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METHANOL FUEL FOR LOS ANGELES AREA TRANSIT BUSES: COSTS AND BENEFITS

Final Report
to
The John Randolph Haynes and Dora Haynes Foundation

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EXECUTIVE SUMMARY

Methanol is widely regarded as a promising alternative to petroleum-based fuels in a wide variety of uses, including transportation. The primary advantages would be lower air pollution and diversification of energy sources. Interest is strong in California and especially in the Los Angeles air basin, leading to recently announced plans by the South Coast Air Quality Management District to begin converting large fleets of cars and buses to methanol.

Transit buses are an especially promising place to begin a strategy of using methanol as a transportation fuel. Their emissions are very visible and affect crowds of people, and the buses themselves are mostly operated in fleets by public agencies. In addition, federal emissions standards for heavy-duty diesel vehicles are especially strict for transit buses starting in 1991.

Our project focused primarily on the air pollution benefits of converting transit buses to methanol fuel. We used two different methods: cost-benefits analysis and cost-effectiveness analysis. Most of the analysis was carried out at one assumed price of diesel fuel and one or more assumed prices for methanol fuel, but we have also considered the mechanisms by which the two prices might be linked together.

Our cost-benefit analysis compares the cost of converting the fleet and using methanol fuel with the measurable health benefits from reducing particulates and sulfur oxides. In order to assess the benefits, we use statistically estimated relationships between those two pollutants and mortality rates in metropolitan areas across the United States, along with econometric evidence on the amount people are willing to pay to create small reductions in their risk of early mortality. The costs were assessed over a range of possible methanol prices. The results indicate that over a wide range of methanol prices, benefits exceed costs, even though many benefits are omitted in this somewhat narrow calculation.

The cost-effectiveness analyses add a number of features. Most importantly, the clean-air potential of methanol is compared with that of other strategies for reducing diesel emissions. The other strategies are low-aromatic fuel (a more expensive and highly refined diesel fuel), and particulate traps (a physical retrofit device that filters then burns particles in the exhaust). In addition, the comparisons are made for three different pollution indices, each combining the effects of particulates and sulfur oxides in a different way. In a further extension, one of these indices is used to add nitrogen dioxide and ozone to the ambient pollutants taken into account.

In all the comparisons, methanol conversion has a higher cost per unit reduction in pollution than the other strategies, at least at foreseeable fuel prices. However, it also reduces pollution by more than the other strategies, so one needs also to consider the incremental cost of the additional reduction it brings about. That can be done only with one of the three indices, that based on the same mortality analysis as the cost-benefit analysis; with this methodology, it appears that the additional reduction in particulate and sulfate levels caused by using methanol instead of particulate traps would reduce cancer deaths at a cost of $1.6 million per statistically expected death, well within the range of values that people appear willing to pay in labor markets in order to reduce their risks. This calculation assumes a methanol price about 44 cents higher than diesel, per amount of energy contained in a gallon of diesel fuel.

The comparisons just mentioned do not take into account the reduced emissions of nitrogen oxides and reactive organic gases that methanol would bring about. We developed a procedure to include these in one of the other indices, based on California's ambient air standards. The somewhat surprising results are, first, that ozone reductions are approximately offset by ozone increases in the coastal areas of the basin, so there is little or no net benefit from ozone; but second, that there is a substantial benefit from the reduction in nitrogen dioxide.

Both of these results are somewhat tentative due to ongoing controversies over the basic scientific facts. First, the offsetting ozone increases are due to a phenomenon known as "ozone scavenging," in which nitric oxide emissions react with ozone in the immediate vicinity; while there is no doubt that this occurs, its quantitative importance is still debated and under intense investigation. Second, the extent of short-term health effects of nitrogen dioxide are still uncertain.

Finally, we considered the plausibility of scenarios in which methanol prices stay as close to diesel prices as they now are. The methanol industry is now depressed with considerable excess capacity, so any major increase in demand for methanol as a fuel would cause its price to rise to one determined by long-run supply costs. To analyize this, we consider the substitution possibilities among residual oil (produced from petroleum), compressed and liquified natural gas (produced from both domestic and foreign natural gas), and methanol (large quantitities of which would most likely be produced from natural gas at sites remote from industrial users). This raises real doubt that methanol price would remain at such a small differential from diesel as now exists if methanol were widely adopted as a fuel. With present information, a precise quantitative analysis is not possible.

The same analysis of energy supply, however, suggests that the ability to rapidly convert a substantial amount of the nation's transportation fuel use to methanol could itself serve as an important deterrent to a repeat of the cartel-induced oil price increases of the 1970s. It might therefore be in the national interest to subsidize the development of equipment, infrastructure, and experience for using methanol even if it does not appear economical as an clean air strategy. In any such decision, the Los Angeles area would be the obvious candidate for a trial. Furthermore, the precise extent of the air pollution benefits themselves is a key consideration, since they could be regarded as an offset to the extra costs of developing methanol as an an energy security policy. Hence the quantitative analyses described here would be invaluable information for policy decisions on such a strategy.

RESULTS OF PROJECT

This project is an extended study of the benefits and costs of converting transit buses in the Los Angeles basin to methanol fuel. Our focus is on using cost-effectiveness analysis to compare alternative policies for reducing the emissions from these diesel-powered buses.

One reason for interest in this topic is that transit buses would be a good test ground for any wider strategy for fuel conversion. Indeed, the South Coast Air Quality Management District (SCAQMD) has declared its intention to convert significant portions of the Los Angeles basin's vehicle fleets to methanol, and recent federal emissions standards for heavy-duty vehicles select transit buses for especially stringent treatment. There are many reasons for selecting transit buses for early conversion: for example, emissions are in crowded areas at street level, the buses are operated in fleets with central maintenance and fueling facilities, and air pollution officials receive a disproportionate number of complaints about diesel buses.

The primary results of the project appear in two technical papers which are being published in professional journals read by researchers and policymakers specializing in transportation. Another paper (the first chronologically) was mostly written prior to the current grant, and provides valuable background information on the costs and benefits of the methanol conversion strategy; its final revision and publication were carried out with the help of the grant. These papers, two of which are coauthored with the research assistants on this project, are attached as part of this report. We discuss each in turn.

<u>Converting Transit to Methanol: Costs and Benefits for California's South Coast Air Basin</u>, by Stephenie J. Frederick, Jane L.C. Morrison, and Kenneth A. Small. (Published by Transportation Research Board in <u>Transportation Research Record</u>, No. 1155, 1987, pp. 12-17).

Methanol offers much promise as an alternative fuel whose combustion produces no sulfates and far fewer particulates and nitrogen oxides than diesel fuel. The benefits from replacing diesel fuel by methanol are estimated from epidemiologic studies of the relation between sulfate and particulate concentrations on the one hand and mortality on the other. These are put in dollar terms by using studies of wage differentials for risky jobs. Costs include the additional cost of the bus itself and the extra cost of the fuel. The comparison of benefits with costs depends greatly upon future methanol prices, which are highly uncertain. Over a considerable range of such prices, however, we find that benefits exceed costs, even though many benefits, including any effects on ozone, are omitted.

We conclude in this paper that the conversion strategy is promising and deserves a careful comparison with alternative strategies such as using improved diesel fuel and equipping busses with particulate traps which reduce particulate emissions. It is just such a comparison that forms the topic of the next two papers.

<u>Reducing Transit Bus Emissions: Comparative Costs and Benefits of Methanol, Particulate Traps, and Fuel Modification</u>, by Kenneth A. Small. (Published by Transportation Research Board in <u>Transportation Research Record</u>, No. 1164, 1988, pp. 15-21.)

The cost-effectiveness analysis in this paper is limited to particulates and sulfur oxides. Three alternative methods of measuring "effectiveness" are considered. One is simply an estimate of the total particulates produced including those formed in the atmosphere from emitted sulfur dioxide. A second uses California's ambient air quality standards as indicators of the relative severity of the pollutants. The third is based on the same statistically estimated effect on mortality as was used in the first paper.

These indices are used to compare four possible control strategies. Each strategy is examined for the improvements in air quality vis-a-vis a base case in which diesel buses are operated as now. The first strategy is requiring fuel which is low in aromatic content, aromatics being chemical components with benzene rings which produce higher emissions. (This is a strategy now under active consideration, and should not be confused with low-sulfur fuel, already required in the Los Angeles basin hence already assumed in our base case). The second is particulate traps, also known as trap-oxidizers, a type of retrofit device now under development and being considered for meeting the 1991 federal emissions requirements on transit buses. The third is low-aromatic fuel combined with particulate traps. The fourth is methanol fuel.

At fuel prices considered most likely, methanol is far more costly than the other strategies per unit reduction in total particulates. But this disadvantage is much less according to the other indices, which give more weight to the sulfur oxide reductions. In addition, methanol achieves a greater total emissions reduction than the other strategies, so one really has to consider the incremental cost-effectiveness of the extra reduction. With the mortaility-based index, the incremental cost of the methanol strategy over that of particulate traps comes to $1.6 million per incremental reduction in number of statistically expected deaths, a value that makes it a serious contender as a policy despite its higher cost.

Cost-Effectiveness of Emissions Control Strategies for Transit Buses: The Role of Photochemical Pollutants, by Kenneth A. Small and Stephenie J. Frederick. (To be published in Transportation Research, 1989.)

This paper tackles the difficult problem of incorporating ozone and other photochemical products of air-pollution chemistry into cost-benefit or cost-effectiveness analyses. We single out two such photochemical products, for which specific air-quality standards are set in California: ozone (O_3) and nitrogen dioxide (NO_2). This topic requires separate treatment because the way ozone forms is fundamentally different from other ambient air pollutants, which can generally be linked closely with a specific emission. Ozone results primarily from the emissions of nitrogen oxides (NO_x) and reactive organic gases (the latter are sometimes called hydrocarbons although they include other compounds). The chemical interactions of these primary emissions are complex and depend on cumulative effects of several factors including sunlight and wind. Mathematical models specific to the Los Angeles basin have been under development for well over a decade, yet there is still disagreement over such fundamental questions as whether reducing NO_x emissions would make ozone better or worse.

We use the results of one such model, developed by Systems Applications, Incorporated (SAI), to estimate the effects of our control strategies on aggregate exposures of the basin's population to ozone and nitrogen dioxide. The control strategies analyzed are the same as in the previous paper, and they are analyzed using the second of the three indices described there.

The somewhat surprising result is that aggregate ozone exposures are affected very little by any of our strategies. There are two reasons. First, the SAI model predicts that decreased NO_x emissions will lower ozone levels in the inland (downwind) regions of the basin but raise them in the coastal regions, with a net effect on population exposures that nearly balances to zero. Second, reactive organic emissions from diesels are so small that reducing them makes very little difference. As already noted, the modeling of ozone is still subject to considerable scientific debate so our finding is tentative.

