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On the Costs of Air Pollution from Motor Vehicles

By Kenneth A. Small and Camilla Kazimi*

1. Introduction

Air pollution is frequently the stated reason for special measures aimed at controlling motor vehicles. In the United States, motor vehicle emission standards are set explicitly in clean air legislation, while policies at several levels of government are designed to reduce the use of cars for particular purposes like commuting. In Europe, high fuel taxes and subsidies to urban mass transit and intercity rail travel in large part aim to reduce car use.

Such measures are often justified by pointing to a gap between the private and social cost of car travel, caused by subsidies and/or externalities such as noise and air pollution. This creates a certain pressure to measure the true social cost of car travel (US Federal Highway Administration, 1982; Mayeres, 1992; Quinet, 1993; DeLucchi *et al.*, forthcoming). These efforts will not produce consensus on a single social cost figure. The costs of air pollution, noise and other environmental damage are not precisely measurable, and even the principles of measurement are not universally accepted. Nevertheless, estimates of pollution costs from motor vehicles can help shape the broad outlines of policy toward pollution control.

This paper focuses on measuring the costs of regional (tropospheric) air pollution from motor vehicles. We discuss some of the analytical and empirical issues involved in estimating such numbers, and provide some estimates for the Los Angeles region under a variety of alternative assumptions. We conclude that the measurable costs of air pollution are high enough to justify substantial expenditures to control vehicle emission rates, but cannot by themselves justify drastic changes in the highway-oriented transport system that has evolved in most of the developed world. At present, it appears that including greenhouse gases would leave this basic conclusion unchanged.

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2. The Policy Context

Transport accounts for substantial fractions of direct emissions (that is, "inventories") of three primary pollutants: volatile organic compounds (VOC), carbon monoxide (CO) and nitrogen oxides (NO_x). (See the appendix for a glossary of acronyms and chemical formulae.) VOCs, also known as reactive organic gases (ROG), give car exhaust its characteristic smell. VOCs react with NO_x in the atmosphere to form a variety of damaging oxidants such as ozone (O_3), and they also produce secondary carbon, a component of particulate matter (PM).

The main ingredient of VOC is hydrocarbons (HC), which are emitted primarily as unburned components of petroleum. Another ingredient is oxyhydrocarbon compounds, such as formaldehyde, which are formed in the combustion process (National Research Council, 1991, p. 23). The lightest hydrocarbon, methane, is less reactive than other components of VOC and is often excluded from regulation; for these reasons, data are sometimes compiled as non-methane hydrocarbons (NMHC) or non-methane organic gases (NMOG). For the most part we ignore these distinctions because the quantitative differences among VOC, ROG, HC, NMOG, and NMHC are small for petroleum-based motor vehicle emissions.

Motor vehicles, especially those using diesel fuel, emit some particulate matter directly and also emit sulphur oxides (SO_x), primarily sulphur dioxide (SO_2). SO_2 is an irritant and contributes to particulate formation and acid rain. The same is true of nitrogen dioxide (NO_2), which is formed in the atmosphere from other NO_x emissions and helps give smog its brown colour. We do not explore the direct effects of ambient SO_2 and NO_2 in this paper, but we do account for their role in particulate formation, which appears to be more important to human health.

Table 1 shows the fractions for direct emissions of several pollutants that are estimated to come from transport activities. Figures such as these are sometimes summarised by the misleading statement that half of all air pollution is from cars. Actually the fraction varies considerably by pollutant and location, and tends to be much higher in urban areas as illustrated in the table by the figures for Los Angeles.

In recent years, concern about motor vehicle emissions has broadened to include the global effects of certain "greenhouse gases," mainly carbon dioxide (CO_2), whose accumulation over decades or centuries may cause a gradual warming of the earth's atmosphere (Cline, 1991). Such warming would be accompanied by largely unknown but possibly dramatic changes in wind and rainfall patterns, and possibly by rising sea levels. We return to the question of global warming at the end of the paper.

Emission control legislation

The primary legislation for controlling vehicle emissions in the United States is the Clean Air Act of 1963, amended in 1970, 1977 and 1990. The act provides a mechanism for setting ambient air quality standards, which define the maximum acceptable concentrations of pollutants in the air. The ambient standards apply to CO, O_3, NO_2, SO_2, particulate matter of less than 10 microns diameter (PM10), and lead. The act also specifies emission

Table 1

Proportion of Emissions Inventories Accounted for by Transport Activities

Country	*CO*	*VOC*[a]	*NO_x*	*SO_x*
USA	66	48	43	N.A
Los Angeles region	98	75[b]	83	68
Europe	78	50	60	4
UK	86	32	49	2
France	71	60	76	10
Germany	74	53	65	6

[a] Inventory for Europe is for hydrocarbons (HC).
[b] Adjusted for assumption that motor vehicle emissions are 2.1 times those assumed in the source (see text)
Sources Ball *et al.* (1991); Whitelegg (1993), SCAQMD (1994).

standards from motor vehicles, including tailpipe emissions of CO, VOC and NO_x, and evaporative emissions of VOC. The state of California has even stricter standards, mainly because its topography and climate tend to concentrate emitted pollutants and to foster chemical reactions involving them.

Table 2 lists many of the past and future tailpipe standards for the United States and for California. The new federal "tier I" and "tier II" standards, as well as the four newly specified vehicle classes for California, arebeing phased in from 1994. California has its own phase-in schedule, with the intent of making motor vehicles account for only ten per cent of ozone in the Los Angeles basin by the year 2010 (Calvert *et al.*, 1993, p. 39).

In Europe, prior to 1992, most countries complied with emission standards set forth by the UN Economic Commission for Europe. Exceptions include Austria, Denmark, Finland, Sweden, Norway and Switzerland, all of which have adopted somewhat stricter standards similar to those in the United States. Since 1992, nations in the European Union have been required to follow regulations promulgated by the Commission of the European Communities. Some of these regulations are shown in Table 3. The standards are based on urban and "extra-urban" driving cycles which differ from the US federal test procedure. Overall the European restrictions are not as stringent as in the United States: for example, the 1992 Euro I regulations for heavy-duty vehicles are similar to US standards in 1988, and the Euro II regulations effective from 1995 are comparable to US standards for 1991.

On-road emissions

Actual on-road emissions are a different matter. Estimates and predictions of emissions in the US have mostly been made using either the federal MOBILE model or California's EMFAC model. These models attempt to account for fleet composition, driving behaviour, and aging of emission control devices.

Table 2

Exhaust Emission Standards for Gasoline-powered Light-duty Vehicles[a] *(grams/mile)*

Model Year	*US Federal Standard*			*California Standard*		
	CO	*VOC*[b]	*NO_x*	*CO*	*VOC*[b]	*NO_x*
Average pre-control car	84.0	10 6	4.1	84 0	10.6	4.1
1968-69	51.0	6.30	—[c]	51 0	6.30	—[c]
1970	34.0	4.10	—	34.0	4.10	—
1971	34.0	4.10	—	34.0	4.10	4.0
1972	28.0	3 00	—	34.0	2.90	3.0
1973	28.0	3.00	3 0	34.0	2.90	3 0
1974	28.0	3.00	3.0	34.0	2.90	2.0
1975-76	15.0	1.50	3.1[d]	9.0	0.90	2.0
1977-79	15.0	1.50	2.0	9.0	0.41	1.5
1980	7.0	0.41	2.0	9.0	0.41[e]	1.0
1981-82	3.4	0.41	1.0	7.0	0.41[e]	0.7
1983-92	3.4	0 41	1.0	7.0	0.41[e]	0 4
1993	3 4	0 41	1.0	3.4	0.25[f]	0.4
Tier I (Calif.TLEV)	3.4	0.25[g]	0.4	3.4	0.125[f]	0 4
Tier II (Calif:LEV)	1.7	0.125[g]	0.2	3.4	0.075[f]	0.2
ULEV				1.7	0.040[f]	0.2
ZEV				0	0[f]	0

[a] Through 1993 the standards apply to new vehicles. Later standards apply at 5 years or 50,000 miles (10 years or 100,000 miles for federal tier II). Diesel vehicles must also meet standards for particulate matter starting in 1984 (California) or 1986 (US federal).
[b] The standard is for hydrocarbons (HC) unless otherwise noted.
[c] NO_x emissions increased when left uncontrolled as HC and CO emissions were reduced.
[d] New federal test procedure.
[e] Optionally, can instead meet standard of 0.39 g/mi NMHC.
[f] The standard is for NMHC.
[g] The standard is for non-methane organic gases.

