as the vehicle, but that typically it is not worthwhile to salvage an LPG tank, which means that the tank lasts only as long as the vehicle.

vehicles will definitely have a longer life than comparable ICEVs because electric motors last much longer than internal combustion engines (DeLuchi et al., 1989). EVs can be in service at least twice as long as comparable ICEVs, but I have assumed conservatively only a 42% longer life in the base case. In a scenario analysis (Table 12), I consider a longer life (this reduces g/mi emissions from vehicle assembly). It is also possible that dedicated natural gas vehicles (NGVs) will have a somewhat longer life than gasoline vehicles; I also consider this in a scenario analysis (Table 12).

In the base case, I assume that all HDVs have the same life. However, the lifetime of dedicated, advanced alternative-fuel heavy-duty vehicles is not known, and alternative fuel, spark-ignited or spark-assisted HDVs may not last as long as diesel fuel HDVs. I examine the effect of a shorter (250,000-mi) life in a scenario analysis.

P.1.8 Summary

Using the preceding data, the model calculates grams of greenhouse gases emitted from vehicle assembly per mile of travel by the vehicle. The results are shown in Table 9.

In the model, the mix of fuels input to electricity plants supplying auto manufacturing facilities is a variable. Changing the mix from the baseline to more or less coal and nuclear (within plausible bounds) changed the g/mi emissions from vehicle manufacture by no more than 7% and did not at all affect the overall standing of the alternatives relative to one another.

Note that I assume that assembly energy is solely a function of the weight of the vehicle, not of the kinds of materials being assembled. Thus, I assume that it takes as much energy to assemble a 2,500-lb EV as a 2,500-lb LPG vehicle also implicitly assume that it takes as much energy to assemble 1 lb of material for an alternative-energy storage medium (e.g., battery, hydride, or compressed natural gas tank) as it does to assemble 1 lb of vehicle. These simplifications are undoubtedly quantitatively, but hopefully not qualitatively, incorrect.

P.2 Greenhouse Gas Emissions from the Manufacture of Materials

Energy is used to make and assemble the materials used in vehicles, trains, ships, pipelines, power plants, vehicles, etc. The manufacture of some kinds of materials, such as aluminum, is quite energy intensive and produces a lot of greenhouse gases. Since many materials are used at every stage of every fuel cycle, and different fuel cycles use different materials, a full accounting of the greenhouse effect of alternative fuels must consider the emissions from making materials. Below, I first calculate the energy intensiveness of materials generally, then greenhouse gas emissions from the manufacture of materials for vehicles specifically. I then consider the use of materials at other points in the fuel cycle.

P.2.1 Amount and Kind of Energy Used to Make Materials

Greenhouse gas emissions from the mining, processing, and fabricating of materials (e.g., steel, iron, aluminum, concrete, etc.) are equal to the amount and kind of energy used to mine, process, and fabricate materials, multiplied by a greenhouse gas emission factor for each kind of energy used. The greenhouse gas emission factors are described and estimated in previous appendixes. This section describes the data and equations used to estimate the amount and kind of energy required to make materials, drawing on a report by Hudson (1982). The results are compared to estimates by Berry and Fels (1972) and Argonne National Laboratory (Aarons and Wolsky, 1978; Darby and Aarons, 1979; Sapp, 1981) in Table P.3.

Hudson (1982) analyzed in detail the energy used to make several materials. He compiled and reviewed estimates of the energy required to mine, refine, and fabricate ores and materials, accounting for recycling, and from these data, he calculated the energy (Btu) required to produce 1 lb of finished materials. He also disaggregated this final energy requirement by type of fuel. His estimates of each type of energy (Btu) per pound of material can be multiplied by the weight of material in a vehicle to provide estimates of the amount of each kind of fuel embodied in the materials of a vehicle, and these estimates, in turn, can be multiplied by the greenhouse-gas emission factors for each kind of fuel.

Hudson was interested in the energy use in the United States, and so did not count energy consumed in foreign countries to make products imported by the United States. From the perspective of the global greenhouse, however, it makes little difference if the carbon dioxide (CO_2) from the energy used to make a product is emitted in Europe or North America; what matters is how much greenhouse gas is emitted from a process, anywhere. Hence, I have adjusted his estimates by setting his "made in the U. S." coefficients to 1.0 in every case. Hudson's formula is:

$$E_T = (A_1 \times a_1 + A_2 \times a_2 + B \times b)V \times Z + D \times W \times Z \times O + C \times V \times Z \times o + C \times W \times Z \times o$$