The other surprising result is that NO_2 exposures are affected enough to significantly modify the cost-effectiveness comparisons. This presumes that California's air-quality standards for NO_2, which (in contrast with federal standards) include a limit on short-term exposure, are based on sound evidence for short-term health effects. We believe this to be the case, but there is some controversy. The result of including NO_2 in our cost-effectiveness measure is to substantially improve the relative cost-effectiveness of low-aromatic fuel and, even more, of methanol, in comparison to particulate traps.

Energy Economics and the Cost of Methanol Fuel.

All cost-benefit and cost-effectiveness analyses involving methanol are very sensitive to the cost of the fuel itself relative to diesel fuel. Like most analysts, we have tried to project the most likely methanol prices from current information, then discuss the effects if it turns out to be cheaper or more expensive. However, a more sophisticated analysis should take into account the interdependence of methanol and crude oil prices.

Professor Linda Cohen has analyzed this process qualitatively as part of the project. The main avenue of interdependence is through natural gas. If crude oil should rise in price, many utilities and industrial users would switch to natural gas, thereby driving up its price. At a high enough oil price, many vehicle fleets would also start to use compressed natural gas, adding further to the pressure on natural gas price. By measuring the cost of shipping natural gas from remote sites such as Saudi Arabia or Indonesia to U.S. domestic users (an expensive process involving liquification and substantial losses from evaporation during transit), it is possible to determine how responsive the price of remote natural gas would be to such developments. Since remote natural gas is thought to be the primary feedstock for methanol plants under scenarios of significant use of methanol as a transportation fuel, this ultimately would cause methanol price to rise as well.

This process could be extremely important, because it might imply that some of the price scenarios in which methanol looks attractive are incompatible with its large-scale use. This would not be much of a factor if one is considering just transit buses in the Los Angeles basin, but it might be if one considers expanding such a pollution-control strategy to other locations or to other vehicles such as trucks and cars. Hence we believe the links between crude oil and methanol prices could well undermine the case for pollution-control strategies involving widespread use of methanol fuel.

At the same time, the very same substitutability can be used to make a case, based upon energy policy, for government programs aimed at creating the equipment and supply infrastructure needed for methanol use. The argument is that having in place a credible capability to quickly expand methanol use would limit the monopoly power of oil-producing nations and prevent them from raising the price of crude oil as much as they might otherwise. The irony is that this strategy is most successful if widespread methanol adoption remains only a threat, in which case the methanol development already undertaken subsequently looks like a bad investment; yet that development produces enormous social benefits to the nation by lowering the price of imported crude oil. Clearly such a strategy requires federal intervention and will not be undertaken either by private companies or by individual localities, since they would reap only a small portion of the benefits.

We believe that working out the quantitative nature of these linkages is a high priority for research into policy toward alternative fuels. A draft paper by professor Cohen explaining the linkages, which may later become the backbone of longer paper providing such a quantitative analysis, is included as an attachment to this report.

APPENDIX:

RESEARCH PAPERS CONTAINING RESULTS OF PROJECT

Converting Transit to Methanol: Costs and Benefits for California's South Coast Air Basin, by Stephenie J. Frederick, Jane L.C. Morrison, and Kenneth A. Small. (Published by Transportation Research Board in Transportation Research Record, No. 1155, 1987, pp. 12-17).

Converting Transit to Methanol: Costs and Benefits for California's South Coast Air Basin

STEPHENIE J. FREDERICK, JANE L. C. MORRISON, AND KENNETH A. SMALL

Methanol offers much promise as an alternative fuel whose combustion produces no sulfates and fewer nitrogen oxides and particulates than diesel fuel. Another advantage is that large quantities could be manufactured from domestic coal supplies. On the basis of the assumption that an extensive methanol program might well begin with public transit, the costs and benefits of converting the bus fleets of California's South Coast Air Basin to methanol are estimated. Benefits are based on the reduced mortality attributable to lower sulfates and particulates; costs encompass both bus conversion and replacement. When these benefits are compared with costs over a wide range of methanol prices, conversion to methanol is found to merit further consideration as an antipollution strategy. It is proposed that the analysis be extended to additional potential benefits and costs and to other locales and types of vehicles.

Replacing petroleum-based fuels with methanol has been suggested as a promising way to improve air quality and reduce dependence on imported oil. Methanol burns more cleanly and has greater supply flexibility because it can be made from natural gas, coal, or even biomass. Because current technology would allow a fairly easy conversion, the idea has found support among government agencies and environmental groups as well as the energy and transportation industries.

Unlike diesel fuel or gasoline, methanol is an alcohol. Its cooler flame produces fewer nitric oxide emissions and so reduces concentrations of derived pollutants such as nitrogen oxides, nitric acid, ozone, and other oxidants. Particulate emissions, a serious problem with diesel engines, are almost eliminated. Because all sulfur content is removed during manufacture, methanol produces no sulfur dioxide and therefore no sulfuric acid, a principal component of acid rain.

The last decade has witnessed extensive investigation of engine design, emissions content, materials compatibility, and methanol production methods. Test vehicles operate at several sites in California, and additional projects are planned or starting up in Jacksonville, Seattle, and New York. Yet there have been few economic evaluations of methanol conversion, and these few have been contradictory or incomplete. The California Institute of Technology's Jet Propulsion Laboratory *(1)* concludes that methanol's market penetration will proceed very slowly, that it can reduce air pollution levels only slightly, and that methanol prices will rise substantially as demand and reliance on domestic feedstocks increase. Gray and Alson *(2)* are far more optimistic, suggesting that nationwide vehicle usage of methanol made from high-sulfur coal would improve air quality, revive eastern coal-mining areas, and reduce U.S. dependence on foreign oil.

However, none of these studies attempts to quantify the benefits in economic terms. The question of whether the benefits of methanol use outweigh its costs has been left to somewhat subjective judgment. To further the economic evaluation of conversion policies, a simple cost-benefit analysis is therefore developed and presented. To make it as clear as possible, it is restricted to a very limited but promising case: methanol conversion of public transit buses in California's South Coast Air Basin. This allows a demonstration, in the simplest possible way, of the kinds of information and assumptions required to compare benefits and costs. At the same time, a case is chosen that ought to highlight the advantages of methanol and provide a first test of whether analysis of more complex policies is warranted.

The South Coast Air Basin, hereafter referred to as "the Basin," includes the urbanized parts of Los Angeles, Orange, San Bernardino, and Riverside Counties in California. The Basin makes a particularly interesting case study because of its national stature as a pollution center; the reason being that if methanol use could not provide significant benefits in this heavily populated and polluted region, it would be unlikely to provide them elsewhere.

Transit buses provide an ideal technology for a first case study: the vehicles are homogeneous, concentrated at a few public enterprises that keep good records, and fueled and maintained at a few central facilities. These same factors also facilitate the methanol conversion process; in addition, buses are an obvious target because they are highly visible polluters that operate in populous areas and emit exhaust directly at street level. A policy designed to abate air pollution might do well to begin with those vehicles that transgress most in the eyes of the public.

The benefits accruing only from a reduction in the mortality rate are estimated. Air pollution, of course, causes many other kinds of harm: it increases nonfatal illness, burns eyes and lungs, soils and damages materials, blights crops, and reduces visibility. There are two reasons for limiting the benefits considered here. First, in this initial analysis, only the most critical policy issues are addressed. Second, several careful empirical studies have established the pernicious effects of air pollution on health and have provided functional relationships that may be used in benefit-cost analysis.

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In addition, only two pollutants are examined: total suspended particulates (TSP) and sulfur oxides (SO_x). These pollutants can be traced reasonably well from tailpipe to lungs, their health effects are known, and their emissions are virtually eliminated in methanol-fueled engines. Reduction of nitrogen oxides (NO_x) may be an equally important feature of methanol buses, but NO_x health effects occur through a complicated path of photochemical changes in the atmosphere that is more difficult to trace.

For simplicity, the authors analyzed a steady state in which all buses are fueled with methanol, one-twelfth being replaced each year because of normal attrition, and in which population, bus mileage, and value of pollution reduction remain constant. Of course, many things would change over time. Most of these would make methanol conversion more favorable. Increased population and higher incomes would increase the benefits, whereas improved technology will almost certainly reduce the extra costs of equipping buses for methanol use. The authors refrained from speculation on future fuel price differentials. The methodology makes no attempt to address transition problems with methanol conversion or to compare it with alternative ways of reducing emissions either now or in the future.

The analysis, then, chooses a particularly favorable case for methanol but analyzes it conservatively. Because the results show benefits exceeding costs over a significant range of assumptions and fuel costs, conversion of transit buses in Southern California appears to be a promising public policy. Also, analysis of other conversion strategies involving other vehicles and other metropolitan areas is warranted. The methodology presented here provides a sound basis for extending the analysis to such cases and for refining it to include additional types of benefits.

TABLE 1 REDUCTIONS IN AMBIENT AIR CONCENTRATIONS OF PARTICULATES AND SULFATES AS A RESULT OF METHANOL USE

Type of Bus	Per-Vehicle Emissions (grams/mi)[a]	Total Annual Emissions (000s kg)[b]	Percent Reduction in Ambient-Air Concentration Compared to Diesel[c]
Particulates			
Diesel	6.275	948.77	
Methanol (M.A.N.)	0.0644	9.74	0.430%
Methanol (GM)	0.6275	94.88	NA[d]
Sulfur Oxides			
Diesel	0.81	122.5	
Methanol (M.A.N.)	0	0	0.226%
Methanol (GM)	0	0	NA

[a]Particulate emissions are from Ullman et al. *(19)*; Grade 2 diesel fuel assumed in diesel engine. SO_x emissions are derived from the sulfur content of the fuel used, which is taken to be 0.05 percent by weight, the maximum now permitted by the state of California for buses in the Basin. Fuel density is 7.163 lb/gal; fuel consumption is 1 gal/4 mi; and sulfur oxide molecules contain 50 percent sulfur by weight, as is the case for SO_2. (Details are presented in an appendix available from the authors.)
[b]Per-vehicle emissions [a] x total annual vehicle miles in 1984 [151.2 million *(20)*].
[c]Total annual emissions (diesel buses) minus total annual emissions (methanol buses) result divided by the total annual emissions from all sources in 1983 *(21)*, which is 218.6×10^6 kg for particulates and 54.1×10^6 for sulfur oxides.
[d]GM data are not used in the analysis because of the comparatively poor performance of the GM methanol bus, which is a preliminary prototype. In the testing performed by Ullman et al. *(19)*, the GM's SO_x emissions and a large portion of its particulate emissions were apparently caused by engine oil scavenged into the exhaust.