Pollutants:
CO = Carbon monoxide
VOC = Volatile organic compounds
NO_x = Nitrogen oxides
NMHC = Non-methane hydrocarbons

California Vehicles:
TLEV = Transitional low emission vehicle
LEV= Low emission vehicle
ULEV = Ultra-low emission vehicle
ZEV = Zero emission vehicle

Sources: Calvert *et al.* (1993, Tables 1-2), California Air Resources Board (1986; 1994, Table 2).

Table 3

European Emission Standards

	Effective Year	*CO*	*HC*	*NO_x*	*Particulates*	*Evaporative HC*
Passenger vehicles (g/km)						
—91/441/EEC[a]	1992	2.72	0.97[b]		0.14	2.0 g/test
—94/12/EC (proposed)						
— gasoline powered	1997	2.2	0.5[b]		N.A	N.A.
— diesel powered	1997	1.0	0.7[b]		0 08	N.A.
Heavy-duty diesels (g/kwh)						
— Euro I	1992	4.5	1.1	8 0	0.36[c]	N.A.
— Euro II	1995	4 0	1.1	7.0	0.15	N.A
— Euro III (proposed)	1999	2.5	0.7	<0.5	<0.12	N.A.

[a] Standard shown is for "type approval". Applies to both gasoline and diesel cars.

[b] A single standard applies to the sum of HC and NO_x emissions.

[c] For engines smaller than 85kW, particulate emissions may be 1.7 times higher, or 0 612 g/kwh.

N A = not available.

Source· CONCAWE (1992), and Inge Mayeres, personal communication, June 1994.

However, three lines of evidence have recently cast doubt on the accuracy of these emissions inventory models (Cadle *et al.*, 1993; St. Denis *et al.*, 1994). The first comes from measurements of ambient concentrations in tunnels, where the volume of air and its rate of flow can be precisely measured. Data from a tunnel under Van Nuys Airport near Los Angeles indicate that emissions of CO and NMHC were 2.7 and 3.8 times higher than predicted by California's EMFAC model, version 7C (Ingalls, 1989; Ingalls *et al.*, 1989). Other tunnels show similar discrepancies with the US federal MOBILE model (Pierson *et al.*, 1990). The second line of evidence is from airshed models that predict ambient concentrations from an assumed spatial distribution of emissions. A comparison using version 7E of EMFAC showed that ambient ratios of CO to NO_x and of VOC to NO_x were higher than could be predicted, by factors of 1.5 for CO and 2-2.5 for VOC (Fujita *et al.*, 1992). These discrepancies are believed to represent under-estimation of CO and VOC emissions. The third line of evidence is from data collected to calibrate newly-developed remote sensing devices that measure pollutants as they flow out of tailpipes on moving vehicles (Lawson *et al.*, 1990; Stephens and Cadle, 1991; Bishop *et al.*, 1993).

At least two reasons have been offered as to why the models underpredict CO and VOC emissions. First, they under-represent both the frequency and the severity of gross polluters. The remote sensing experiments have shown that roughly 10 per cent of the vehicles produce over 50 per cent of CO emissions, and there are strong indications that inspection programmes are ineffective in maintaining emission control systems (Glazer *et al.*, 1995). Second, the models are based upon a federal test procedure that inadequately accounts for occasional hard accelerations (St. Denis *et al.*, 1994).

As a result of this evidence, more recent versions of the models have been modified by raising the assumed emissions rates from motor vehicles. The current version of EMFAC (EMFAC7F) has largely eliminated the discrepancy for CO, but is still believed to underpredict VOC by a factor of approximately 2.1.[1]

Table 4 lists estimated average emission rates of CO, VOC, NO_x, SO_x, and PM10 for each of sixteen vehicle types.[2] These vehicle types include the fleet-average on-road emissions for cars using gasoline, light-duty diesel trucks, and heavy-duty diesel trucks for the years 1992 and 2000.[3] The figures for CO are provided for completeness, but are not subsequently used in this paper. We have increased the VOC estimates for gasoline cars by a factor of 2.1 to correct for the underprediction problems with EMFAC7F. (Later, we similarly increase the motor vehicle portion of the total VOC inventory by the same factor.) Other vehicle types shown in the table are recent model diesel cars and trucks, and new cars meeting some of the new federal and California standards discussed earlier. These new car standards are not adjusted for the VOC discrepancy or any type of deterioration. For comparison, we also show emission rates for a composite of cars of various ages that met 1977 standards when new. Throughout the table, diesel SO_x emissions assume use of the low-sulphur diesel fuel now required in the Los Angeles air basin.

3. Methodology of Damage Estimation

There are at least three ways to infer the costs of air emissions. The best developed, and the one adopted here, is the direct estimation of damages. In this method, one traces the links between air emissions and adverse consequences, and attempts to place economic values on those consequences. Examples include Small (1977), US Federal Highway Administration (1982, p. E-47), Krupnik and Portney (1991), and Hall *et al.* (1992). Other methods include hedonic price measurement, in which observed price differentials are related to air quality; and revealed preference of policy-makers, in which pollution costs are inferred from the costs of meeting pollution regulations.

In the method of direct damage estimation, several links in the causal chain must be separately measured (Hall *et al.*, 1992). A pollutant emitted into the atmosphere changes

[1] Telephone conversation with Bart Croes, California Air Resources Board, Sacramento, August 1994. See also Fujita and Lawson (1994), p.1-4 and Table 3-11, for estimates of roughly the same magnitude.

[2] For the sake of clarifying our calculations, we show figures in this and subsequent tables with more significant digits than is justified by the precision of the estimates.

[3] The fleet average consists of all types and ages of the appropriate vehicle category, weighted by vehicle-miles travelled in the entire state of California as calculated by the EMFAC7F model. Total particulate matter (PM) emissions are converted into PM10 based on the estimate that 99 4 per cent of gasoline-powered vehicle PM emissions and 96 per cent of diesel-powered vehicle PM emissions are PM10 (telephone conversation with Robert Effa, California Air Resources Board, Sacramento, May 1994).

Table 4

Emission Factors for Selected Vehicles
(grams per mile)

	CO	VOC	NO_x	SO_x	PM10
1977 Model car, aged[a,b]	3 900	4 700	0.660	0.130	0 248
1992 Fleet averages[c]					
— Gasoline car[b]	13.000	3.757	1.260	0.038	0 011
— Light-duty diesel truck[d]	1.607	0.362	1.492	0.122	0.395
— Heavy-duty diesel truck[d]	9.326	2.356	15.683	0.576	2 359
Emission rates at 50,000 miles[e]					
— 1993 Heavy-duty diesel truck[d]	11.620	2 290	10 500	0.526	0.662
— 1994 Light-duty diesel truck[d]	1.220	0 351	1.110	0.122	0 106
— 1994 Light-duty diesel car[d]	1 120	0.351	0.980	0 107	0.106
US new car standards[f]					
— 1993	3.400	0 410	1 000	0	0
— Tier I	3 400	0.250	0 400	0	0
— Tier II	1 700	0.125	0 200	0	0
California new car standards[f]					
— 1993	3.400	0.250	0.400	0	0
— LEV	3 400	0.075	0 200	0	0
— ULEV	1.700	0.040	0.200	0	0
2000 Fleet averages[b,c]					
— Gasoline car	5.946	1.804	0.688	0.008	0.010
— Light-duty diesel truck	1.825	0.401	1.535	0.111	0.121
— Heavy-duty diesel truck	8.815	1.978	12.307	0.520	1.181