TABLE P.3 Btu per Pound Intensiveness of Auto Assembly and Materials Manufacture

Process or Material	Reister (1977)	Berry & Fels (1972)	Hudson (1982)[a]	ANL (1978-1983)
Cold-rolled steel		26,366	20,264	15,000-21,000[b]
Cold-rolled alloy steel		27,680		
Cold-rolled stainless steel		39,408	30,816	
Cast iron		12,500	9,774	
Cast steel		23,300		
Automotive sheet		25,368		
Rolled/drawn aluminum	174,050	125,289	103,800	92,000-130,000[c]
Cast aluminum	173,050	107,086	9,500[d]	
Forged aluminum		125,127		
Nickel			64,101	
Titanium			185,677	
Plastics/composites	12,2450		64,598	
Rubber			55,612	
Glass			9,800	
Rolled copper	110,900	63,975	64,022	
Cast copper		62,243		
Rolled zinc		39,622	47,860[e]	
Cast zinc		43,853	47,860[e]	
Sound deadener			7,000	
Battery lead			11,678	
Sodium			46,000	
Sulfur			443	
Concrete[f]	1,165-2,205			543-1,936
Auto assembly		8,994	6,660	
Auto shipping		217		
Materials transport		630		

Btu/lb for materials manufacture are based on Btu-thermal, except as noted; that is, energy used to generate electricity has been taken into account (although this is not entirely clear in the case of Hudson's data; see footnote a). Berry and Fels assume that electricity generation is 32.8% efficient; Hudson appears to assume 30% for electricity generation and distribution.

[a] Projections for the year 1980, "Scenario I." I show the 1980 scenario here (rather than the year 2000 scenario results I use in the actual base-case analysis) to allow more direct comparison with the other estimates, which are not projections.

As far as I can tell, Hudson counts electricity as Btu-thermal. In a footnote to one of his tables (one which reviews estimates for steel), Hudson says "as far as can be determined, all data [are] either stated in or converted to a fossil fuel Btu basis" (p. 9). Although Hudson does not repeat this note for any of the other materials, I assume it applies to all materials.

[b] From Darby and Arons (1979), counting direct energy consumption from ore mines to steel mills and fabrication of semifinished products, for several kinds of products. They appear to count electricity at 3,412 Btu/kWh (energy from electricity is referred to as "electrical energy," which implies Btu-electric).

TABLE P.3 (Cont.)

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[c] From Arons and Wolsky (1978). The 130,000 figure is Btu of fuel/lb of fabricated aluminum product, from a 1976 report. The 92,000 figure is total energy use (counting all electricity, even hydro, as Btu-thermal) in the entire industry — mining to fabrication — in 1977, divided by the weight of aluminum shipments (imports excluded) in 1977. The difference may result from aluminum recycling, which is very energy efficient and is included in the lower figure (the one based on national energy and product totals) but not in the higher one. Note that Hudson's estimate accounts for recycling.

According to Arons and Wolsky (1978) and Shen (1981), the production of primary aluminum (the steps after mining and before fabrication) requires about 85,000 Btu/lb of aluminum.

[d] Assumes 100% made from scrap.

[e] Type of zinc manufacture not specified.

[f] In the manufacture of concrete, there are four sources of CO_2: (1) the use of energy to make cement, (2) the use of energy to transport cement and make bags for cement, (3) the use of energy to process and transport gravel, and (4) the conversion of $CaCO_3$ to CaO in the manufacture of cement. In this table, Reister (1977) and ANL (Sapp, 1981) estimate the total amount of energy (sources 1-3) required to produce a pound of ready-mix concrete (low end of the range) or finished concrete product (high end of the range). Most concrete for building is used as ready mix. In the following paragraphs, I examine sources 1-4 in some detail, relying partly on Sapp's estimates.

1. *Energy required to make cement.* Sapp's (1981) detailed analysis shows that the manufacture of Portland cement required about 3,700 Btu/lb in 1976 (based on Btu-thermal for electricity use). The Portland Cement Association (1986, 1989) reports about 3,650 Btu/lb for cement manufacture in 1976, 3,030 Btu/lb in 1985, and 2,950 Btu/lb in 1988. (I have converted their electricity-consumption figures to thermal Btu.)
2. *Energy required to make bags and transport cement.* According to Sapp (1981), this requires about 100 Btu/lb-cement. Thus, a reasonable value for cement manufacture and transport is about 3,000 Btu/lb (Btu-thermal). To convert this estimate of Btu/lb-cement to Btu-due-to-cement/lb-concrete, one must know how much cement is in concrete. Data in Sapp (1 metric ton of cement makes 4 cubic yards of concrete), Meridian (1989; density of concrete is 127 lb/ft^3), and EPA (AP-42; density of concrete is 148 lb/ft^3) indicate that Portland cement is about 15% the weight of concrete. Therefore, concrete contributes about 450 Btu of embodied energy/lb-concrete.
3. *Energy required to make, process, and transport noncement ingredients in concrete.* Sapp's (1981) analysis indicates that cement accounts for most of the energy embodied in ready-mix concrete. This seems reasonable, since gathering and transporting the gravel should not require much energy. I assume 50 Btu/lb-concrete for noncement components.