DATA AND METHODOLOGY

Pollution Reduction

The first step in the analysis is to establish the percentage reductions in ambient air TSP and SO_x concentrations attributable to conversion to methanol fuel. This requires knowing the emissions per mile of each type of bus, the total annual miles traveled by transit buses in the Basin, and the total emissions from all sources in the Basin. The results are given in Table 1. Because buses account for only a tiny fraction of emissions in the Basin, conversion would reduce ambient air concentrations by a minuscule 0.43 percent of TSP and 0.226 percent of sulfates.

Mortality Reduction

The second step is to establish the effect on the mortality rate of a unit decrease in the level of each pollutant. The effect of these pollutants has been established by the detailed regression analysis of Lave and Seskin *(3)* and Chappie and Lave *(4)* who used mortality and pollution data from more than 100 U.S. metropolitan areas, and by numerous epidemiological studies, reviewed and extended by Ozkaynak and Spengler *(5)*. The latter authors conclude that as much as 6 percent of the mortality in urban areas can be attributed to particulates and to sulfates, a derivative of sulfur oxides (*5*, p. 54).

The precise relationship between emissions and ambient concentrations of particulates and sulfates is not one to one (though it is far more straightforward than for nitrogen oxides and ozone, which is one reason for omission of the latter here). In the case of particulates, recent evidence suggests that it is mainly fine particles that cause health damage *(5)*, whereas the data used by Lave and Seskin do not distinguish by particle size. Because a high proportion of the particulates emitted by diesels are fine, their harmful effects are probably underestimated by ignoring that feature. This belief is supported by a replication of the Lave and Seskin work for a more recent year, which shows that where fine particles are a smaller proportion of all particulates, a weaker relationship exists between particulates and mortality.

In the case of sulfur oxides, most of these emissions are transformed into sulfates through atmospheric reactions. The common assumption is that atmospheric sulfate concentrations are proportional to sulfur oxide emissions. This assumption has some support from atmospheric simulation models, at least in the case of the clear weather that characterizes Southern California *(6)*. Note that even though sulfates are a component of particulates, they can be treated separately without double counting because they are also treated as separate pollutants in Chappie and Lave's statistical work.

The most comprehensive estimates of the quantitative relationship are those by Chappie and Lave *(4)*. Their work remains the most careful and complete study of the effects of air pollution on mortality in actual urban populations and includes data from 1960, 1969, and 1974.

For each pollutant, the three estimated elasticities of mortality with respect to concentration were averaged, one for each of the three years (*4*, p. 349). This average was then adjusted downward by 0.0303 (sulfate elasticity) and 0.0234 (particulate elasticity) on the basis of the difference, in the 1974 results, caused by adding a socioeconomic variable that was unavailable in the earlier years' data (*4*, p. 352). The assumption is that including that variable in the earlier years would have made the same difference in the results for those years. (Further details are provided in an appendix available from the authors.) This procedure is conservative in that without this adjustment, the sulfate and particulate elasticities would be 61 and 197 percent higher, respectively. Alternatively, if the best regression estimates from the 1974 data were used, ignoring the earlier years, the sulfate elasticity would be about twice as high, and the particulate elasticity would vanish, with a slight overall increase in the benefits estimated in the next sections.

The resulting changes in mortality rates and total mortality are given in Table 2.

TABLE 2 REDUCTION IN MORTALITY DUE TO METHANOL CONVERSION

Pollutant	Elasticity of Mortality with Respect to Ambient Air Concentrations[a]	Reduction in Total Mortality Rate (annual deaths per million)[b]	Reduction in Annual Deaths in Los Angeles Basin[c]
Particulates	0.0119	0.41	4.36
Sulfates	0.0500	0.91	9.63
Total	0	1.32	13.99

[a]Percentage change in total mortality rate, divided by percentage change in ambient air pollutant concentration (see text for sources).
[b]Elasticity times pollutant reduction from Table 1, times total mortality rate in South Coast Air Basin (8,025 per million, computed from data provided by the Departments of Public Health of Los Angeles, Orange, San Bernardino, and Riverside counties).
[c]Reduction in total mortality rate times population of Los Angeles Basin (10.62 million).

Value of Mortality Reduction

The third step is to express in dollars the benefits from reducing the mortality rate. This requires multiplying the reduced mortality rate by a dollar value assigned to the reduction in risk of death. The assignment of this explicit value is crucial because it allows the quantification of benefits; hence it is necessary to digress to present the conceptual basis with some care.

Many studies have stumbled on the apparent paradoxes inherent in placing a dollar value on policies that save lives. Discounted value of lifetime earnings has often been used, despite the obvious defects that most earnings are for the person's own consumption and that this measure places no value on the lives of retired people.

Here the now widely accepted concept of willingness to pay is followed: How much do people pay to reduce hazards, or how much extra compensation do they demand for working under hazardous conditions *(7–9)*? Rather than ask the value of saving an identifiable person's life, we ask the value of reducing the ongoing risk of fatality that everyone faces. This is more consonant with the way in which policies actually affect people because most policies, including those concerned with air pollution control, make very small changes in the mortality risk facing large numbers of people.

For example, suppose that a clean air policy reduced everyone's annual risk of dying from 1 in 100 to 0.99 in 100. How much would the average person be willing to pay for such a change? This is an answerable question, because people can be observed making choices involving risk changes of this magnitude, such as purchasing safety equipment or choosing among jobs involving various degrees of hazard. [In fact, changing jobs from one of average occupational risk to one of no occupational risk involves a reduction of about this amount (.01 in 100).] If such observed behavior indicates that people are willing to pay $800 per year for this reduction (or to forego wages of that amount), then the willingness to pay for a reduction in risk from 0.0100 to 0.0099 is $800.

In a community of 10,000 people, such a risk-reduction policy lowers the expected annual death rate from 100 to 99. It could be stated, somewhat loosely, that it saves one life per year. Because in the aggregate these people are willing to pay 10,000 × $800 = $8 million/year for the risk reduction, it could be said that the "value of life is $8 million." This is just shorthand, however, for the more precise earlier statement. It does not mean that Sara Jones's life is worth $8 million; it means that 10,000 people are willing to pay $800 each for a reduction in risk that, in aggregate, will probably save one life.

Kahn *(10)* discusses the methodological weaknesses and strengths of some of the best-known attempts to estimate people's willingness to pay for risk reduction. She presents a strong case for relying on the estimates derived from labor market analyses. For example, estimates based on markets for safety equipment have ignored the inconvenience associated with installation, maintenance, and use of the safety devices.

Kahn also presents a comprehensive analysis of sources of bias in the labor market studies and thereby offers a convincing basis for choosing estimates by Olson *(11)* and Viscusi *(12, 13)* that are among the highest of the various studies. Kahn in particular advocates using the "value of life" obtained by Olson for a combined sample of union and nonunion workers, which is $8 million in 1984 dollars. The subsequent and widely cited work by Viscusi *(14)* also results in estimates of comparable magnitude. Nevertheless, current practice in government analyses of safety practices uses much lower values, typically $0.5 to $1.5 million, resulting from the earlier studies and from the method of present discounted value of lifetime earnings. In this analysis both figures, $1.5 and $8 million, are used to test the sensitivity of the results. At the higher of these figures, the mortality reduction given in Table 2 is valued at $113 million annually, of which 69 percent results from reduced sulfates and the remainder from reduced particulates.

Implicit in this calculation is a value per kilogram of emissions removed for each pollutant, obtained by valuing the reduced deaths given in Table 2 (last column) at this value and dividing by the corresponding emissions reductions given in Table 1 (middle column). At the higher value of mortality reduction, each kilogram of particulates or sulfur oxides emitted costs society $37 or $629, respectively—startling figures considering that a typical diesel bus emits a kilogram of sulfur oxides in about 2 weeks (1,370 mi) and of particulates in less than 2 days (159 mi).

Costs

The fourth step is to calculate the costs of the methanol strategy. There are two main costs: a capital expenditure for conversion and an operating expenditure for fuel.

Building methanol buses is relatively expensive because they are manufactured in small quantities. For example, Seattle Transit paid $175,000 each for 10 methanol buses while paying only $126,000 each for new diesels. General Motors, however, in testimony to Congress in 1984, indicated that annual production of 250 to 300 methanol buses could bring the cost differential down to between $6,000 and $7,000 (*2*, p. 125). This appears to be a more pertinent estimate for this study. This estimate is also more consistent with the evidence from Florida's retrofitting experiment in which the Florida Department of Transportation estimated the actual cost of converting an existing bus, once substantial scale is attained, at $7,500 to $10,000 (*15*, p. 73). However, to accommodate both possibilities and to remain conservative, a range of $6,500 to $49,000 is adopted here as the additional cost of replacing a diesel with a methanol bus. In estimating the average life of a transit bus at 12 years, it is assumed that one-twelfth of the vehicles in the Basin fleet will be replaced annually. Multiplying this number (369) by $6,500 to $49,000 gives a range of the annual additional capital cost of purchasing methanol rather than diesel buses (Table 3).

TABLE 3 REPLACING DIESEL WITH METHANOL BUSES: ANNUAL ADDITIONAL COST

Additional Cost per Bus Replaced ($)	Average Bus Lifetime (years)	Total Annual Additional Cost[a] ($ millions)
6,500[b]	12[c]	2.40
49,000[d]	12	18.08

[a]Additional cost per bus multiplied by total number of transit buses in the South Coast Air Basin (4,432), result divided by average life of transit bus.
[b]Gray and Alson (*2*, p. 125).
[c]Wachs and Levine *(20)*.
[d]Based on actual prices paid by Metro Transit, Seattle, Washington, in 1986.

The instability of the world oil market implies instability in the price of diesel fuel, increasing or diminishing its present price advantage over methanol. The current price of methanol reflects a worldwide oversupply, but a substantial increase in demand could drive the price up. In light of these uncertainties, the results of this analysis are presented as a function of price differentials between diesel and methanol fuels.

It is convenient and common to state fuel prices on the basis of equivalent energy content rather than equivalent volume. A gallon of methanol contains fewer Btu (57,000) than a gallon of diesel (128,000), and so the price per gallon of methanol is multiplied by 128,000/57,000 to obtain a price per 128,000 Btu of fuel. No further adjustment is required because the fuel efficiencies of methanol and diesel engines are comparable *(16)*. The total annual fuel cost differential is found by multiplying the price differential computed in this way by the annual number of gallons of diesel fuel currently burned by all of the transit buses in the Basin (37.8 million).

It should be noted that some costs are neglected in the analysis. Because methanol is toxic, burns with an invisible flame, and produces harmful vapors, there may be an additional cost to handle it safely. In addition, because of the discrepancy in energy content, buses will require twice as many gallons of methanol as diesel, which will increase the costs of refueling and storage (costs of larger fuel tanks on the buses themselves are already taken into account). However, these and similar costs appear to be relatively small.

RESULTS

The results of the analysis are shown in Figure 1 as functions of the excess of methanol price over diesel price. There are two alternative assumptions on value of life ($8 million and $1.5 million), leading to two alternative estimates of benefits, shown as horizontal lines. There are two alternative assumptions on differential bus acquisition cost ($6,500 and $49,000), leading to two alternative estimates of costs, shown as sloped lines. Costs, of course, rise as the methanol price increases relative to the diesel price.