[a] From Small (1977) Table 5, row 9. PM10 is assumed to be 99.4 per cent of PM (see note 3).
[b] VOC emissions for gasoline cars are adjusted by a factor of 2.1 to reflect underestimation (see text)
[c] EMFAC7F estimates from the California Air Resources Board.
[d] The assumed SO_x emissions from diesels in the EMFAC model overstate those for the Los Angeles region in the early 1990s because the sulphur content of fuel is lower in the region than elsewhere in California. We recalculated the emissions by assuming 0.05 per cent elemental sulphur (S) content by weight, the legal standard in the region. We assume that fuel density is 3,249 grams/gallon and that each gram of S generates two grams of SO_2: then SO_x emissions = 0.0005 x 6498 = 3.249 grams/gallon of fuel. The 1992 fleet average light-duty and heavy-duty diesel trucks have fuel economy of 26.62 and 5.64 mpg from EMFAC. We assume that 1993 and 1994 diesel vehicles have the same fuel economy as the projected year 2000 fleet averages for the same vehicle types (for example, 6.18 mpg for heavy-duty trucks and 26.5 mpg for light-duty trucks) For the year 2000 fleet averages, the EMFAC model is accurate for Los Angeles because by then most of the state will have the same sulphur standard for fuel content as Los Angeles currently has.
[e] Air Resources Board's exhaust emission rates which are used as input to the EMFAC7F model.
[f] From Table 2

the spatial and temporal patterns of ambient concentrations of that pollutant and perhaps others. These patterns are determined by atmospheric conditions, topographical features and the presence of other natural or man-made chemicals in the air. The resulting ambient concentrations then interact with people, buildings, plants and animals in a way that depends on their locations and activity levels. The results may be physical and/or psychological effects: coughing, erosion of stone, retarded plant growth, injury to young, loss of pleasurable views, and so forth. Finally, these effects have an economic value.

Most studies of direct damages find that human health effects are the dominant component of air pollution costs. We therefore focus our efforts on the various causal links that lead to deterioration of human health.

In any concrete application, there are several conceptual and practical issues to be resolved. We discuss three in this section.

Willingness to pay

Economists widely accept the principle that in market economies the social cost of a change in economic outcomes is most usefully measured by the sum of individuals' willingness to pay for that change, given their current economic circumstances. The underlying argument is that if public policy consistently uses such valuations, most people will find that their overall welfare is improved (Hotelling, 1938; Hicks, 1941). Thus, to measure the cost of increasing frequency of headaches, say, one tries to determine how much people would be willing to pay to reduce that frequency, assuming they are presented with full information and clear choices.

Similarly, the appropriate valuation of small increases in the risk of dying from lung disease is the amount that people are willing to pay to reduce that risk marginally. It turns out that there are many situations in which one can observe such a quantity, particularly in the labour market where jobs with varying risk levels command compensating wage differentials. Excellent reviews and conceptual discussions include Kahn (1986), Fisher *et al.* (1989), Jones-Lee (1990) and Viscusi (1993). When studies with clear biases are eliminated, the evidence consistently shows that people in nations with high standards of living are willing to pay over one thousand dollars per year to reduce their annual mortality risk by 1 in 1,000: that is, their average "value of life" is more than one million dollars. Fisher *et al.* (1989) make an excellent case for a valuation between $2.1 million and $11.3 million at 1992 prices.[4] We take this as the most reasonable range, and choose the geometric mean ($4.87 million) as our best estimate. When price levels are adjusted for, this mean value is very close to the figure adopted by Hall *et al.* (1992), and is about half the value recommended by Kahn (1986).[5]

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[4] We inflate by US gross domestic product per capita, on the assumption that people's valuations grow with income. GDP per capita grew 33.3 per cent from 1986 to 1992 (from US Council of Economic Advisors, 1994, Table B-6).

[5] Krupnik and Portney (1991) argue that people place a lower value on risk of death from air pollution than is inferred from labour market studies because deaths from air pollution occur on average to older people who would have shorter life expectancies. Fisher *et al.* (1989), in contrast, point out that people demand greater compensation for risks that are out of their control.

Joint costs and non-linear effects

Non-linear relationships may occur at several points in the chain of causality linking emissions to costs: in the formation of ozone from VOC and NO_x emissions, in the exposure of people to ambient pollution concentration, and in the biological response to such exposure. This means that the cost of a given emission depends on what pollutants are already in the air, and total costs cannot necessarily be allocated among its constituents.

Ozone formation provides an especially difficult example (National Research Council, 1991). Ozone in the lower atmosphere is created through complex chemical reactions involving volatile organic compounds and nitrogen oxides. If the ratio of ambient levels of VOC to NO_x is high, ozone formation is said to be "NO_x-limited," that is, at the margin ozone depends just on NO_x emissions. If the VOC/NO_x ratio is low, ozone formation is "VOC-limited" so that reducing NO_x has little marginal effect (and may even increase ozone in the immediate vicinity of the sources). Recent evidence has led scientists to conclude that VOC emissions from both natural and man-made sources are higher than previously believed; this has raised estimates of VOC/NO_x ratios, so there is a renewed interest in control of NO_x (National Research Council, 1991).

Most other relationships appear reasonably approximated by linear ones at the relatively low chemical concentrations, measured in parts per million or parts per billion, with which we are concerned. This is an important and essential simplification, so it is worth defending in some detail.

The ambient concentration of a primary (that is, directly emitted) pollutant will be proportional to emissions if it is removed from a given volume of air at a rate proportional to its concentration. This is normally the case, and therefore a proportional linear relationship is widely used for primary pollutants (Ball *et al.*, 1991, p.30). Even for secondary pollutants such as ozone, concentrations tend to be close to linear functions of equiproportionate increases in precursor emissions over much of the range of interest. For example, graphs relating predicted average ozone concentrations in the Los Angeles basin to percentage region-wide cutbacks in VOC and NO_x emissions show that ambient ozone declines very close to linearly as its precursors are reduced by equal percentages (Milford *et al.*, 1989, fig. 13).[6] Secondary pollutant formation appears more non-linear if one measures it by extremes, such as the peak concentration regardless of location in an air basin.[7] Such a measure is appropriate for analysing compliance with legal air quality standards, but not for computing social costs throughout a region.

With respect to health effects, there are several reasons to accept linearity between concentrations and aggregate health costs as a good approximation for aggregate analysis. Numerous reviews of health effects of various pollutants have found no convincing

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[6] This can be seen by the nearly equal spacing between isopleths as one moves along a diagonal line connecting the points representing no control and 100 per cent control of both precursors. Note that our argument for ozone is for linearity, not proportionality

[7] An example is Milford *et al.* (1989), fig. 11, reprinted in National Research Council (1991), p 176. Even this figure is very close to linear along a ray from baseline emissions down to about 50 per cent cutbacks in both precursors, and along several other rays as well

evidence of thresholds in most cases: for example National Academies of Science and Engineering (1974, pp.6, 190, 366-367, 400) and Schwartz *et al* (1988). Non-linearities in dose-response relationships are sometimes assumed without evidence because of the legal status of ambient air standards; yet these standards are determined more by the sensitivity of the studies upon which they are based than by any evidence of thresholds (Horowitz, 1982, pp.113-115). Even when a threshold was thought to exist, further research has often uncovered evidence of effects below that threshold, as described by Dockery *et al.* (1993) for particulates, and Lippmann (1993) for ozone. Finally, any non-linearities that may exist for individuals or for specific locations will be reduced or eliminated in aggregate populations by the effects of averaging over individual susceptibilities, behaviour patterns and locations.

Exposure

While considerable scientific effort has gone into the atmospheric modelling necessary to predict ambient air concentrations, far less has gone into understanding where and when people actually become exposed to those concentrations. Yet concentrations vary widely over time and space, and people's exposure depends heavily on whether they are indoors or outdoors, in or out of a vehicle, and how strenuously they are exercising. Hall *et al.* (1992) pioneer the use of comprehensive models to quantify this exposure, and some of our estimates make use of their results.