The total of items 1-3 is 500 Btu/lb for ready-mix concrete.

TABLE P.3 (Cont.)

4. *CO_2 emissions from $CaCO_3$ to CaO.* Cement manufacture also produces CO_2 via the transformation of limestone ($CaCO_3$) to lime (CaCO):

   $CaCO_3 \text{-->} CaO + CO_2$.

   Given that cement is about 63.5% CaO by weight, one weight unit of cement produces 0.5 weight unit of CO_2 (after Marland et al., 1989). If concrete is 15% cement (see above) by weight, then making 1 lb-concrete results in 0.075 lb-CO_2 from this source, which is in addition to the emissions above from the use of energy.

   One can express the CO_2 release from $CaCO_3$ conversion in terms of the amount of energy use that would produce the same CO_2 emission. The use of an additional 400 Btu/lb in cement manufacture would result in 0.075 lb-CO_2/lb-concrete. Hence, total "effective" energy use in the production of ready-mix concrete could be expressed as 500 Btu/lb (direct use) plus 400 Btu/lb (energy-use equivalent of emissions from $CaCO_3$ to CaO). The grand total for ready-mix concrete is thus 900 Btu/lb. To account for manufacture and use of concrete products, I use 1,000 Btu/lb for concrete in general.

where:

E_T = total energy in Btu/lb, in 2000;

A_1 = mining or feedstock energy in Btu/lb, in 1975;

a_1 = fraction of ore or material that is mined or extracted in the United States;

A_2 = processing energy for other ores in Btu/lb, in 1975;

a_2 = fraction of other ore that is processed in the United States;

B = refining energy in Btu/lb, in 1975;

b = refining efficiency factor (year 2000 Btu/lb divided by year 1975 Btu/lb);

V = fraction of semifabricated material that originates from virgin material, in 2000;

Z = fraction of semifabricated material that is produced in the United States;

D = scrap processing energy in Btu/lb, in 1975;

O = scrap processing efficiency factor (year 2000 Btu/lb divided by year 1975 Btu/lb);

W = fraction of semifabricated material that originates from recycled material (1 – V);

C = semifabrication energy in Btu/lb, in 1975; and

o = semifabrication efficiency factor (year 2000 Btu/lb divided by year 1975 Btu/lb).

I have recalculated E_T for all materials for which Hudson had data by using Hudson's equation and his values for all the variables except a_1, a_2, and Z, which I have set equal to 1.0. I use the average of his "scenario-I" and "scenario-II" year 2000 values. In most cases, this results in a 15% reduction in Btu/lb energy use in 2000 compared with 1975. For the materials for which Hudson did not have enough information to specify all of the variables for his equation (glass, plastics, and sound deadeners), I use the final, undisaggregated estimates for 2000 (the average of scenarios I and II). I have adjusted Hudson's estimate for plastic to remove his assumption that 22% of the feedstock petroleum would have been imported in 1980 and, hence, for his purposes, need not have been included.

Table P.3 shows the results of the calculation. Hudson's results are reasonably consistent with those of two other studies, as shown in Table P.3.

The estimates in Table P.3 count electricity as Btu-thermal; that is, they count fossil-fuel input energy to electricity, not electrical output energy. I have left the estimates in this form because it is not possible to convert all of them to a Btu-electric basis. In Table P.4, I have converted Hudson's estimates to a Btu-electric basis.

I broke down Hudson's Btu/lb totals into types of fuel (coal, oil, NG, and electricity) by using data in Hudson (the results are shown in Table P.4). Hudson provided estimates of how much of the total energy used to make materials comes from coal, NG, petroleum, electricity, and "other," counting electricity as Btu-thermal. I converted his Btu-thermal to Btu-electric by multiplying by 30% (the value he apparently used for the efficiency of electricity generation), and I counted "other" energy as electricity. In most cases the "other" category was zero or very small; in the one case where it was large (aluminum manufacture), Hudson noted specifically that "other" referred to hydroelectric power.

TABLE P.4 Amount and Kind of Energy Required to Make a Pound of Material, and Fractions of Materials in LDGVs

Material[c]	Energy in 2000 (Btu/lb)[a]	Energy Share[a]				Fraction Material in LDGV[b]
		Coal	Oil	NG	Power	
Plain carbon steel	13,315	0.59	0.06	0.23	0.13	0.451
High strength steel	20,876	0.59	0.06	0.23	0.13	0.075
Stainless steel	22,220	0.63	0.06	0.20	0.11	0.010
Other steels	13,315	0.59	0.06	0.23	0.13	0.015
Iron	8,445	0.65	0.06	0.25	0.04	0.146
Plastics/composites	61,433	0.00	0.28	0.70	0.02	0.071
Fluids and lubricants	0					0.057
Rubber	38,307	0.20	0.30	0.41	0.10	0.043
Aluminum	44,352	0.04	0.05	0.60	0.31	0.050
Titanium	60,498	0.03	0.18	0.10	0.69	0.000
Glass	8,408	0.02	0.18	0.75	0.05	0.027
Copper	46,303	0.56	0.19	0.13	0.11	0.016
Zinc die castings	32,743	0.35	.00	0.54	0.10	0.006
Powdered metal components	3,926	0.03	0.38	0.29	0.29	0.007
Other materials (lead)	6,273	0.37	0.03	0.30	0.30	0.026
Sodium	15,658	0.26	0.01	0.00	0.73	0.000
Sulfur[d]	443	0.60	0.00	0.30	0.10	0.000