It is clear that the alternative assumptions shown make a great deal of difference to the conclusion. The authors have argued that the higher value-of-life estimate ($8 million) and the lower capital cost estimate ($6,500) are the more accurate ones. If that is true, benefits exceed costs even when methanol prices (per energy content of a gallon of diesel) are as much as $2.93 higher than diesel. Over the past year, the average price differential has been $1.00, at which point benefits exceed costs by a ratio of three to one.

On the other hand, comparison at the lower estimate of value of life is not as favorable. Only if the price difference drops to $0.50 do benefits outweigh costs, assuming General Motors' estimate of $6,500 as the extra cost of building a methanol-fueled bus. Many possible benefits of methanol have been omitted; for example, methanol use in buses would reduce NO_x emissions as well as weaken the impact of direct street-level exhaust. Also omitted are the advantages of improved visibility and lessened morbidity, soiling, and materials and crop damage. All these benefits must be taken into account in deciding whether a policy of methanol conversion would still be worthwhile, given the less favorable assumptions on the value of mortality reduction.

CONCLUSION

This first try at a cost-benefit analysis of a methanol conversion strategy leads to several tentative conclusions. On the substantive side, there is real promise for a policy of converting transit buses in the Los Angeles basin. Given recent evidence about

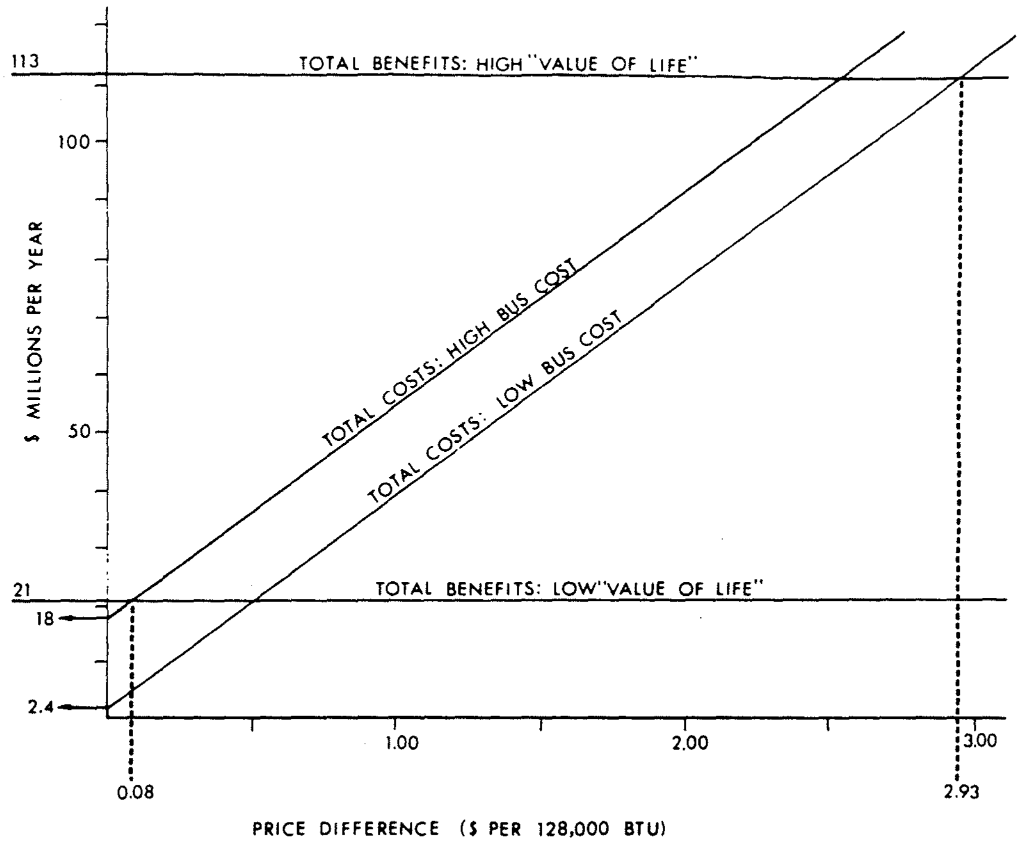

FIGURE 1 Benefit-cost analysis in terms of methanol-diesel price difference.

people's willingness to pay for lower mortality risk, the policy is justified over a wide range of methanol prices. When the older estimates of value of life are used, the case is not as clear cut. Both evaluations are quite conservative, however, because the analysis was limited to the negative effects of only two pollutants—sulfates and particulates—and examined only one positive effect—the change in mortality.

In terms of a research agenda, three sources of uncertainty need further work. One is the effect of methanol use on other pollutants, particularly photochemical oxidants. These are compounds often believed to cause the worst problem in the South Coast Air Basin; therefore, a careful analysis of the potential for reducing them through lessened nitric oxide emissions might show considerable benefits. The second is the possible existence of important benefits from reduced sickness, reduced materials and crop damage, and improved visibility. The third is the question of whether the same benefits can be attained in other ways such as by using diesel fuel with less sulfur and aromatic hydrocarbons or by fitting buses with particulate traps and catalytic converters.

The work of Weaver and his colleagues *(17, 18)* suggests that starting with diesel fuel typical of that used in the United States and adopting a low-sulfur and low-aromatic fuel (similar to that taken as the baseline in this analysis and already required in the Los Angeles basin) is the most cost-effective means of reducing particulate emissions. They also suggest that in terms of the incremental cost of making further particulate reductions, particulate traps compare favorably with methanol. An extension of the methodology described here could provide further evidence on the comparative merits of these strategies, taking into account more pollutants than did Weaver.

A deeper policy question underlying this analysis of transit buses is the benefits that might be achieved from a wider methanol conversion strategy, including cars, trucks, and perhaps stationary sources as well. The answer cannot be confidently predicted. Whether the favorable case for methanol extends to other types of vehicles or other locations is likely to depend critically on extensions of the research methodology.

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REFERENCES

1. R. O'Toole et al. *California Methanol Assessment, Vol. 1, Summary Report.* JPL Publication 83-18. Division of Chemistry and Chemical Engineering, Pasadena, Jet Propulsion Laboratory and California Institute of Technology, March 1983.
2. C. L. Gray, Jr., and J. A. Alson. *Moving America to Methanol.* University of Michigan Press, Ann Arbor, 1985.
3. L. B. Lave and E. P. Seskin. *Air Pollution and Human Health.* The Johns Hopkins University Press, Baltimore, Md., 1977.
4. M. Chappie and L. B. Lave. The Health Effects of Air Pollution: A Reanalysis. *Journal of Urban Economics,* Vol. 12, 1982, pp. 346–376.

5. H. Ozkaynak and D. Spengler. Analysis of Health Effects Resulting from Population Exposures to Acid Precipitation. *Environmental Health Perspectives*, Vol. 63, 1985, pp. 45–55.
6. C. Seigneur, P. Saxena, and P. M. Roth. Computer Simulations of the Atmospheric Chemistry of Sulfate and Nitrate Formation. *Science*, Vol. 225, Sept. 7, 1984, pp. 1028–1030.
7. E. J. Mishan. Evaluation of Life and Limb: A Theoretical Approach. *Journal of Political Economy*, Vol. 79, 1971, pp. 687–705.
8. R. Thaler and S. Rosen. The Value of Saving a Life: Evidence from the Labor Market. N. J. Terleckyj, ed. *Household Production and Consumption*. Columbia University Press, New York, 1975, pp. 265–298.
9. A. Marin and G. Psacharopoulos. The Reward for Risk in the Labor Market: Evidence from the United Kingdom and a Reconciliation with Other Studies. *Journal of Political Economy*, Vol. 90, No. 4, 1982, pp. 827–853.
10. S. Kahn. Economic Estimates of the Value of Life. *IEEE Technology and Society Magazine*, Vol. 5, June 1986, pp. 24–31.
11. C. Olson. An Analysis of Wage Differentials Received by Workers on Dangerous Jobs. *Journal of Human Resources*, Vol. 16, 1981, pp. 167–185.
12. W. K. Viscusi. *Employment Hazards: An Investigation of Market Performance*. Harvard University Press, Cambridge, Mass., 1979.
13. W. K. Viscusi. Unions, Labor Market Structure, and the Welfare Implications of the Quality of Work. *Journal of Labor Research*, Vol. 1, Spring 1980, pp. 175–192.
14. W. K. Viscusi. *Risk by Choice*. Harvard University Press, Cambridge, Mass., 1983.
15. *Alternative Fuels: Status of Methanol Vehicle Development*. Briefing Report to the Chairman, Subcommittee on Fossil and Synthetic Fuels, Committee on Energy and Commerce, U.S. House of Representatives. Report GAO/RCED-87-10BR. U.S. General Accounting Office, Oct. 1986.
16. *Preliminary Perspective on Pure Methanol Fuel for Transportation*. Report EPA 460/3-83-003. Office of Mobile Sources, Emission Control Technology Division, U.S. Environmental Protection Agency, Sept. 1982.
17. C. S. Weaver, R. J. Klausmeir, and L. M. Erickson. *Feasibility of Retrofit Technologies for Diesel Emissions Control*. SAE Technical Paper Series 860296. Society of Automotive Engineers, Warrendale, Pa., 1986.
18. C. S. Weaver, C. Miller, W. A. Johnson, and T. S. Higgins. *Reducing the Sulfur and Aromatic Content of Diesel Fuel: Costs, Benefits, and Effectiveness for Emissions Control*. SAE Technical Paper Series 860622, Society of Automotive Engineers, Warrendale, Pa., 1986.
19. T. L. Ullman, C. T. Hare, and T. M. Baines. *Emissions from Two Methanol-Powered Buses*. SAE Technical Paper Series 860305. Society of Automotive Engineers, 1986.
20. M. Wachs and N. Levine. *Vehicle Fleets in the South Coast Air Basin*. Survey performed for the South Coast Air Quality Management District, University of California, Los Angeles, 1985.
21. South Coast Air Quality Management District. *Draft 1983 Emissions Inventory, South Coast Air Basin*. Working Paper 1. 1987 Air Quality Management District revision. El Monte, California, May 1986.

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Reducing Transit Bus Emissions: Comparative Costs and Benefits of Methanol, Particulate Traps, and Fuel Modification

KENNETH A. SMALL

The cost-effectiveness of three strategies for reducing particulate and sulfur oxide emissions from diesel transit buses is investigated. The strategies, in order of increasing effectiveness, involve low-aromatic fuel, particulate traps, and methanol fuel. All three are evaluated under optimistic assumptions. Three alternate indices of emissions are considered: one equal to total particulates (including those formed in the atmosphere from emitted sulfur dioxide), one based on California's ambient air quality standards, and one based on statistically estimated effects on mortality. At the fuel prices considered most likely, methanol is far more costly than the other strategies per unit reduction in total particulates, but this disadvantage is greatly reduced according to the other indices. In addition, methanol achieves the greatest absolute reduction in emissions. With the mortality-based index, the incremental cost of the methanol strategy over that of particulate traps in the Los Angeles basin comes to $1.6 million per incremental reduction in expected deaths.