4. Estimates for the Los Angeles Region

In this section we estimate the per-mile costs of air emissions from various classes of motor vehicles in the Los Angeles region. We do so within a framework that can be readily updated to reflect new information or alternative assumptions. Unless otherwise stated, all monetary figures are in US dollars at 1992 prices.

Because the topography of Los Angeles tends to trap pollutants and the climate favours photochemical reactions, the costs of a given amount of emission are likely to be higher than in most areas. For example, Small (1977, pp.123-124) assembles evidence that meteorological conditions produce about six times as many days with low mixing heights and wind speeds in Los Angeles as in twelve US cities outside California. In addition, our valuation of risk of death is applicable only to developed nations.

Attempting to focus on the avenues of damage that current evidence suggests are most important, we estimate three main categories of costs: mortality from particulates, morbidity from particulates, and morbidity from ozone. We do not deal with carbon monoxide despite its known adverse health effects and its importance in pollution control strategies because there is little or no quantitative information suitable for measuring its health costs (Hall *et al.*, 1992). The only estimates available are extremely low and not applicable to ground-level emissions (Davis and Chaudhry, 1993). Work in progress may remedy this deficiency (DeLucchi *et al.*, forthcoming), but we believe it will show CO to be of less importance in costing motor vehicle emissions than either particulates or ozone.

Mortality: Particulates

Evans *et al.* (1984) review the extensive statistical evidence linking mortality in U.S. metropolitan areas to ambient pollution concentrations. They conclude:

> "We are of the opinion that the cross-sectional studies reflect a causal relationship between exposure to air-borne particles and premature mortality.... However, we are in the minority in taking this view." (p.78)

They also conclude that "the apparent association is weak to moderate in strength, [and] that the specific type of particle responsible has not been identified...." (p.78)

Subsequent studies provide further support for their opinion that particulate matter causes increased mortality. These studies also suggest that inhalable particles, that is, particulate matter of less than 10 microns diameter (PM10), are the most responsible. Among the components of PM10, the most consistently found effects are from fine particles (FP), which are those with a diameter of less than 2.5 microns, and from sulphates (SO_4), which are mainly aerosols of aluminum sulphate.[8]

Özkaynak and Thurston (1987) and Dockery *et al.* (1993) provide two of the most compelling such studies. Özkaynak and Thurston use cross-sectional data from 1980, when pollution measurements were much more complete and accurate than in earlier years. As in the studies reviewed by Evans *et al.* (1984), they relate mortality in metropolitan areas to ambient pollutant concentrations in the downtown areas of their central cities. They use four alternative measures of particulates: total suspended particulates (TSP), PM10, FP and SO_4. The strongest and most consistent relationship is obtained using SO_4, and the next strongest using FP. The broader categories, PM10 and TSP, are not reliably correlated with mortality when other control variables are taken into account. An effort to trace fine particulates to their source type provides suggestive though inconclusive evidence that particulates arising from the metal and coal industries are more damaging than those arising from motor vehicles, the oil industry or windblown soil.

Dockery *et al.* (1993) describe the only study based on micro data to address these same issues. They construct proportional hazard models to explain mortality among 8,111 adults in six US cities, each followed for up to 16 years. They find strong effects of lagged values of pollution as measured by SO_4, FP or PM10. These effects are not greatly affected by controls for smoking, education, occupational exposure and other variables. The effects are so strong that adjusted mortality rates (that is, rates after controlling for other factors) go up by 26 per cent as particulate levels are raised from those in the least polluted of the six cities to those of the most polluted. This magnitude seems implausibly large to us, so it is not used in our quantitative estimates.

From the results reviewed here, it seems clear that sulphate aerosols cause increased mortality, and that other components of PM10 may also. These findings are consistent with other evidence that finer particles are more readily deposited on human and animal airways than coarser particles, and that acidic aerosols are especially damaging (US EPA, 1982; Lippmann and Lioy, 1985). (As a result of such findings, the TSP ambient standard

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8 Sulphate particles that are small enough in size are included in PM10 and FP

in the US was replaced by a PM10 ambient standard during the 1980s.) These observations provide two alternative hypotheses for estimating the dose-response relationship: one based on the coefficient of FP, the other based on that of SO_4, when each is separately included as the sole pollution-related independent variable explaining mortality. We choose the results of Özkaynak and Thurston (1987), Table III, model M1. The relevant coefficients are measured as the increased annual metropolitan-area deaths per 100,000 population for a unit increase in particulate concentration in micrograms per cubic meter ($\mu g/m^3$) as measured at a central monitoring station. The coefficients (standard errors in parentheses) are 6.6 (1.5) for SO_4 and 2.2 (0.8) for FP. If we assume that all the constituents of PM10 (including FP) are equally harmful and that FP is a fixed proportion of PM10 throughout the sample, then a coefficient for PM10 can be imputed. Lacking such data nationwide, we use information for the Los Angeles region, where 59 per cent of PM10 is FP (SCAQMD, 1994, Appendix I-D); hence the imputed PM10 coefficient is 0.59x2.2 = 1.298.

An alternative and somewhat lower pair of coefficients is obtained by using the reanalysis by Evans *et al.* (1984) of the 1960 data used by Lave and Seskin (1977). These estimates again show the results of using one pollution measure at a time. The coefficients (standard errors in parentheses) are 3.72 (1.90) for SO_4 and 0.338 (0.198) for TSP (Evans *et al.*, 1984, Table 18). Based on the consensus that PM10 rather than TSP causes health damages, we assume that among the constituents of TSP, only PM10 causes mortality; then the same coefficient (0.338) applies to changes in PM10.[9]

Another line of evidence comes from time-series studies relating daily mortality rates to daily concentrations of particulates. Schwartz (1994) performs a meta analysis of more than a dozen data sets, finding that each increase in PM concentration of 1 $\mu g/m^3$ increases total mortality by 0.06 per cent. Multiplying this by the current mortality rate in the Los Angeles region of about 870 deaths per 100,000 people (US Department of Health and Human Services, 1993, p.1), we obtain a coefficient of 0.522, which lies between the higher and lower values just discussed. This is reassuring, although we believe the time-series evidence is less appropriate because one cannot separate the long-term causal effects of interest from the short-term timing of the course of a fatal illness. For example, a single day of high pollution might trigger the deaths of many people weakened by diseases unrelated to air pollution; or at the other extreme, daily deaths might show no correlation with daily pollution levels even though chronic illnesses are caused by long-term exposures.

PM10 concentrations result directly from emission of particulates and indirectly from emission of VOC, NO_x, and SO_x. NO_x and SO_x react in the atmosphere, particularly in clouds, to form droplets of nitric and sulphuric acid and also particles of ammonium nitrate and ammonium sulphate (Charlson and Wigley, 1994). SCAQMD (1994, Appendix I-D) estimates that direct particulate emissions from motor vehicles account for 10.6 per cent of PM10, and that the indirect components from motor vehicle emissions are as follows:

[9] Alternatively, Hall *et al.* (1992) impute a higher coefficient to PM10 by implicitly assuming that all components of TSP are equally harmful and that PM10 is serving as a proxy for all TSP, just as we assume that FP is a proxy for all PM10

4.4 per cent of PM10 is secondary carbon from VOC emissions, 10.5 per cent is ammonium nitrate from NO_x, and 5.3 per cent is ammonium sulphate from SO_x.

Our calculations require estimates of ambient pollution levels and emissions, so that a given reduction of emissions can be translated into a reduction in ambient levels. For the ambient PM10 concentration, which fluctuates from year to year due to weather, we average four years of annual-average observations at the downtown Los Angeles monitoring site from 1987 through 1990 (Cohanim *et al.*, 1991, Tables 34-37). This gives a value of 57.8 $\mu g/m^3$. We use the Los Angeles site because it is most comparable to the downtown locations of the monitors used in the cross-sectional mortality studies that underlie our mortality calculations. As an alternative scenario, we recalculate using the San Bernardino monitoring site, whose annual average PM10 concentration of 77.75 $\mu g/m^3$ is the highest in the region.