[a] From Hudson (1982) year 2000 projections, as detailed in the text. Shares here are based on Btu-electric. See also Reister (1977).

[b] From *Ward's 1989 Automotive Yearbook.*

[c] The materials categories in *Ward's Yearbook* and in Hudson did not match exactly. I used Hudson's cold-rolled steel factors for Ward's "plain carbon steel," "high-strength steel," and "other steels"; his cast-iron factor for Ward's "iron"; his tire factors for Ward's "rubber"; his zinc factor for Ward's "zinc die castings"; his sound deadener factors for Ward's "powdered metal components"; and his lead factor for Ward's "other materials." I ignored "fluids and lubricants" in Ward's breakdown. (Hudson did not have data). Most of these substitutions and assumptions probably are reasonable. I chose sound deadeners and lead for the last two Ward's categories because data in Millar et al. (1982) indicate that of the materials not mentioned explicitly in *Ward's Yearbook*, these are the heaviest.

[d] Breakdown for sulfur is my assumption.

P.2.2 Greenhouse Gas Emissions from the Manufacture of Materials Used in Motor Vehicles

Grams of CO_2-equivalent emissions of greenhouse gases from the manufacture of the materials used in a vehicle are calculated as the sum of emissions from the manufacture of the materials in the vehicle, excluding the fuel-storage equipment, plus emissions from the manufacture of materials in the fuel-storage equipment. I distinguish between the vehicle and its storage equipment because all ICEVs, storage equipment aside, are comprised of essentially the same materials, regardless of the fuel; whereas the weight and composition of fuel-storage systems is variable (a battery is very different from a hydride or a gasoline tank). Ideally, one would distinguish between EVs and ICEVs, but the data to do this are not available.

Formally, the calculation is:

$$T_g = \text{SUM [for all f, m]}\ E_{f,m} \times W_m \times G_f/\text{VMT}$$

where:

T_g = total greenhouse gas emissions from the manufacture of materials for one vehicle per mile of travel or for one piece of storage equipment per mile of its life;

$E_{f,m}$ = total Btu of fuel, f, required from mining to fabrication to produce 1 lb of material, m;

W_m = weight of material, m, in the vehicle or the fuel storage equipment;

G_f = grams of CO_2-equivalent emissions of greenhouse gases from the recovery, refining, transport, and combustion of 1 Btu of fuel, f; and

VMT = lifetime mileage of the vehicle or its storage equipment (Table 2).

The amount and kind of energy required to make materials, $E_{f,m}$, was discussed in the preceding section, and the greenhouse emission factor for each fuel, G_f, is described generally in the main text of this report and in detail in the various appendixes. All that is needed, then, is a calculation of W_m, the weight of each material in each kind of vehicle and fuel storage system.

For all vehicles or storage tanks, the weight of each material, m, is calculated as the product of the total curb weight of the vehicle (less the weight of the fuel tank) or the total fuel storage system weight, and the fraction of the total weight that is m. Therefore, we need to know four things: the weight of the vehicle, the weight of the fuel storage system, the composition of the vehicle, and the composition of the fuel storage system.

Table 2 shows the weight of the vehicles and fuel storage systems considered in this analysis. For the composition of vehicles, I use data in *Ward's 1989 Automotive Yearbook*, which gives a detailed breakdown of the weight of materials in a typical light-duty car. I assume that this distribution of materials in LDVs is the same for all vehicles: LDVs and HDVs; all alternative-fuel vehicles, including EVs; and petroleum-fueled vehicles. (Obviously, this assumption is not accurate in the case of EVs; moreover, it may be important because it takes much more energy to make a pound of copper, aluminum, or plastic than a pound of steel. Nevertheless, I calculate emissions from manufacture of materials, assuming the same composition as ICEVs.) This breakdown is shown in Table P.4. The article in *Ward's* notes that, since 1977, the amount of plain carbon steel in the typical U. S. car has decreased; the amount of high-strength steel, plastics, and aluminum has increased, and that this trend is expected to continue. However, I have used the Ward's data without adjustment.

The composition of fuel storage systems and batteries can be estimated from data in the literature. Table P.5 documents my estimates.

I also assume, implicitly, that the energy required to assemble a pound of battery is the same as the energy required to assemble a pound of vehicle. This, too, may not be accurate.