Two recent policies on air pollution and energy have combined to focus attention on urban transit buses. First, new federal emissions standards for diesel-powered vehicles are especially strict for transit buses and will probably force early decisions on technologies with substantial start-up costs. Second, a broad interest in methanol as a motor fuel brings attention to transit buses as a test case and possible starting point for methanol conversion: reasons include easily regulated public agencies, central fueling facilities, high current emissions of particulates and sulfur oxides (two of the most well-established health hazards), and emissions at street level in places with high population exposures.

An earlier study (*1*) found evidence that reducing the number of deaths from cancer associated with particulates and sulfates may by itself justify the likely costs of converting transit buses in the Los Angeles air basin from the low-sulfur diesel fuel now required there to methanol. Sulfate reduction accounted for about two-thirds of the estimated benefits.

However, alternative means of reducing diesel emissions such as cleaner fuel and trap oxidizers (also known as particulate traps) must also be considered. Weaver et al. (2) review these and other technologies and compare the costs of reducing particulates by various methods assuming successful technological development. Several findings are noteworthy.

First, they find that lowering the sulfur content of diesel fuel to that now required in Southern California (0.05 percent by weight, about one-sixth the national average) more than pays for itself in reduced engine wear and less frequent changes of lubricating oil, and that refiners would find it to their advantage to simultaneously lower the fuel's aromatic content. (Aromatics are compounds containing a benzene ring.) As a bonus, this would reduce emissions of sulfur oxides, particulates, hydrocarbons, and nitrogen oxides. They also estimate that refiners could lower aromatic content still further at a small extra cost. These results are controversial and hard to reconcile with the authors' expectation that, absent government regulation, the quality of diesel fuel will deteriorate. Nevertheless, low-sulfur fuel is an attractive strategy even under much more pessimistic assumptions. For these reasons, it appears best to include 0.05 percent sulfur fuel as part of a base case for analyzing any more ambitious strategies.

Weaver et al. also find that once low-sulfur, low-aromatic fuel is adopted as a baseline, trap oxidizers offer a cheaper means than methanol of removing additional particulates from the air. The cost estimates are $4.71 and $10.34 per kilogram of particulates for two different trap designs, compared with $13.03 for methanol under their most optimistic assumptions.

In this paper, such cost-effectiveness comparisons are further explored by introducing several variations and refinements to the analysis of Weaver et al. First, as just noted, low-sulfur fuel is adopted as a baseline, but with less optimistic assumptions about engine wear and aromatic content. Second, sulfur-oxide (SO_x) emissions are incorporated into the effectiveness measure, and the consequences of various estimates of their noxiousness relative to that of particulates are explored. Third, the incremental cost-effectiveness of using a methanol strategy to achieve reductions beyond those achieved by clean fuel or particulate traps, or both, is examined. Finally, the price of methanol fuel is varied. The results are a confirmation of the promise of particulate traps and a clearer delineation of the potential role of methanol.

Relatively optimistic assumptions are adopted throughout for both particulate traps and methanol, assuming success of current efforts to overcome technological barriers. Data from

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the Los Angeles air basin are used for many of the needed parameters, though the comparisons of pollution control strategies should be representative of most U.S. urban areas.

MEASURES OF EFFECTIVENESS

Three different methods of weighing the damaging effects of particulates and SO_x are considered. [Nitrogen oxides (NO_x) are not considered because of their more complex role in photochemical-oxidant formation.] The first is the measure of "total particulates" that Weaver et al. use in the findings discussed previously; it incorporates the fact that SO_x become particulates in the atmosphere, a phenomenon they term "indirect particulates." The second weighs each emission according to its contribution to causing any of the ambient pollution standards to be reached in the air basin, a concept introduced by Babcock (*3*). The third weighs them according to their relative contributions to mortality, using the statistical evidence of Lave and his coworkers (*4, 5*). Each of these is discussed in the subsections that follow.

All of these measures ignore distinctions among particulates of different sizes. It is now known that the most damaging particulates are the smaller ones (*6*). Indeed, California has replaced its ambient particulate standard with one for particles of 10 microns or less in diameter. Because diesel emissions fall mainly in this size category, the severity of their effects is probably greater than implied by the methods used here. This would make particulate traps relatively more attractive compared with methanol. On the other hand, omission of methanol's NO_x reductions biases the results in the other direction (presuming that any local ozone-scavenging benefits of NO_x are more than offset by its contribution to areawide smog). Both of these limitations can be overcome through further research.

Total Particulates

Total particulates are the result of both direct particulate emissions and atmospheric reactions involving gaseous emissions. The sulfur in diesel fuel is emitted in oxygenated compounds known collectively as sulfur oxides (SO_x). A small portion of these emissions, mainly consisting of sulfuric acid droplets, belongs to a category of particulates known as sulfates. The rest of the SO_x emissions are sulfur dioxide (SO_2), a gas that reacts in the atmosphere to form additional particulates of the sulfate class, including sulfuric acid and ammonium sulfate. On the basis of atmospheric modeling (*7*), the California Air Resources Board staff estimates that each gram of SO_2 emitted produces 1.2 g of particulates in the atmosphere (*8*, pp. 60–63). Citing this estimate, Weaver et al. (2) define

$$\text{Total particulates} = P + SO_4 + 1.2(SO_2) \quad (1)$$

where P, SO_4, and SO_2 denote direct emissions of carbonaceous (i.e., nonsulfate) particulates, sulfates, and sulfur dioxide, respectively, from a transit bus.

Severity Index

This index is based on California's ambient air quality standards and is constructed somewhat analogously to the federal Pollutants Standards Index, as described in the U.S. Code of Federal Regulations (40 C.F.R. Part 58, Appendix G). The idea is simply to assume that all relevant effects, such as health, visibility, and damage to plants and materials, have been incorporated in setting these standards. Hence the relative severity of a pollutant is measured by the increase in ambient concentration, as a fraction of the relevant standard, that it causes. Computing this requires not only knowledge of the standard but a model of the relationship between emissions and ambient concentrations.

That relationship is complicated because ambient standards are set for both sulfates and SO_2 and because the standard for SO_2 consists of two joint standards, one with particulates and one with NO_x. The latter is ignored here, but the joint standard for SO_2 and particulates, based on a well-established synergism (*9*, p. 16), is accounted for in the same way as in the Pollutants Standards Index: by assuming that the standard establishes a degree of severity for the product of the two concentrations.

The specific assumptions follow:

1. Ambient concentrations of total suspended particulates are proportional to the "total particulate" emissions as defined in the previous subsection (except that, for simplicity, the slight difference between the two components of SO_x is ignored here):

$$C_p = a_p E_{tp} \quad (2)$$

$$E_{tp} = E_p + 1.2E_{sox} \quad (3)$$

$$E_{sox} = E_{so4} + E_{so2} \quad (4)$$

where C_p is ambient particulate concentration and E designates total emissions of a pollutant throughout the air basin.

2. Ambient concentrations of sulfates and of SO_2 are each proportional to SO_x emissions, with different proportionality constants:

$$C_{so4} = a_{so4} E_{sox} \quad (5)$$

$$C_{so2} = a_{so2} E_{sox} \quad (6)$$

3. The damage from an ambient concentration according to a given standard is proportional to the ratio of the concentration to the standard, for each of the following three standards: $\overline{C_p}$, $\overline{C_{so4}}$, and $\overline{C_{pso2}}$, the latter being the product of the particulate concentration and the SO_2 concentration that together define the standard. Furthermore, the damage from these three ratios is additive, and the amount of damage that occurs when any of the three standards is reached is the same. Denoting damage by D and a proportionality constant by b, this implies that

$$D = b\,[(C_p/\overline{C_p}) + (C_{so4}/\overline{C_{so4}}) + (C_p \cdot C_{so2}/\overline{C_{pso2}})] \quad (7)$$

By substituting Equations 2–6 into Equation 7, the relative severities of the two types of emissions (particulates and SO_x) can be calculated as the partial derivatives of D with respect to E_p and E_{sox}. Dividing by b, denoting the results by D_p and D_{sox}, and using Equations 2, 5, and 6 to eliminate the proportionality constants yields

$$D_p = (1/E_{tp})\ [(C_p/\overline{C_p}) + (C_p \cdot C_{so2}/\overline{C_{pso2}})] \quad (8)$$

$$D_{sox} = (1.2/E_{tp})\ [(C_p/\overline{C_p}) + (C_p \cdot C_{so2}/\overline{C_{pso2}})] + (1/E_{sox})\ [(C_{so4}/\overline{C_{so4}}) + (C_p \cdot C_{so2}/\overline{C_{pso2}})] \quad (9)$$

The three standards are those that applied in California in July 1983, just before the new fine particle standard went into effect. In all three cases the averaging period is 24 hr (when there is more than one standard for the same pollutant, only the 24-hr average is used). Ambient concentrations are taken to be the highest 24-hr average observed at the downtown Los Angeles monitoring station during 1985. Emissions are those estimated for the South Coast Air Quality Management District, which includes Los Angeles and Orange counties plus those parts of San Bernardino and Riverside counties that are geographically part of the basin; unfortunately, emissions data are for 1983 because 1985 estimates are not yet available.

Table 1 gives the data. Note that neither of the standards involving sulfur was violated, though they were violated at monitoring stations further inland. Hence the proportionality assumption, which implies that a given increase in concentration is just as damaging whether or not any particular threshold has been reached, is important. This assumption is supported by several lines of evidence. First, most epidemiological studies have failed to find thresholds [e.g., Lave and Seskin (*4*, p. 51)], though some possible evidence is noted by Lipfert (*13*, p. 208). Second, beliefs in thresholds have failed to hold up under scrutiny by four separate panels of the National Academies of Sciences and Engineering for four separate pollutants (*14*, pp. 6, 190, 366–367, 400). Third, even if thresholds exist for individuals, averaging over time, space, and people with varying sensitivities will tend to remove the threshold effects from aggregate population responses. See Small (*15*, pp. 111–112) for further discussion.