For emissions, which are more stable, we use region-wide data for the year 1990.[10] These data include both natural (such as soil and marine salt) and human sources. For those scenarios in which all components of PM10 are assumed to contribute equally to mortality, costs are allocated to precursor emissions using source contribution percentages from SCAQMD (1994).[11] We also compute a scenario in which only sulphates are assumed to be responsible for mortality which, as noted earlier, is consistent with the evidence; this is done by attributing all mortality to SO_x emissions.[12]

The total PM10 inventory used as the basis for the above percentages includes geological sources such as soil particles and road dust. We have not included road dust as a motor vehicle emission, partly because its emissions inventory is thought to be especially unreliable and is currently under review.

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10 Originally estimated emissions in tons per day during 1990 were. VOC, 1,469; NO_x, 1,290, SO_x, 121; and PM10, 838 Of these the following amounts were direct emissions from on-road vehicles: VOC, 761, NO_x, 762, SO_x, 31; and PM10, 70. Source: SCAQMD (1994), Table 3-2A. Applying the correction factor of 2 1 to on-road VOC emissions (see text) increases the VOC figures to 2,306 total and 1,598 on-road

11 For example, 10.5 per cent of ambient PM10 at the downtown Los Angeles monitoring station is ammonium nitrate traceable to on-road vehicle NO_x emissions (calculated from SCAQMD, 1994, Appendix I-D, Table 3-2). Therefore the 762 tons/day (278,130 tons/year) of on-road NO_x emissions quoted in note 10 add $0.105 \times 57.8 = 6.069$ $\mu g/m^3$ to the PM10 concentration Applying Evans's mortality coefficient, for example, these emissions are responsible for an increase in annual mortality of $0.338 \times 6.069 = 2.0513$ deaths per 100,000 people per year. In a region with 12 million people this means $2.0513 \times 120 = 246$ deaths caused by 278,130 tons of NO_x emissions. Assuming a proportional rollback model and valuing deaths at our central estimate of \$4.87 million yields an estimate of cost of mortality of $246 \times \$4.87$ million/278,130 = \$4,310/ton Similar calculations were performed for VOC, SO_x and direct PM10 emissions.

12 The calculation is similar to the NO_x example in note 11, except that ammonium sulphate is the only component of PM10 that is assumed to contribute to mortality An estimated 5 33 per cent of ambient PM10 at the downtown Los Angeles monitoring station is ammonium sulphate traceable to on-road SO_x emissions Therefore the 31 tons/day (11,315 tons/year) of on-road SO_x emissions add $0.0533 \times 57.8 = 3.0807$ $\mu g/m^3$ to ambient sulphate concentrations. Applying the geometric average of Özkaynak-Thurston's and Evans's sulphate coefficients, which is $(6.6 \times 3.72)^{1/2} = 4.955$, these emissions are responsible for an increase in annual mortality of $4.955 \times 3.0807 = 15.265$ deaths per 100,000 people Given a population of 12 million, this means $15.265 \times 120 = 1{,}831.8$ deaths from 11,315 tons of SO_x emissions, valuing each statistical death at \$4.87 million yields a cost of \$788,400/ton.

Morbidity: Particulates and Ozone

Hall *et al.* (1992) report the results of an elaborate exposure model for the Los Angeles region in which the spatial distributions of pollution and people's activity patterns are combined with estimated hours of exposure under various locations (indoors, outdoors, in-transit) and breathing rates (sleeping, at rest, exercising). Dose-response functions for a variety of acute effects such as coughing and eye irritation are applied to these exposures, and the resulting effects are valued according to the results of contingent valuation surveys. Certain details of their model have been criticized by Harrison and Nichols (1989), who claim that Hall *et al.* overestimated the costs. As a result, Krupnick and Portney (1991) apply their own less elaborate modelling scheme for the case of ozone effects, though not for particulates. Krupnik and Portney also incorporate newer evidence on dose-response relationships, including many of the studies recently reviewed by Ostro (1994).

Because we are unable to disentangle the many parts of these very detailed models, we use the resulting costs reported by Hall *et al.* and by Krupnik and Portney as high and low estimates, respectively, of morbidity costs. Their results appear to estimate the costs that would be saved if pollution levels in the Los Angeles region were lowered according to the 1989 Air Quality Management Plan (SCAQMD and SCAG, 1989). We estimate the precursor emissions with and without this plan and attribute the morbidity costs to these precursors in accordance with simple models of how the precursors produce ambient concentrations of PM10 and ozone.

Ozone is formed as a result of emissions of VOC and NO_x. Suppose as an approximation that the function $f(E_V, E_N)$ relating them is homogeneous of degree one, so that doubling both types of emissions would double ozone concentration. Then the total effect of the two types of emission is allocable to each in amounts proportional to their marginal effects, $f_V \equiv \partial f/\partial E_V$ and $f_N \equiv \partial f/\partial E_N$: that is, $f(E_V, E_N) = f_V E_V + f_N E_N$. (This is an application of the Euler Theorem.) Equivalently, the elasticity of ozone concentration with respect to VOC emissions, $(E_V/f)(\partial f/\partial E_V)$, is the fraction $E_V f_V/(E_V f_V + E_N f_N)$.

Empirical measurements of the function f indicate that in realistic situations it is entirely plausible for this fraction to lie anywhere between zero (NO_x-limited) and one (VOC-limited), or even to be greater than one (when ozone declines as NO_x increases). For example, Mayeres *et al.* (1993, fig.6) present iso-ozone curves for Belgium which imply that the fraction ranges from zero (under scenarios with 30 per cent or more NO_x reduction from present levels) to about 1.5 (under scenarios with 15 per cent or more VOC reduction from present levels). A similar range applies to the iso-ozone curves for several sites in the Los Angeles region computed by Milford *et al.* (1989, figs.4, 11). According to them, meeting US federal ozone standards would require very large reductions in both NO_x and VOC, and ozone would then be NO_x-limited so that f_V would be zero. (One reason for this is that there are substantial natural sources of VOC.) Under current conditions, on the other hand, their graphs show modest NO_x reductions to be ineffective or even counter-productive in lowering ozone in the most heavily populated areas of the region, so that f_N is near zero or even negative. Between these extremes are many starting points for which both VOC and NO_x reductions are effective.

We therefore consider three alternative scenarios in which the elasticity of ozone with respect to VOC emissions is zero, one, or one-half. This amounts to allocating the health benefits of ozone reduction entirely to NO_x, entirely to VOC, or equally to NO_x and VOC.

For particulate morbidity, Hall *et al.* estimate an annual cost saving of \$919 million in 1992 prices[13] for meeting federal ambient air standards in the year 2010. The same estimate is used by Krupnick and Portney. It is based on modelling done for the 1989 air quality management plan, which shows a required reduction in PM10 concentrations, projected for the year 2010, of 48.3 μg/m³ (from 73.8 to 25.5) (SCAQMD and SCAG, 1989, Figure 5-7). This gives us a cost estimate of \$19.03 million per year for each one-unit change in PM10 concentration. We allocate this cost to the four primary emissions in exactly the same way as for particulate mortality.[14]

For ozone morbidity, both Hall *et al.* (1992) and Krupnick and Portney (1991) estimate the annual cost of failing to meet US federal ozone standards. Their results, again in 1992 prices, are \$3.2 billion and \$0.36 billion, respectively. Meeting those standards would require an 87.5 per cent reduction in total VOC emissions (including natural sources) and a 79.9 per cent reduction in NO_x emissions, or 1,275 and 813 tons/day, respectively.[15] This enables us to allocate the health costs of morbidity to NO_x, VOC, or a combination of the two, as described earlier.[16] Admittedly we are stretching the linearity assumptions by applying them to such large reductions.

Results: Costs per Unit of Emission from Motor Vehicles

Our calculations of the cost per ton of emissions of various pollutants are shown in Table 5, for alternative sets of assumptions. The table reveals that mortality from particulate matter is the dominant component of costs of VOC, NO_x, SO_x and PM10 emissions. For NO_x, costs arising from the production of secondary particulate matter are several times higher than those from the production of ozone, although there are some combinations of assumptions for which this is not the case. For VOC, costs from secondary particulate

[13] We have inflated their \$775 million estimate, which was in 1988 prices, by the consumer price index, consistent with their treatment of price levels. It makes virtually no difference if we inflate by GDP per capita instead.