P.2.3 Greenhouse Gas Emissions from the Manufacture of Materials Used to Make Power Plants, Uranium Mines, Pipelines, Trains, Ships, etc.

Greenhouse gases are emitted from the consumption of fossil-fuels used to make and assemble materials for the equipment and facilities associated with alternative fuels: power plants, refineries, methanol production plants, pipelines, oil wells, coal mines, tankers, transmission towers, automobile assembly plants, etc. If the use of alternative fuels will require the construction of new facilities and equipment, emissions from the materials and assembly of these facilities and equipment should be calculated and divided by the amount of vehicle mileage associated with the output or throughput of the facility or equipment. However, my calculations and literature review indicate that, with a few exceptions, the amount of energy used to make and assemble the materials of equipment and facilities is very small compared to energy output or throughput.

Table P.6 shows the amount of energy embodied in the major materials used in several kinds of energy facilities, and compares this with the output or throughput by those facilities. In almost every case, the energy embodied in the materials is less than 1% (sometimes much less) of the energy produced or carried by the system.

Table P.6 does not account for energy used at the construction site. However, available evidence indicates that this is small compared to the energy embodied in the materials (e.g., compare the energy required to assemble vehicles with the energy embodied in the materials in vehicles). Mortimer (1989) has estimated that the use of fuels during the construction of a coal-fired power plant accounts for only 0.1% of total life-cycle CO_2 emissions from the power plant

TABLE P.5 Breakdown of Weight in Fuel Storage Tanks

Material	CNG[a]	LNG[b]	EVs[c]	Hydride[d]	LH_2[e]	LPG[f]
Carbon steel	0.000	0.400	0.000	0.000	0.000	0.900
Stainless steel	0.000	0.550	0.600	0.066	0.300	0.047
Other steels	0.050	0.050	0.050	0.034	0.050	0.000
Iron	0.000	0.000	0.020	0.450	0.000	0.010
Plastic/composites	0.500	0.000	0.020	0.000	0.250	0.000
Aluminum	0.450	0.000	0.050	0.450	0.350	0.033
Titanium	0.000	0.000	0.000	0.450	0.000	0.000
Glass	0.000	0.000	0.047	0.000	0.000	0.000
Copper	0.000	0.000	0.033	0.000	0.050	0.010
Sodium	0.000	0.000	0.060	0.000	0.000	0.000
Sulfur	0.000	0.000	0.120	0.000	0.000	0.000

[a] The composition of a CNG container depends on cost, pressure, weight, and bulk considerations. Carbon-wrapped, aluminum-lined containers are among the strongest per unit weight but also are very expensive, kevlar wrapping is less expensive and less strong, and fiberglass wrapping is least expensive and least strong. Most CNGVs to date have used either all steel or fiberglass-wrapped aluminum cylinders. I assume that future CNGVs will use fiberglass-wrapped, aluminum-lined cylinders.

Morris, Segimoto, and Lynn (1986) report that a carbon-wrapped, aluminum-lined pressure vessel, 50 inches long by 10 inches in diameter, is 52% aluminum, 42% carbon/epoxy composite, and 6% insulation. They also show that a kevlar-wrapped aluminum container that is not as strong is 63% aluminum and 37% kevlar. According to Morris et al., a fiberglass/resin matrix weighs 2 to 3 times more than a carbon/resin matrix and weighs 1.5 to 2 times more than a kevlar/resin matrix, at constant volume and pressure. This indicates that a fiberglass-wrapped, aluminum-lined vessel for automobiles would be at least 50% composite material and less than 50% aluminum. There also would be some additional in weight in mounting brackets and other hardware, which I assume would be made of steel. With these considerations, I assume that about half of the total vessel-plus-hardware weight is composite material, 5% is miscellaneous steel, and the rest aluminum.

[b] Constable et al. (1989) state that LNG containers consist of an inner vessel of stainless steel or aluminum alloys, an outer casing of low-carbon steel, and stainless-steel piping and valving. Adkins (1989) gives a similar description. From this description, I assume a vessel consisting primarily of stainless steel and low-carbon steel, with a small amount of other steel for tank support and mounting.

TABLE P.5 (Cont.)

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[c] This is the estimated weight breakdown for the Na/S battery only; it does not account for the differences between the materials in an ICEV powertrain and an EV powertrain.

Auxer (1991) estimates, roughly, that a Na/S battery is about 12% sulfur, 6% sodium, 20% ceramics, and the rest steels, wiring, insulation, and other materials. The steel battery housing accounts for a large portion of the total weight. He notes that Na/S batteries contain many kinds of materials.

I use Auxer's estimates for sodium and sulfur and assume that most of the rest of the weight is stainless steel in the housing and other components. I have arbitrarily distributed a small amount of weight among several other materials.

See also Haase (1987), Nelson (1989), and Quinn et al. (1989).