The resulting values have the ratio $D_{sox}/D_p = 4.17$. Hence,

$$\text{Severity index} = P + 4.17\ (SO_x) \quad (10)$$

Mortality Index

The statistical work reviewed by Frederick et al. (*1*) indicates that particulate and sulfate concentrations affect mortality across U.S. metropolitan areas. The results are measured as elasticities of .0119 and .0500, respectively. Particulate concentration is assumed to be proportional to carbonaceous particulate emissions, and sulfate concentration to SO_x emissions. Hence the proportional rise in mortality ($\Delta M/M$) cause by bus emission of particulates and SO_x is:

$$\Delta M/M = .0119\ (P/E_p) + .0500\ (SO_x/E_{sox}) \quad (11)$$

Total emissions (E) in the air basin are again taken from the last two rows of Table 1, resulting in

$$\Delta M/M = 54.4 \times 10^{-12}\ [P + 17.0\ (SO_x)] \quad (12)$$

Hence,

$$\text{Mortality index} = P + 17.0\ (SO_x) \quad (13)$$

Note that all three of the indices are defined in units of kilograms of carbonaceous particulate emissions.

SCENARIOS

Five scenarios, a baseline and four control strategies, are analyzed. Each is described in a subsequent subsection. The resulting parameters are summarized in Table 2.

Baseline

Weaver et al. (*2*) make a persuasive case that low-sulfur fuel similar to that already required in Southern California is an attractive measure for any area with an air pollution problem. Using the U.S. Department of Energy's Refinery Evaluation Modeling System, a linear programming model of refinery operations, they project the additional cost to be well within the 3 cent per gallon differential now observed between Southern California and other areas (*2*, p. 234). This projection allows diesel fuel to be segregated from residual oil in the refining process, but it does not permit the sulfur content of residual oil to be increased; instead, the extra sulfur is recovered and sold. Because of this segregation, it becomes feasible (and, according to the model's results, even cheaper) to lower the aromatic content of the diesel fuel by about 8 percentage points, providing possible side benefits of better cold starting and lower emissions of particulates, hydrocarbons, and NO_x. Furthermore, recent laboratory evidence suggests that lowering sulfur content would substantially reduce engine wear and associated maintenance requirements. Finally, the lower sulfur content improves the operation of particulate traps by permitting catalytic oxidation of hydrocarbons without creating excessive sulfates (*2*, p. 236).

The findings on both engine wear and aromatic content are novel and await verification, but even without those advantages, desulfurization is an attractive control strategy because of its simplicity, ease of introduction, and applicability to all existing diesel vehicles. Hence, in this paper it is assumed that

TABLE 1 DATA FOR SEVERITY INDEX[a]

	Standard ($\overline{C}$)	Actual (C)	Ratio ($C/\overline{C}$)
Concentrations			
Particulates (*p*)	100 μg/m³	208 μg/m³	2.08
Sulfates (*so4*)	25 μg/m³	20 μg/m³	0.80
Particulates and SO_2 (*pso2*)	(100 μg/m³) × (.050 ppm)	(208 μg/m³) × (.021 ppm)	0.874
Emissions (E)			
Particulates (*p*)	218.6×10^6 kg/year		
Sulfur oxides (*sox*)	54.1×10^6 kg/year		

[a]SOURCE: South Coast Air Quality Management District for standards (*10*, pp. 14, 44); concentrations (*11*, pp. 41, 42, 45); and emissions (*12*, p. 17).

TABLE 2 ASSUMPTIONS

	Baseline	Fuel Modification	Particulate Traps	Fuel Modification and Particulate Traps	Methanol
Extra vehicle cost					
Capital ($)	0	0	1,100	1,100	5,200
Maintenance ($/yr)	0	0	315	315	582
Fuel quality					
Sulfur (%)	0.05	0.05	0.05	0.05	0.00
Aromatics (%)	28.70	17.00	28.70	17.00	NA
Fuel economy (mi/gal)	3.81	3.81	3.70	3.70	1.81
Fuel price ($/gal)	0.78	0.791	0.78	0.791	0.55
Emissions (g/mi)					
Carbonaceous particulates	6.080	4.256	0.608	0.304	0.304
SO_4	0.026	0.026	0.080	0.080	0.000
SO_2	0.836	0.836	0.809	0.809	0.000

NOTE: Annual mileage = 34,115; real interest rate = 8.0 percent; bus life = 12 years; and capital recovery factor = 0.1296. NA = not applicable.

any area giving serious consideration to methanol would first adopt the 0.05 percent sulfur standard for diesel fuel, and all strategies are analyzed relative to that standard. Neither the reduction in aromatics nor the increase in engine life suggested by Weaver et al. is assumed because those benefits have not yet been confirmed. Included, however, are the reduced maintenance requirements that they estimate: an $8,000 engine overhaul at 234,000 instead of 180,000 mi, plus a $35 oil change every 6,500 instead of every 5,000 mi.

It is assumed that each bus runs 34,115 mi per year and lasts $T = 12$ years; this was the case for Southern California in 1984 (*16*), and is similar for other areas of the United States. Following Weaver et al., the baseline fuel economy is set at 3.81 mpg. A real interest rate (r) of 8 percent per year compounded continuously is also assumed; thus expenses occurring at t years are discounted by the factor e^{-rt}, and an initial capital expense is annualized by the capital recovery factor $r/(1 - e^{-rT}) = 0.1296$.

Virtually all sulfur in the fuel is emitted as some sulfur compound. According to Weaver et al., about 2 percent of the sulfur (atomic weight 32) is emitted as sulfates, mainly H_2SO_4 (atomic weight 98); the rest is emitted as sulfur dioxide (SO_2, atomic weight 64). With fuel weighing 3.249 kg/gal and containing 0.05 percent sulfur by weight, a bus burning 1 gal every 3.81 mi therefore emits 0.026 g/mi sulfates and 0.836 g/mi SO_2.

Emissions of carbonaceous particulates, in contrast, depend greatly on engine design, fuel, age, maintenance policies, and method of measurement. The most appropriate data for present purposes are from buses in actual use, tested with the Environmental Protection Agency's (EPA's) transient bus cycle. Three buses measured in this way by the Southwest Research Institute had particulate emissions averaging 6.24 g/mi (*17*, Table 12). Subtracting 0.16 g/mi of sulfates (obtained by the same method but for fuel with 0.3 percent sulfur) yields carbonaceous particulate emissions of 6.08 g/mi.

Low-Aromatic Fuel

As already noted, Weaver et al. find that some reduction in aromatics, to 20.3 percent, would occur as a by-product of producing low-sulfur fuel. They also analyze a fuel in which aromatics are lowered still further, to 17 percent, and find that this adds only 0.3 cent per gallon to the cost. Extrapolating linearly to estimate the cost of reducing aromatic content from the baseline value of 28.7 percent to 17.0 percent yields 1.1 cents per gallon as the extra cost of this low-aromatic fuel. Refiners surveyed by the California Air Resources Board (*8*, pp. 74–79) were more pessimistic, but the basis for their estimates and their assumptions about sulfur requirements are unclear.

Other properties of low-aromatic fuel are taken directly from Weaver et al. No change in engine life or maintenance is attributed to the reduction of aromatics. Fuel economy tends to be lower during steady operations but higher during warm-up, so it is assumed to be unchanged on average. Carbonaceous particulate emissions are reduced 30 percent, based on engine tests (*18*).

Particulate Traps

Weaver et al. analyze two types of traps now under development: ceramic monolith and wire mesh. Although the ultimate comparative advantages of these and other types are still in doubt, Weaver et al. find the ceramic monolith to be both cheaper and more effective. Their estimates for the ceramic monolith with a catalytic afterburner (permitted by the low-sulfur fuel) are therefore adopted as representing a realistically optimistic strategy.

These estimates are $1,100 capital cost; $350 maintenance cost every 45,500 mi; 3 percent degradation of fuel economy; 85 percent reduction in carbonaceous particulates from the trap and an unspecified reduction from the afterburner, which is taken to be an additional 5 percent; and a 4 percentage point rise in the portion of sulfur emitted as sulfates, caused by oxidation of SO_2 in the afterburner.

Low-Aromatic Fuel and Particulate Traps

This scenario combines the extra cost of low-aromatic fuel with the extra vehicle costs and fuel economy penalty of particulate traps. Weaver et al.'s estimate of a 95 percent reduction in carbonaceous particulates is used.

Methanol

In this scenario, use of methanol fuel in buses is made possible either by retrofitting during engine overhaul or by purchasing new buses designed for methanol. The extra cost for a new bus has been estimated at $6,000 to $7,000 by General Motors, assuming regular production (*19*, p. 125). Of course, further refinement of the technology may reduce this differential. Weaver et al.'s "optimistic" estimate of $5,200 is used here.

The effects on engine life, routine maintenance, and frequency of engine overhaul are not yet known because of the brevity of field tests of methanol-powered buses. However, there is good reason to fear that methanol's corrosiveness will cause at least as much piston wear and degradation of lubricating oil as does current high-sulfur fuel. This is what Weaver et al. adopt as their optimistic case; with the assumptions outlined in the baseline scenario, this adds $582 per year to the annualized cost of upkeep.

Weaver et al.'s "optimistic" fuel economy of 1.81 mpg for methanol is adopted. Because methanol's energy content is about 45 percent that of diesel fuel, this is equivalent to assuming that a methanol engine is about 7 percent more efficient than a diesel engine—a figure probably at the optimistic end of the range of reasonable claims. Weaver et al.'s optimistic estimate of a 95 percent reduction in carbonaceous particulates is adopted; sulfur oxides are entirely eliminated.

Fuel Prices

The comparisons to be made here are quite sensitive to the price differential between diesel and methanol fuel. Because world markets are in flux, this differential is quite uncertain and its effects on the cost-effectiveness comparisons are explored later. In this section, however, it is useful to use a single price for each scenario.

The price of No. 2 diesel fuel delivered directly by refiners to large end users has varied widely; it ranged between 40 and 86 cents per gallon in 1985–1987 and was in the neighborhood of 55 cents for most of 1987 (*20*, Table 9.7). The future price will probably show a long-term upward trend as petroleum becomes scarcer. Hence a reasonable price for scenarios with 12-year time horizons is somewhat above the midpoint of the 40 to 86 cent range. The figure of 75 cents plus 3 cents for desulfurization is used.

The market for methanol is even more uncertain. The industry is currently depressed, with a lot of excess capacity. Chemical-grade methanol has recently been purchased for California fleets at delivered prices of from 55 to 60 cents per gallon. A significant increase in demand would help relieve the excess capacity and could force the market up a rising short-run supply curve; along with a general upward trend in world energy prices, this would tend to raise the price of methanol. On the other hand, economies of scale in transportation (which accounts for a substantial portion of the delivered price) and the marketing of a lower-purity fuel-grade product would have the opposite effect. Hence, for the optimistic scenario, a price equal to the lower end of the recent range, 55 cents per gallon, is adopted. Note that when energy content is corrected for, this is $1.22 for the amount of energy contained in 1 gal of diesel fuel; hence the price differential assumed here is $1.22 – $0.78 = $0.44 per diesel-equivalent gallon.