[14] Again using the ammonium nitrate portion of PM10 as an example, a 48.3 μg/m³ reduction in ambient PM10 valued at \$919 million leads to a value of NO_x emissions of (\$919,000,000/48.3 μg/m³) x (0 105 x 57 8 μg/m³)/(762 x 365 tons/year) = \$415/ton.

[15] The applicable air quality management plan shows emissions in a baseline scenario (1989 control policies) for the year 2010, and reductions needed to meet the federal standards (SCAQMD, 1989, Tables 4-1, 4-11 and 4-14). For VOC, baseline emissions are 1,130 tons/day, the reduction is 948 tons/day, of which 327 tons/day is from transport sources. We assume that the transport-related VOC reductions required are underestimated by a factor of 2.0 (that is, by 327 tons/day), and that total baseline VOC emissions are underestimated by the same absolute amount (327 tons/day). Hence total baseline emissions of 1130 + 327 = 1457 tons/day would be reduced by 948 + 327 = 1275, or 87 5 per cent. We use the factor 2.0 instead of 2 1, used elsewhere, because we apply it to all transport sources instead of just gasoline-powered vehicles.

[16] For example, using the higher ozone morbidity estimate of \$3.2 billion from Hall *et al.*, allocating the cost equally to both types of emissions yields cost estimates for VOC of (\$1 6 billion/year)/(1275 x 365 tons/year) = \$3,438/ton, and for NO_x of (\$1.6 billion/year)/(813 x 365 tons/year) = \$5,392/ton.

Table 5

Costs for the Los Angeles Region
(1992 $1000's per ton of emission)

	VOC	*NO_x*	*SO_x*	*PM10*
Mortality from Particulates				
$2.1M value of life				
– Low mortality coefficient	0.37	1.86	23.1	21.4
– Geometric average mortality coefficient	0.73	3.64	45.2	41.9
$4.87M value of life				
– Low mortality coefficient	0.86	4.31[c]	53.5	49.6
– High mortality coefficient	3.31	16.55	205.4	190.5
– Geometric average mortality coefficient[a]	**1.69**	**8.45**	**104.8**	**97.2**
– Geometric average mortality coefficient using San Bernardino PM10[b]	2.27	11.36	141.0	130.7
$11.3M value of life				
– High mortality coefficient	7.67	38.41	476.5	442.0
– Geometric average mortality coefficient	3.92	19.60	243.2	225.5
All damage from SO_x, $4.87M value of life				
– Geometric average mortality coefficient	0	0	788.4[d]	0.0
Morbidity from Particulates				
– Cost used by Hall *et al.* and Krupnick-Portney[a]	**0.08**	**0.42[e]**	**5.2**	**4.8**
– Same cost as above, but using San Bernardino PM10[b]	0.11	0.56	6.9	6.4
Morbidity from Ozone				
Krupnick-Portney's cost figure				
– Attributing 1/2 to NO_x, 1/2 to VOC	0.39	0.61	0	0
Hall *et al.*'s cost figure				
– Attributing 1/2 to NO_x, 1/2 to VOC	3.44[f]	5.39[f]	0	0
Geometric average of Krupnick-Portney and Hall *et al*				
– Attributing all damage to NO_x	0	3.62	0	0
– Attributing all damage to VOC	2.31	0	0	0
– Attributing 1/2 to NO_x, 1/2 to VOC[a]	**1.15**	**1.81**	**0**	**0**
Total Figures				
– Baseline assumptions[a]	**2.92**	**10.67**	**109.9**	**102.0**
– Baseline, but attributing ozone morbidity to NO_x	1.77	12.48	109.9	102.0
– Baseline, but attributing ozone morbidity to VOC	4.08	8.86	109.9	102.0
– Baseline, but attributing particulate mortality to SO_x	1.24	2.22	793.6	4.8
– Baseline, but using San Bernardino PM10[b]	3.53	13.73	147.9	137.1

[a] Baseline assumptions use a $4.87M value of life, the geometric average of the high and low particulate mortality coefficients, the geometric average of two ozone morbidity figures with the costs equally attributed to NO_x and VOC, and the only particulate morbidity figure

[b] All other calculations use PM10 readings from the downtown Los Angeles monitor.

[c] For the derivation of this estimate, see footnote 11.

[d] For the derivation of this estimate, see footnote 12.

[e] For the derivation of this estimate, see footnote 14

[f] For the derivation of this estimate, see footnote 16

Table 6

Cost of the 1992 California Fleet Average Gasoline Car in the Los Angeles Region
(cents per vehicle-mile)

	VOC	*NO_x*	*SO_x*	*PM10*	*Total*
Mortality from Particulates					
– Low mortality coefficient and $2 1M value of life	0.15	0.26	0.10	0.03	0.54
– Geometric average mortality coefficient and $4.87M value of life[a]	**0.70**	**1.17**	**0.44**	**0.12**	**2.43**
– Geometric average mortality coefficient, $4 87M value of life, and San Bernardino PM10[b]	0 94	1.58	0.59	0 16	3 27
– High mortality coefficient and $11.3M value of life	3 18	5.33	2.00	0.55	11.06
– All damage from SO_x using geometric average mortality coefficient and $4.87M value of life	0	0	3.30	0	3 30
Morbidity from Particulates					
– Cost used by Hall *et al.* and Krupnick-Portney[a]	**0.03**	**0.06**	**0.02**	**0.01**	**0.12**
– Same cost as above, but using San Bernardino PM10[b]	0 07	0 08	0 03	0 01	0 16
Morbidity from Ozone					
– Krupnick-Portney's cost, 1/2 to NO_x, 1/2 to VOC	0.16	0.08	0	0	0.24
– Hall *et al.*'s cost, 1/2 to NO_x, 1/2 to VOC	1.42	0 75	0	0	2 17
– Geometric average cost, all to NO_x	0	0.50	0	0	0.50
– Geometric average cost, all to VOC	0.96	0	0	0	0.96
– Geometric average cost, 1/2 to NO_x, 1/2 to VOC[a]	**0.48**	**0.25**	**0**	**0**	**0.73**
Total Figures:					
– Baseline assumptions[a]	**1.21**	**1.48**	**0.46**	**0.13**	**3.28**
– Baseline, but high particulate mortality coefficient and $11.3M value of life	3 69	5.64	2.02	0.56	11 91
– Baseline, but low particulate mortality coefficient and $2.1M value of life	0 67	0.57	0 12	0 03	1 38
– Baseline, but Hall *et al.*'s ozone morbidity	2.16	1.98	0.46	0.13	4 72
– Baseline, but Krupnick-Portney's ozone morbidity	0.89	1.32	0.46	0.13	2.80
– Baseline, but attributing ozone morbidity to NO_x	0.73	1.73	0.46	0.13	3.05
– Baseline, but attributing ozone morbidity to VOC	1.69	1.23	0.46	0.13	3 51
– Baseline, but attributing particulate mortality to SO_x	0.51	0.31	3.32	0.01	4 15
– Baseline, but using San Bernardino PM10[b]	1 49	1.91	0.62	0.17	4 16

[a] Baseline assumptions use a $4.87M value of life, the geometric average of the high and low particulate mortality coefficients, the only particulate morbidity figure, and the geometric average of two ozone morbidity figures with the costs equally attributed to NO_x and VOC.
[b] All other calculations use PM10 readings from the downtown Los Angeles monitor.