[d] The weight breakdown composition depends on the composition of the hydride. See Stewart (1986) and DeLuchi (1989) for discussions and examples of hydrides for automobiles. Some hydrides may be fairly energy intensive. For example, iron-titanium hydrides, which are quite common, may be about half iron and half titanium by weight (as was the hydride for a Post Office Jeep converted to hydrogen, according to Stewart [1986]), and titanium is very energy intensive. I assume an iron-titanium hydride here.

[e] LH_2 dewars have been made of stainless steel, aluminum, aluminum alloys, and combinations of these (Peschka, 1987; Stewart, 1986). Peschka (1987) expects that future LH_2 tanks will be made of advanced, lightweight, composite materials. The inner vessel and the outer vessel may not be the same material. Dewars also may have copper radiation shields, and insulation.

Given the wide range of possible materials for future LH_2 vessels, the possibility of having different materials for the inner and outer vessels, and the use of yet different materials for things like radiation shields, I specify a generic dewar that contains stainless steel, aluminum, and composites. The weight share of composites is slightly less, because composites are lighter per unit of strength.

[f] Webb (1990) reports that LPG tanks generally are made of steel, but that fiber-reinforced plastics are being used experimentally in Europe. Bechtold and Timbario (1983) state that LPG tanks are manufactured from low-carbon steel. The Western Liquid Gas Association (1988) cites a company that makes aluminum LPG tanks. A 70-lb steel tank holds about 12.5 gallons of propane; a fiber-reinforced plastic tank weighs half as much (Webb, 1990). I assume low-carbon-steel tanks.

TABLE P.6 Energy Embodied in the Major Materials of Large Energy Facilities and Other Equipment

Facility and Major Materials[a]	Energy in Materials Divided by Energy Produced or Carried over Life[b]
Open-pit uranium mine: concrete, steel, iron	0.00009
Underground uranium mine: concrete, steel, iron	0.0003
Nuclear-fuel conversion plant: concrete, steel	0.00008[c]
Gaseous diffusion plant: concrete, steel, iron	0.0009[c]
Boiling water nuclear power plant:	0.002[d]
Western unit coal train: steel, aluminum, copper	0.0005
Coal truck: steel	0.0002
Coal power plant, western coal: concrete, steel, copper	0.005[e]
High-voltage power transmission: concrete, steel, al., copper	0.008
Onshore oil well: concrete, steel	0.0007
Enhanced oil recovery: concrete, steel	0.003
161,000 dwt oil tanker: steel[f]	0.0014
Onshore gas extraction: concrete, steel	0.008
Natural gas pipeline: steel	0.004
Medium-Btu biogas facility: concrete, steel	0.008
Large hydroelectric plant: concrete, steel, iron, al., copper	0.005
Solar-photovoltaic plant: concrete, steel, glass[g], silicon	0.015
Biofuels:	
Corn-to-ethanol plant: presumably concrete, steel	0.01-0.06[h]
Farm equipment, corn-to-ethanol: steel	~0.10[i]
SRIC equipment, wood-to-fuel: steel	0.01-0.04[j]

[a] Data on materials from the *DOE Handbook* (1983), except as noted. The handbook has quantitative estimates. In virtually every case, steel accounts for most of the embodied energy.

[b] Except as noted, energy in materials calculated as the amount of materials, as given in the *Handbook*, multiplied by the thermal energy embodied per unit of material (25,000 Btu/lb for all kinds of steel, 1,000 Btu/lb of concrete [which accounts for CO_2 released from the transformation of $CaCO_3$ to CaO; see note f, Table P.3], 125,000 Btu/lb of aluminum, 10,000 Btu/lb of iron, 50,000 Btu/lb of copper and silicon; these count electricity as Btu-thermal at roughly 10,500 Btu/kWh, for simplicity). Lifetime energy output or throughput is equal to yearly output or throughput of facility, as given in the *Handbook*, multiplied by the plant lifetime in years, as given in the *Handbook* (generally 20 to 30 years, except for the hydro plant, which is 60 years). I assume 30 years for the high-voltage power line.

TABLE P.6 (Cont.)

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[c] The sum of these two, 0.001, compares well with the result of a calculation based on British data. From 1971 to 1978, the construction of the uranium-processing buildings of British Nuclear Fuels Limited (BNFL) required 2.2 million tons of concrete (Ferguson, 1981; I have recalculated the weight given in the original). The manufacture of this concrete required about 4.4 trillion Btu (direct energy and "effective" energy; see Table P.3, note f). It appears that a similar amount of energy was embodied in the steel. Over the same 8-year period, BNFL processed about 14,000 metric tons of natural uranium, which should have produced about 1.3 quads of electrical energy (assuming 0.0395 ton uranium/gWh generated, from Table I.3). Assuming that the BNFL plants will produce for a total of 30 years, total effective output will be around 4.9 quads-electric, and the material-energy/output-energy ratio will be about 0.002.