RESULTS

Cost-Effectiveness

Table 3 gives the extra cost, compared with the baseline scenario, of each of the four control strategies under the previously discussed assumptions. It also gives, for each of the three alternate effectiveness measures, the percentage reduction in that measure and the cost per unit of reduction, labeled "cost-effectiveness." Recall that, in each index, a change of one unit produces pollution damage equivalent to one kilogram of particulates; hence the indices may be thought of as being in units of "particulate-equivalent kilograms."

These comparisons verify at least two of Weaver et al.'s findings. First, lowering the aromatic content of fuel is the most cost-effective way to achieve relatively small pollution reductions, even starting with low-sulfur fuel as a baseline. This is true for all three measures, despite the pessimistic assumptions about the cost of reducing aromatics. However, this strategy does not achieve a very high degree of control, especially when sulfur oxides are given high weight.

Second, particulate traps achieve pollution reductions at lower unit cost than does methanol. Again, this is true using any of the three measures. Using Weaver et al.'s total-particulates measure, for example, particulate traps cost $3.63 per kilogram removed, whereas methanol conversion costs nearly $20. By way of comparison, the California Air Resources Board estimates the cost of reducing emissions of fine particulates from industrial boilers and oil-fired utility boilers at from $1.59 to $2.67/kg (*8*, pp. 89–90).

Nevertheless, the use of weights reflecting the damaging potential of sulfur emissions substantially reduces the cost disadvantage of methanol relative to other strategies. For example, the mortality index is reduced at a cost of $3.95/kg by particulate traps or $6.65/kg by methanol.

Incremental Cost-Effectiveness

No matter which effectiveness measure is used, control stringency and cost-effectiveness both increase from left to right in Table 3. To determine whether the more stringent strategies are justified, the incremental cost of achieving a higher degree of stringency must be examined and compared with the social benefit of further control or with the cost of achieving the same reduction in other ways.

The rows labeled "incremental cost-effectiveness" show, for each strategy, the per unit cost of reducing an emissions index below its value for the next most stringent strategy. These figures show the classic rising marginal control cost presented in the standard economic theory of pollution control (*21*, p. 89). There is one exception: using the mortality index, the per unit incremental cost of adding fuel modification to a particulate trap strategy is higher than that of going to methanol (which is $7.53/kg relative to particulate traps alone, not shown in the table).

TABLE 3 RESULTS OF THREE COST-EFFECTIVENESS MEASURES

	Fuel Modification	Particulate Traps	Fuel Modification and Particulate Traps	Methanol
Cost increase per bus ($/yr)	98	674	776	4,638
Total particulates				
Emissions reduction (%)	25.7	76.7	80.9	95.7
Cost-effectiveness ($/kg)	1.58	3.63	3.95	19.98
Incremental cost-effectiveness ($/kg)	1.58	4.65	9.79	107.70
Severity index				
Emissions reduction (%)	18.9	55.4	58.5	96.9
Cost-effectiveness[a]	1.58	3.69	4.02	14.51
Incremental cost-effectiveness[a]	1.58	4.77	9.79	30.53
Mortality index				
Emissions reduction (%)	8.8	24.1	25.6	98.5
Cost-effectiveness[a]	1.58	3.95	4.28	6.65
Incremental cost-effectiveness[a]	1.58	5.30	9.79	7.49
Expected mortality reduction (deaths/yr)	1.28	3.51		14.33
Incremental cost-effectiveness ($/$10^{-6}$ death)	0.34	1.14		1.62

[a]Cost-effectiveness is expressed in dollars per unit reduction in the index [i.e., in dollars per reduction in pollution that is equivalent (as measured by that index) to 1 kg of particulates].

Using total particulates or the severity index as measures, the additional reduction involved in going from particulate traps (with or without low-aromatic fuel) to methanol comes at a markedly higher cost than previous reductions. With the mortality index, however, the figures exhibit a modest upward progression from fuel modification to particulate traps to methanol. The incremental cost of reducing the mortality index from 76 percent of the baseline value to 1.5 percent of the baseline value by means of methanol conversion is about $7.50/kg, only $2.20 more than the incremental cost of particulate traps themselves.

Cost-Effectiveness of Mortality Reduction

Because the mortality index is derived from estimates of reduced mortality, its results can be restated directly in terms of reduced risk of death to residents of the air basin. Multiplying Equation 12 by the Los Angeles air basin's annual mortality rate of 8,025 per million, and by its population of 10.62 million, gives the change in expected annual deaths due to a unit change in the index. The result, 4.64×10^{-6}, is used to compute the last two rows of Table 3. (Because the combination of particulate traps and fuel modification does not appear promising using this index, it is omitted as a control strategy in these two rows.) The reduction in expected mortality from controlling a single bus is multiplied by 4,432, the number of buses operating (*16*), in order to express it as the reduction in expected annual deaths in the air basin. For example, converting the entire fleet to methanol would reduce deaths in the basin by an expected 14.33 deaths per year.

These numbers make it possible to assess the value that would have to be placed on a small reduction (Δp) in an average person's annual risk of dying in order to justify each increasing degree of control stringency for transit buses. This value, divided by Δp, is called the "value of life," somewhat misleadingly because it is not the amount that a person would pay to avoid certain death (*1*, *22*). Freeman (*23*, p. 39) calls it the "value of statistical life." The data in Table 3 imply that fuel modification is worthwhile if the value of statistical life is between $340,000 and $1.14 million; that particulate traps are warranted if the value of life is between $1.14 million and $1.62 million; and that methanol conversion is warranted at values above that.

By way of comparison, recent studies of labor markets carefully reviewed by Kahn (*24*) suggest that workers in the United States are willing to forgo about $800 per year in order to reduce their risk of fatal injury by 1 in 10,000 per year. This implies a value of statistical life of $8 million. This value of statistical life would amply justify the most stringent control strategy considered here, namely methanol. Another way to view this number is to multiply it by 4.64×10^{-6}, the estimate derived of change in expected deaths per kilogram of particulates removed, to obtain a social value of particulate reduction of $37/kg. The corresponding value for SO_x is $630/kg.

At the more conservative $2 million value of statistical life recommended by Viscusi (*25*, p. 106), methanol is still justified if the estimated costs and mortality reductions are correct. It must be remembered, moreover, that these figures include only particulates and SO_x; that they include mortality but not sickness, material damage, impaired visibility, or other adverse effects; and that they ignore the higher population exposures caused by transit buses' proximity to crowds of people. Hence the overall effectiveness of the control strategies may be substantially higher than indicated here.

Effect of Methanol-Diesel Price Differential

The cost of the methanol strategy presented here is dominated by its higher fuel cost. At the prices assumed, methanol costs 56 percent more than diesel for the same amount of energy. Even with a more efficient engine, this leads to an extra fuel cost of $3,382 per year per bus, nearly three times as much as the annualized extra vehicle cost. Hence, any comparison of strategies is sensitive to fuel prices, which are very uncertain.

Table 4 gives just the comparison of particulate traps and methanol, but with the methanol-diesel price differential ranging from zero to $1.11 per amount of energy contained in a

TABLE 4 EFFECTS OF VARYING METHANOL PRICE

	Particulate Traps	Methanol		
Methanol price ($/gal)		0.35	0.55	0.85
Methanol-diesel price differential ($/diesel-equiv gal)		0.00	0.44	1.11
Cost-effectiveness ($/kg)				
Total particulates	3.63	3.74	19.98	44.33
Severity index[a]	3.69	2.72	14.51	32.20
Mortality index[a]	3.95	1.25	6.65	14.77

[a]Cost-effectiveness is expressed in dollars per unit reduction in the index [i.e., in dollars per reduction in pollution that is equivalent (as measured by that index) to 1 kg particulates].

gallon of diesel fuel. A zero price differential could occur, for example, if methanol could be made from coal at 71 cents per gallon as estimated by Gray and Alson (*19*, p. 27) and if diesel fuel prices were to rise to $1.29/gal, about 30 percent above their 1981 level.

If the energy-equivalent price differential were to fall to zero, particulate traps would become a distinctly less desirable strategy because methanol conversion would equal or dominate it on all three effectiveness measures. Even at the highest methanol price shown, methanol's cost per unit reduction in the mortality index is a moderate $15/kg, well below the estimated social value of $37. (Methanol's incremental cost-effectiveness relative to particulate traps, not included in the table, is $18/kg at that price.) Hence a strong case can be made for methanol even at this substantially higher price if mortality reduction is believed to be worth the amount suggested by the preceding discussion.

Low-Sulfur Baseline

The same methodology can be used to check the internal consistency of the argument that low-sulfur fuel is a sensible baseline scenario. As discussed earlier, a pessimistic estimate of the cost of reducing sulfur content from the current national average of 0.29 percent (2, p. 232) to 0.05 percent is only 3 cents per gallon. Making no allowances for offsetting savings in maintenance or engine life, this strategy still costs only $269 per year per bus; it reduces annual emissions of SO_4 and SO_2 by 4.3 and 136.9 kg per bus. This produces very favorable cost-effectiveness values: $1.59 for total particulates, $0.46 for the severity index, and an astonishing $0.11 for the mortality index. The latter implies a cost of only $24,000 per statistical life "saved." Even using the total-particulate measure, which assigns no more damage to sulfates than to any other particulate matter, low-sulfur fuel has a cost-effectiveness as good as that of any of the strategies considered in the rest of this paper, and better than particulate traps or methanol.

There can be little doubt that reducing the sulfur content of diesel fuel, at least to 0.05 percent, is a sound first step for control of particulates and sulfur compounds. The case is so strong as to immediately suggest the need to carefully estimate the cost of reducing it even further. Such a strategy might turn out to be more cost-effective than any of the strategies considered here. And as noted earlier, it has the additional advantages of simplicity, ease of introduction, and applicability to existing vehicles.

CONCLUSION

The comparison of strategies for reducing diesel emissions depends critically on the weight placed on sulfur oxides relative to carbonaceous particulates. If account is taken of particulates only, even including those produced indirectly in the atmosphere from gaseous emissions, methanol appears a far more costly strategy than either low-aromatic fuel or particulate traps. No seriously proposed estimate of benefits would justify the incremental cost of $108/kg entailed in going from particulate traps to methanol. Only if methanol prices drop nearly to par with those of diesel fuel would particulate reduction alone justify a methanol strategy, assuming a particulate trap strategy is feasible.

If sulfur is taken into account, however, the picture changes. The incremental cost of using methanol to reduce noxious emissions by the equivalent of 1 kg of particulates is either $30.50 or $7.50, depending on which of two estimates of sulfur's noxiousness is believed. The latter is well within the range that could justify a methanol strategy. Furthermore, if methanol's price were to drop so that it was the same as diesel's on an energy-equivalent basis, its cost-effectiveness would become more favorable than that of particulate traps using either measure, and a higher degree of control would be achieved as well.