Table 7

Cost of the 1992 California Fleet Average Heavy-duty Diesel Truck in the Los Angeles Region
(cents per vehicle-mile)

	VOC	*NO_x*	*SO_x*	*PM10*	*Total*
Mortality from Particulates					
– Low mortality coefficient and $2.1M value of life	0.10	3.21	1.46	5.56	10.33
– Geometric average mortality coefficient and $4.87M value of life[a]	**0.44**	**14.60**	**6.65**	**25.27**	**46.97**
– Geometric average mortality coefficient, $4.87M value of life, and San Bernardino PM10[b]	0.59	19.64	8.95	34.00	63.18
– High mortality coefficient and $11.3M value of life	1.99	66.40	30.26	114.91	213.56
– All damage from SO_x using geometric average mortality coefficient and $4.87M value of life	0	0	50.06	0	50.06
Morbidity from Particulates					
– Cost used by Hall *et al.* and Krupnick-Portney[a]	**0.02**	**0.72**	**0.33**	**1.24**	**2.31**
– Same cost as above, but using San Bernardino PM10[b]	0.03	0.97	0.44	1.67	3.11
Morbidity from Ozone					
– Krupnick-Portney's cost, 1/2 to NO_x, 1/2 to VOC	0.10	1.05	0	0	1.15
– Hall *et al.*'s cost, 1/2 to NO_x, 1/2 to VOC	0.89	9.32	0	0	10.21
– Geometric average cost, all to NO_x	0	6.25	0	0	6.25
– Geometric average cost, all to VOC	0.60	0	0	0	0.60
– Geometric average cost, 1/2 to NO_x, 1/2 to VOC[a]	**0.30**	**3.13**	**0**	**0**	**3.43**
Total Figures:					
– Baseline assumptions[a]	**0.76**	**18.45**	**6.98**	**26.51**	**52.70**
– Baseline, but high particulate mortality coefficient and $11.3M value of life	2.31	70.24	30.59	116.16	219.30
– Baseline, but low particulate mortality coefficient and $2.1M value of life	0.42	7.06	1.79	6.80	16.07
– Baseline, but Hall *et al.*'s ozone morbidity	1.35	24.64	6.98	26.51	59.49
– Baseline, but Krupnick-Portney's ozone morbidity	0.56	16.37	6.98	26.51	50.43
– Baseline, but attributing ozone morbidity to NO_x	0.46	21.57	6.98	26.51	55.53
– Baseline, but attributing ozone morbidity to VOC	1.06	15.32	6.98	26.51	49.87
– Baseline, but attributing particulate mortality to SO_x	0.32	3.84	50.39	1.24	55.80
– Baseline, but using San Bernardino PM10[b]	0.92	23.74	9.39	35.67	69.72

[a] Baseline assumptions use a $4.87M value of life, the geometric average of the high and low particulate mortality coefficients, the only particulate morbidity figure, and the geometric average of two ozone morbidity figures with the costs equally attributed to NO_x and VOC.
[b] All other calculations use PM10 readings from the downtown Los Angeles monitor.

Table 8

Cost of Selected Vehicles using Baseline Assumptions in the Los Angeles Region
(cents per vehicle-mile)

	VOC	*NO_x*	*SO_x*	*PM10*	*Total*
1977 Model car, aged	1 51	0 78	1 58	2.79	6 65
1992 Fleet averages					
– Gasoline car	1 21	1 48	0 46	0.13	3 28
– Light-duty diesel truck	0 12	1 75	1 48	4.44	7 79
– Heavy-duty diesel truck	0.76	18.45	6 98	26.51	52 70
Emission rates at 50,000 miles					
– 1993 Heavy-duty diesel truck	0 74	12.35	6.37	7 45	26 91
– 1994 Light-duty diesel truck	0 11	1 31	1 48	1.19	4 09
– 1994 Light-duty diesel car	0 11	1 15	1.30	1.19	3.76
US new car standards					
– 1993	0 13	1.18	0	0	1 31
– Tier I	0 08	0 47	0	0	0.55
– Tier II	0 04	0.24	0	0	0.28
California new car standards					
– 1993	0.08	0 47	0	0	0 55
– LEV	0 02	0.24	0	0	0.26
– ULEV	0.01	0.24	0	0	0.25
2000 Fleet averages					
– Gasoline car	0 58	0.81	0.10	0.11	1.61
– Light-duty diesel truck	0 13	1.81	1.35	1.36	4 64
– Heavy-duty diesel truck	0 64	14.48	6.30	13.27	34.69

formation are about 50 per cent higher than those from ozone production under baseline assumptions. It appears likely, then, that particulate matter creates a more serious health problem than ozone, even in Los Angeles with its notorious ozone concentrations.

Because mortality is the dominant cost, our cost estimates are sensitive to assumptions about mortality coefficients and value of life. Using the lowest assumptions about both reduces our cost estimates for SO_x and PM10 by a factor of about 4.5; using the highest assumptions raises them by the same factor. Cost estimates for VOC and NO_x emissions are somewhat less sensitive. Cost estimates for specific emissions are altered a great deal if particulate damage is assumed to be due only to sulphates. Only the costs of VOC, and to a lesser extent of NO_x, are sensitive to how ozone damage is allocated.

Results: Costs of Motor Vehicle Emissions

Table 6 shows the results of applying these estimates of cost per ton to the emissions of a 1992 California fleet-average gasoline-powered car. (Those emissions were shown in Table 4.) Under the baseline assumptions, total cost is just over 3 cents per vehicle-mile. Nearly half of this cost is from NO_x, due largely to its role in particulate formation. Close behind in damaging potential is VOC, followed by SO_x and finally directly emitted PM10.

Under various assumptions, costs range from 1.4 to nearly 12 cents per vehicle-mile. Surprisingly, the cost goes up if we assume that mortality arises only from sulphates instead of all PM10; this result would change if the sulphur content of gasoline were reduced. Whether ozone damage is allocated to VOC or NO_x makes little difference to these estimates.

Table 7 shows a very different situation for a 1992 California fleet-average heavy-duty diesel truck using the low-sulphur fuel required in Los Angeles. Its costs are very high: 53 cents per vehicle-mile under baseline assumptions, with a range from $0.16 to $2.19. More than half the damage comes from direct emissions of PM10, and another third from NO_x. Total costs are even larger if all mortality is assumed to be from sulphates.

Table 8 shows our baseline cost estimates — that is, under intermediate assumptions — for each of the vehicles whose emissions were previously presented in Table 4. Diesel cars, which differ little from light-duty diesel trucks, are two to three times as costly as gasoline cars; this is mainly due to higher SO_x and PM10 emissions. New cars and trucks in 1993 are about half as costly as the 1992 fleet averages just discussed. In the case of gasoline cars, the federal Tier I controls will reduce pollution cost from a new vehicle by more than half compared to 1993 cars, to 0.6 cents per vehicle-mile. This remaining cost is probably insignificant to vehicle use decisions, although it is equivalent to roughly half the total gasoline taxes now paid by such a vehicle. These estimates for new cars are under the legally defined test cycle and do not necessarily reflect on-road emissions.

Tighter standards will further reduce emissions from new trucks and buses after 1993; but slow turnover produces a very high cost — 35 cents per mile — for the fleet average heavy-duty diesel truck in the year 2000. This suggests that driving restrictions on heavy vehicles in urban areas may well be justified, at least in areas like Los Angeles where the topography tends to concentrate pollutants. As a sensitivity check, we recalculated these results under the assumptions that dust from paved roads is attributable to motor vehicles and allocable on a per vehicle-mile basis. The result is to add about 4.3 cents/mile to the cost of all light-duty vehicles.[17] As noted earlier, SCAQMD's inventory of road dust is highly suspect so we do not give much credence to this estimate. Because paved road dust is not reduced by current emissions control policies, it seems especially important to refine our knowledge about this possibly large source of pollution costs.

[17] Allocating SCAQMD's inventory of PM10 from paved road dust, which is 405.2 tons/day, among the approximately 300 million daily vehicle-miles yields emissions of 1.23 g/mile. However, the unit cost imputed to PM10 emitted as road dust is only about one-third of that from other on-road motor-vehicle sources. The reason is that emitted quantities of PM10 in the SCAQMD inventory are about six times higher for road dust than for other motor-vehicle sources, whereas the fraction of ambient PM10 attributable to individual sources is only about twice as high; presumably this difference is because road dust settles out of the air more quickly than does other particulate matter from motor vehicles.