[d] The estimate of Rotty et al. (1975), which includes energy used by diesel generators, is an order of magnitude higher. See Table I.7. The output here is electricity, in Btu-electric.

[e] This result compares well with the result of a calculation based on British data. The Central Electricity Generating Board in England estimates that a 1,000-MW coal-fired power plant requires 500,000 tons of concrete and 18,750 tons of steel (Ferguson, 1981). With the Btu/lb figures of note b above, a 30-year life, and 70% capacity figure, the energy embodied in the materials is 0.003 of power output.

The output here is taken to be Btu-electric. For a coal-fired plant, it may be more accurate to divide the Btu embodied in the materials by the Btu of fuel input to the power plant rather than by the Btu-electric output; doing this would reduce the ratio by a factor of three and reinforce the conclusion that emissions from materials manufacture are small.

[f] Data on steel content from Marks (1982). He shows a calculation in which a 161,000-dwt tanker weighs 28,171 tons (virtually all steel) and completes 8.49 round trips per year. Elsewhere, he assumes that ships lasts 20 years. With these data, and assuming that 96% of the dwt is oil, one can calculate that the ship carries 26.2 million tons of crude — about 1 quad — over its life.

[g] The DOE *Handbook* does not give the amount of glass, so I have not included glass in this calculation.

[h] From Table K.7. I have dropped the estimate by Parisi (1983) because, as explained in the notes to K.7, it appears to be unreliable.

[i] From Table K.7. Relative to ethanol produced by the whole system.

[j] From Table K.9, converted to per-Btu-fuel-produced basis.

(this value apparently refers to direct use of fuels during construction; not to fuels used to make materials). He also estimated that emissions during the construction of a nuclear power plant are three times larger, but the result (0.3%) is still insignificant.

Several other estimates confirm that the manufacture of major materials used in major facilities, such as power plants, adds very little to the life-cycle CO_2-equivalent emissions from highway vehicles. Meridian Corporation (1989) estimates that the final manufacture of concrete and steel used to build coal or nuclear power plants produces about 300 g of $CO_2/10^6$ Btu of power output; to build photovoltaic power plants, about 1,600. These are about 0.1% and 0.6% of the g of $CO_2/10^6$ Btu-power emissions from the operation of coal-fired plants, percentages that are of the same order of magnitude as the estimates in Table P.6. Meridian apparently does not count CO_2 emissions from mining and refining the material and definitely does not count CO_2 from the actual construction of the plants.

Rotty et al. (1975) estimate that $2{,}311 \times 10^9$ Btu of coal and 447×10^9 Btu of electricity (at 3,412 Btu/kWh) are consumed to make a 1,000-MW nuclear reactor. Assuming the national average power mix, this construction energy use would produce about 370×10^9 g of CO_2-equivalent emissions (including emissions from mining and transporting the coal and the electricity fuels and emissions from non-CO_2 greenhouse gases) or about 540 g of CO_2-equivalent emissions per 10^6 Btu of power output. This figure is quite consistent with Meridian's estimate when one remembers that Meridian considers only CO_2 and only the final manufacture of materials.

White (1980, p. 3) remarks that, in most cases, emissions from the manufacture of materials used for construction are "quite small compared with the major conversion and product combustion emissions of CO_2."

Biofuel cycles may be an exception to this rule. The data in Table P.6 (based on the data of Tables K.7 and K.9) show the energy embodied in the equipment and facilities used in the biomass cycle exceeds the energy embodied in the coal, oil, gas, and uranium cycles by one order of magnitude. This is not implausible because biomass resources are considerably less energy dense and less concentrated than fossil fuels, and they require much more intensive use of collection, transport, and processing equipment and facilities. If accurate, this result is important. For example, if the energy invested in all equipment in the corn-to-ethanol cycle is assumed to be 10% of the energy in the output ethanol, then grams per mile emissions for the ethanol cycle increase by 5-9%, depending on the source of the energy used to make the materials in the equipment. This is a nontrivial amount, and should be included in a fair comparison. I have included it in a scenario analysis here (Table 12).

I conclude from this that, with the possible exception of biofuel-production facilities, large energy-producing facilities embody relatively little energy, compared to what they produce. In this report, I have ignored emissions associated with the building of major facilities, such as power plants, petroleum refineries, and nuclear-fuel fabrication plants. It seems likely that consideration of such items will not change the rankings among the alternatives (with the possible exception of biofuels). I have included indirect energy consumption by trains, pipeline, trucks, and ships, where indirect includes energy used for maintenance, construction, manufacturing, and other activities (see Appendix E).

Note, however, that the use of alternative fuels will not automatically require immediate construction of new facilities. The timing and scale of the construction of new facilities will depend on the demand for a new fuel relative to existing capacity to make and transport the fuel. In some cases, existing capacity will be used up relatively quickly; in other cases not nearly so quickly. For example, in the case of methanol, all existing plants will be operating at capacity at relatively low levels of demand for methanol. Therefore, to support a large-scale methanol program, new plants will have to built, and the very minor greenhouse emissions attributable to building these facilities will be assigned to methanol. On the other hand, the use of EVs is not likely to require new generating and transmission capacity for many years because until the EV program becomes very widespread, most EVs will be recharged at night by using otherwise idle capacity. Consequently, if a more detailed analysis shows, for certain fuel cycles, a significant amount of energy use and emissions generation in the manufacture and assembly of the materials of energy facilities, one must then determine the rate at which new facilities will be required.

P.3 Greenhouse Gases from Making Chemicals and Other Materials

For two reasons, this report does not estimate greenhouse gas emissions from the manufacture of chemicals used in the various fuel cycles. First, it is difficult to get data on all the chemicals used in all the fuel cycles analyzed here and difficult to get information on the amount and kind of energy required to make all the chemicals. Second, estimates in the literature indicate that the total energy required to make chemicals and other non-structural materials is, in most cases, negligible. Rotty et al. (1975) found that, generally, direct energy use at the several stages of the nuclear fuel cycle dominates indirect use (chemicals, materials, and construction).

Similarly, Sapp (1981) found that direct energy use in the cement manufacturing industry is 94% of total energy use. Since the total energy used to make the cement that is used to build the facilities that generate power or fuel is small compared to the output of those energy facilities, the energy used to make chemicals is tiny compared to energy output from energy facilities. And finally, Lynd et al. (1991) estimate that the energy required to make the chemicals used in the wood-to-ethanol process is 1% of the energy in the ethanol produced.

As an illustration, suppose that the energy used to make chemicals bought by petroleum refineries is as much as 10% of the total process energy used by petroleum refineries (a figure which seems high by severalfold). Even so, this energy for the chemicals would be no more than 1% of the energy output of petroleum refineries (because total process energy consumed is about 10% of output). Therefore, I tentatively conclude that consideration of emissions of greenhouse gases from the manufacture of chemicals used in the fuel cycles would change the absolute results by less than 1%, and not change the relative results at all.

Again, the biofuel cycles may be exceptions. As noted in Table K.7, the energy required to make the ammonia used by the Katzen (1979) corn-to-ethanol design amounts to 0.02-0.04 Btu/Btu-ethanol, and the process uses other chemicals as well.

P-26

Appendix Q:

Chlorofluorocarbons, Ozone, and Water Vapor

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Q.1 Chlorofluorocarbons

Q.1.1 Background

The various chlorofluorocarbon (CFC) compounds are among the most potent of the greenhouse gases. A CFC molecule has several thousand times the radiative forcing of a carbon dioxide (CO_2) molecule and can persist in the atmosphere for centuries before being photodissociated in the stratosphere (Watson et al., 1990). The Intergovernmental Panel on Climate Change (IPCC) (Watson et al., 1990) expects CFC emissions to account for 11% of the total global warming over the next century, in spite of multinational agreements limiting production of CFCs. (Absent the agreements limiting production, CFCs would be responsible for more future warming than methane.) Moreover, the chlorine released by the decomposition of CFCs destroys odd-numbered oxygen molecules (O or ozone [O_3]), thus depleting the protective blanket of stratospheric ozone that absorbs ultraviolet radiation (Brasseur, 1987). CFC emissions are therefore harmful both as greenhouse gases, directly and indirectly, and as an ultimate cause of increased exposure to high-energy solar radiation.

All CFCs come from industrial processes; there are no natural sources. Over the past few decades, the concentration of CFCs has grown rapidly and now is increasing at least 4% per year (Watson et al., 1990). The Montreal Protocol will reduce the rate of increase, but CFCs will remain in the atmosphere for many years because of their long lifetimes.

Q.1.2 CFCs from Vehicles

The only significant source of CFCs from highway vehicles is the air-conditioning system. The CFCs leak from this system during regular operation and are released entirely when the coolant is changed or the auto is scrapped. This contribution to the greenhouse effect of CFCs from automobiles is independent of the type of fuel used. Strategies for reducing CFC releases, such as recovering cooling equipment from scrap yards, are also independent of type of the fuel used.

The refrigerant used in automobile air-conditioning systems is Freon (CF_2Cl_2), or CFC-12. The concentration of CFC-12 has increased from a preindustrial level of zero to about 340 parts per trillion volume (pptv) in the early 1980s and has been increasing at about 6%/yr (WMO, 1985). The greenhouse model here calculates grams per mile of CFC-12 given the amount of refrigerant consumed per light-duty vehicle (LDV) or heavy-duty vehicle (HDV) over its life and the lifetime mileage of the vehicle. I assume 3,400 g released (based on information from Shiller (1989); see Table Q.1, footnote e) and 108,000 miles, resulting in 31.5 mg/mi. This