Some Comparisons

How large is 3 cents per vehicle-mile, our baseline estimate for an average 1992 vehicle in California? We can provide three useful points of comparison: earlier estimates of the same cost, likely responses to internalising the estimated cost, and estimates of other environmental costs of motor vehicles.

An earlier estimate by Small (1977) of air pollution costs resulted in 0.61 cents/mile for a post-1977 car operated in a California metropolitan area, stated in 1992 prices.[18] This is only about one-tenth of our estimate for the same vehicle shown in the top row of Table 8. The main reason for the difference is that our baseline value of life is 29 times higher than the discounted lost earnings used by Small (1977); that adjustment alone would bring the earlier estimate up to 5.34 cents/mile.[19] The earlier work also used different mortality estimates and a different method to allocate costs to specific emissions.

Next, consider what might happen if people were assessed charges or taxes of magnitude 3 cents per vehicle-mile. At retail gasoline prices about $1.20 per gallon and fleet average fuel economy 23 miles per gallon,[20] this additional charge would be a bit more than half as large as current fuel cost per mile. Past experience suggests that people pay attention to costs of such a magnitude, but not by altering their amount of driving very much. For example, Greene (1992) estimates that raising the fuel cost per mile by 10 per cent (either by raising fuel prices or by lowering fuel economy) reduces driving by 0.5-1.5 per cent; so if cars were charged a pollution tax equal to 50 per cent of current fuel prices, people would drive just 2.5-7.5 per cent fewer vehicle-miles. If the pollution tax varied with the vehicle's emission characteristics, however, people would be given strong incentives to purchase cars with better pollution controls; so in the long run the average tax per mile would decline and the effect on motor vehicle use would be even less, while adoption of cleaner vehicles would be greatly facilitated.

In contrast to the case for cars, our baseline cost estimate of 53 cents per mile for the average heavy-duty diesel truck is quite large. With fuel costs around 16 cents per mile and total operating expenses $1.30-$4.20 per mile,[21] charging this pollution cost would cause a significant change in trucking operations. Presumably, it would also greatly hasten the introduction of new lower-polluting vehicles, thereby lowering the appropriate charges.

[18] This inflates both health and materials costs by 3.466, the ratio of GDP per capita in 1992 to 1974 (US Council of Economic Advisers, 1994, Table B-6). California costs per unit of emissions, in $1000's per ton, are: 0.06 for CO; 0.97 for VOC; 3.18 for NO_x; 3.95 for SO_x; and 1.87 for PM (at that time PM10 was not distinguished either in data or regulations). These figures are computed as US costs multiplied by 2.9 to account for California topography and climate, as explained in Small (1977), pp.123-24.

[19] Small (1977) included materials cost, and just under half of health costs were for premature deaths, so raising value of life by a factor of 29 (4 87/0.166, to be exact) raises the final estimate only by a factor of 9 The adjusted costs per unit emission in $1000's per ton are. 0.88 for CO; 9.16 for VOC; 22.85 for NO_x; 21.25 for SO_x; and 26.45 for PM.

[20] The 1992 fleet average, from EMFAC7F input data.

[21] The fuel cost assumes a fuel price of $0.90 per gallon and fuel economy of 5.64 miles per gallon. Operating expenses and intercity vehicle-miles for four classes of Class I intercity carriers are given in US Bureau of the Census (1993), Table 1047.

Finally, how much might global warming add to these cost estimates? The scientific basis for estimating damage from global warming is especially shaky because the nature of the phenomenon is so uncertain and its anticipated effects, if they occur at all, will be cumulative and mostly in the distant future. However, there is considerable evidence from recent international agreements and negotiations about control measures that may be adopted in the near term, and several analysts have estimated the costs of implementing them. Manne and Richels (1992) estimate the lowest cost by which the United States could reduce emissions of carbon dioxide along the following path: stabilisation at 1990 levels up to the year 2000, then a gradual 20 per cent reduction over the ensuing decade, followed by a maintenance of that reduced level indefinitely. The marginal cost of emissions reduction eventually reaches about $208 per ton of carbon (1990 prices), which is more than twice the size of a carbon tax proposed but never implemented in the European Community. Updated to 1992 prices, this figure is equivalent to $0.67 per gallon of gasoline, or 3.1 cents per vehicle-mile for a car with US average fuel economy.[22]

Such a cost would approximately double the pollution cost of the average car in Los Angeles. Under the political scenarios underlying those hypothetical policies, global warming would become the dominant air-pollution cost element as newer vehicles meeting stricter pollution standards are incorporated into fleets. It seems likely that a better assessment of future damage from greenhouse gases will become increasingly important in determining the environmental damage from motor vehicles.

5. Conclusion

Environmental policy is frequently forced into tradeoffs between cleaner air and economic viability. We believe that the underlying basis for policy is clarified by making these tradeoffs explicit. Accordingly we have estimated the air pollution costs from motor vehicles, focusing mainly on the Los Angeles region. Our results probably give considerably higher estimates than would be obtained from areas lacking the mountain barriers that trap pollutants in the Los Angeles air basin.

Our estimates measure the health costs from particulate matter and ozone. Although other costs may be important, current evidence suggests that these constitute the bulk of the economic damage caused by air pollution from motor vehicles in developed nations like the United States.

What these estimates suggest is that poorly controlled vehicles have significant pollution costs. Costs are especially high for diesel cars and trucks, due to their high direct and indirect contributions to ambient concentrations of particles. For most vehicles typical of the 1990s, however, air pollution costs are not high enough for internalising them to alter car use very much. Our best estimate for the cost of the average car on the road in Los Angeles in 1992 is $0.03, falling to half that amount in the year 2000.

[22] Based on one ton carbon content for every 335 gallons of refined petroleum product, calculated from Manne and Richels (1992), p.59; and 1991 average fuel consumption of one gallon per 21.7 miles, from US Bureau of the Census (1993), Table 1037 Prices are deflated by the consumer price index for all items, from US Bureau of the Census (1993), Table 756

These findings make a strong *prima facie* case for policies that reduce the emissions from individual vehicles. Changes to both vehicles and fuel can offer substantial aggregate cost savings. But our findings do not provide much support for policies to reduce motor vehicle use overall. Such policies might be justified on other grounds, but with good technological measures in place they cannot be justified from the known costs of air pollution.

We do not address whether the particular pollution control policies now in place are good ones, a question cogently discussed by Calvert *et al.* (1993). The kinds of cost estimates derived here can contribute to the assessment of how stringent control policies should be.

Estimates of environmental costs are inherently uncertain. Among the many refinements that would improve ours are measuring the health effects from carbon monoxide, nitrogen dioxide and sulphur dioxide; including damage to vegetation and building materials; and providing more sophisticated modelling of the formation and transport of secondary pollutants. Probably the most important is clarifying the role of road dust.

Nevertheless, we believe that estimates such as ours put the burden of proof on those who argue that car use is greatly reducing the quality of life. It is not sufficient simply to point out that motor vehicles are polluting; unless researchers can find quantitative links to economic loss that are much stronger than those we have found, motor vehicle pollution seems best addressed by measures that reduce the emissions associated with a given amount of motor vehicle use. Our estimates provide one source of guidance as to which emissions it is most important to reduce.

Appendix
Glossary of Acronyms and Chemical Symbols

CO	Carbon monoxide
CO_2	Carbon dioxide
EPA	Environmental Protection Agency
FP	Fine particles
GDP	Gross Domestic Product
HC	Hydrocarbons
NMHC	Non-methane hydrocarbons
NMOG	Non-methane organic gases
NO_2	Nitrogen dioxide
NO_x	Nitrogen oxides
PM	Particulate matter
PM10	Particulate matter less than 10 microns in diameter
ROG	Reactive organic gases
SCAG	Southern California Association of Governments
SCAQMD	South Coast Air Quality Management District
SO_2	Sulphur dioxide
SO_4	Sulphates
SO_x	Sulphur oxides
TSP	Total suspended particulates
VOC	Volatile organic compounds

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