Lowering the aromatic content of diesel fuel has promise for achieving modest reductions in particulates. This is especially important because of the possibility of immediate application to the entire vehicle fleet, without waiting for old vehicles to be replaced, and because it can also be applied to trucks without disrupting fueling arrangements or incurring administrative costs. However, the estimates used here of the cost and effectiveness of lowering aromatic content need confirmation. It would also be worthwhile to investigate the cost of reducing sulfur content even below Southern California's limit of 0.05 percent.

These results give considerable support to both particulate traps and methanol as possible strategies. The promise of each warrants further development of the hardware and further refinements in assessing the benefits. The wide range of possible outcomes in such an assessment supports the adoption of emissions regulations that are flexible enough to permit either strategy to emerge as the "winner" as more evidence accumulates.

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REFERENCES

1. S. J. Frederick, J. L. C. Morrison, and K. A. Small. Converting Transit to Methanol: Costs and Benefits for California's South Coast Air Basin. In *Transportation Research Record 1155*, TRB, National Research Council, Washington, D.C., 1987, pp. 12–17.
2. C. S. Weaver, R. J. Klausmeier, L. M. Erickson, J. Gallagher, and T. Hollman. Feasibility of Retrofit Technologies for Diesel Emissions Control. SAE Paper 860296. Society of Automotive Engineers, Warrendale, Pa., 1986.
3. L. R. Babcock, Jr. A Combined Pollution Index for Measurement of Total Air Pollution. *Journal of the Air Pollution Control Association*, Vol. 20, Oct. 1970, pp. 653–659.
4. L. B. Lave and E. P. Seskin. *Air Pollution and Human Health*. The Johns Hopkins University Press, Baltimore, Md., 1977.
5. M. Chappie and L. Lave. The Health Effects of Air Pollution: A Reanalysis. *Journal of Urban Economics*, Vol. 12, No. 3, Nov. 1982, pp. 346–376.
6. H. Ozkaynak and J. D. Spengler. Analysis of Health Effects Resulting from Population Exposures to Acid Precipitation Precursors. *Environmental Health Perspectives*, Vol. 63, 1985, pp. 45–55.
7. G. R. Cass. Sulfate Air Quality Control Strategy Design. *Atmospheric Environment*, Vol. 15, No. 7, 1981, pp. 1227–1249.
8. *Status Report: Diesel Engine Emission Reductions Through Modification of Motor Vehicle Diesel Fuel Specifications*. Energy Section, Strategy Assessment Branch, Stationary Source Division, California Air Resources Board, Sacramento, Oct. 1984.
9. J. L. Horowitz. *Air Quality Analysis for Urban Transportation Planning*. MIT Press, Cambridge, Mass., 1982.
10. *1983 Summary of Air Quality in California's South Coast Air Basin*. South Coast Air Quality Management District, El Monte, Calif., Aug. 1984.
11. *1985 Summary of Air Quality in California's South Coast Air Basin*. South Coast Air Quality Management District, El Monte, Calif., Dec. 1986.
12. *Draft 1983 Emissions Inventory: South Coast Air Basin*. South Coast Air Quality Management District, El Monte, Calif., May 1986.
13. F. W. Lipfert. Air Pollution and Mortality: Specification Searches Using SMSA-Based Data. *Journal of Environmental Economics and Management*, Vol. 11, 1984, pp. 208–243.
14. *Air Quality and Automobile Emission Control*, Vol. 2: *Health Effects of Air Pollutants*. Coordinating Committee on Air Quality Studies, National Academy of Sciences and National Academy of Engineering, Washington, D.C., 1974.
15. K. A. Small. Estimating the Air Pollution Costs of Transport Modes. *Journal of Transport Economics and Policy*, Vol. 11, No. 2, May 1977, pp. 109–132.
16. M. Wachs and N. Levine. *Vehicle Fleets in the South Coast Air Basin: Results of a Survey Performed for the South Coast Air Quality Management District*. Urban Innovations Group, Los Angeles, Calif., 1985.
17. R. D. Wilson and C. Gray, Jr. The Environmental Impacts of Methanol as a Transportation Fuel. Presented at National Science Foundation Workshop on Automotive Use of Methanol-Based Fuels, Ann Arbor, Mich., Jan. 1985.
18. E. G. Barry, L. J. McCabe, D. H. Gerke, and J. M. Perez. Heavy-Duty Diesel Engine/Fuels Combustion Performance and Emissions: A Cooperative Research Program. Paper SAE 852078. In *Diesel Fuel Quality and Trends*, SP-639, Society of Automotive Engineers, Warrendale, Pa., 1985.
19. C. L. Gray, Jr., and J. A. Alson. *Moving America to Methanol*. University of Michigan Press, Ann Arbor, 1985.
20. *Monthly Energy Review*. U.S. Department of Energy, Sept. 1987.
21. E. S. Mills. *The Economics of Environmental Quality*. W.W. Norton, New York, 1978.
22. E. J. Mishan. Evaluation of Life and Limb: A Theoretical Approach. *Journal of Political Economy*, Vol. 79, 1971, pp. 687–705.
23. A. M. Freeman III. *Air and Water Pollution Control: A Benefit-Cost Assessment*. John Wiley and Sons, New York, 1982.
24. S. Kahn. Economic Estimates of the Value of Life. *IEEE Technology and Society Magazine*, Vol. 5, No. 2, June 1986, pp. 24–31.
25. W. K. Viscusi. *Risk by Choice*. Harvard University Press, Cambridge, Mass., 1983.

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COST-EFFECTIVENESS OF EMISSIONS CONTROL STRATEGIES FOR TRANSIT BUSES: THE ROLE OF PHOTOCHEMICAL POLLUTANTS†

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Abstract—We extend a previous cost-effectiveness analysis of methanol vs. other means of controlling emissions from urban transit buses by developing a method to incorporate their effects on two end-product pollutants: ozone and nitrogen dioxide. Using published simulation results from an airshed grid model of ozone formation, we find that the measures we consider have varying effects on ozone at 23 sites in the Los Angeles air basin. The effects are offsetting, leading to a negligible net effect when aggregated across the basin's population; this is true assuming either that damage is proportional to concentration times population exposed, or that damage is represented by nonlinear concentration–response functions for specific health conditions. In contrast, either low-aromatic diesel fuel or methanol would lower ambient concentrations of nitrogen dioxide enough, relative to the federal or California ambient standard, to significantly affect cost-effectiveness comparisons.

1. INTRODUCTION

Alternatives to conventional motor vehicle fuels have been subjected to evaluation by a variety of technical, political, emotional, and scientific means. Increasingly there is interest in evaluating them by economic means as well. One way to do this is to apply cost–benefit analysis, which assigns dollar values to the costs and benefits of a proposed policy. Another is to use cost-effectiveness analysis, which compares a proposed policy with alternative policies having similar aims.

We have contributed to both types of analysis, focusing on the air quality benefits of methanol fuel for transit buses in the Los Angeles air basin (Frederick *et al.*, 1987; Small, 1988). Transit buses seem a particularly promising case for methanol because they are such visible emitters of particulates and sulfates, and because most bus fleets are centrally fueled and government owned.

Cost-effectiveness analysis has several advantages over cost-benefit analysis. It avoids placing monetary values on benefits, a source of uncertainty and controversy. It permits scientific analysis of a target outcome even if the target itself is politically rather than scientifically derived. It focuses attention on comparisons rather than on absolutes, thereby facilitating agreement on methodology.

The cost-effectiveness approach, however, has an important limitation: it is unlikely that each alternative policy will achieve precisely the same benefits, especially if benefits are multidimensional. It then becomes necessary to assign weights to benefits of different types, which may be nearly as hard as assigning monetary values.

In this paper we consider this problem for a specific example: how to incorporate ambient concentrations of ozone (O_3) and nitrogen dioxide (NO_2) into the cost-effectiveness comparisons of Small (1988), which was concerned only with particulates and sulfates. These photochemical pollutants would be affected by adoption of methanol fuel because it produces lower emissions of nitrogen oxides (NO_x) and a different mix of reactive organic gases (ROG) than diesel. Does consideration of O_3 and NO_2 substantially alter the relative advantages of methanol, particulate traps, and cleaner diesel fuel?

Our tentative answer is "no" for ozone and "yes" for NO_2. It appears that diesel's ROG emissions are too small to make a difference, and that the NO_x reductions have offsetting effects on ozone, lowering them in some places and raising them in others. Two different ways of accounting for these offsetting effects, both using an air chemistry model specific to the Los Angeles basin, lead to a negligible net effect from ozone. However, accounting for the direct effects of NO_x emissions on NO_2 concentrations does increase the value of methanol relative to other strategies.

The paper is organized as follows. In the next section we describe the three control strategies that we consider and the baseline from which the effects of each are measured. In Section 3 we describe a way to combine several pollutants into a single "severity index" on which to compare the strategies. In Section 4 we present a method for predicting how ROG and NO_x emissions affect ozone exposures at locations distributed throughout the Los Angeles basin; a model of some complexity is required because of ozone's indirect and geographically varied

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process of formation. The results of the cost-effectiveness calculations are presented and discussed in Section 5, which is followed by a conclusion.

The severity index incorporates several pollutants simultaneously by making greatly simplified assumptions about their effects. To explore the effect of changing some of these assumptions, we provide in Appendix B an analysis of ozone health benefits that uses nonlinear concentration–response functions to predict the incidence of several specific health conditions at locations throughout the basin. This work supports the conclusion that the ozone changes have negligible net effects.

2. CONTROL STRATEGIES

We consider three control strategies for diesel transit buses: cleaner diesel fuel, particulate traps, and methanol fuel. Each of these is analyzed relative to a baseline that approximates mid-1980s conditions in the South Coast Air Basin in California, consisting of Los Angeles and Orange Counties plus the nondesert parts of San Bernardino and Riverside Counties. These baseline conditions include use of low-sulfur diesel fuel (0.05% sulfur by weight, the legal maximum in Southern California), which we believe to be a likely first step toward more stringent controls on diesel vehicles anywhere (Weaver *et al.*, 1986; Small, 1988).

We analyze each control strategy under plausible but optimistic assumptions. Hence, our results should not be taken as predictions of what conditions will prevail for a particular control strategy, but rather as calculations of what would happen if technological and economic factors turn out as favorably as may reasonably be hoped. For example, we assume that current technical problems with particulate traps are resolved without significant extra cost and that buses can be adapted to methanol at low cost without experiencing severe corrosion; both of these are problems currently under study with results as yet unproved.

Many of our assumptions follow those of Small (1988), which in turn rely heavily on Weaver *et al.* (1986) (see Table 1 for a summary). Costs are at 1986 price levels. We assume 4,432 buses, each running 34,115 miles per year for 12 years (Wachs and Levine, 1985) at 3.81 miles per gallon of fuel. Capital expenses are annualized assuming continuous compounding at a real interest rate of eight percent per year. We assume the maintenance requirements that Weaver *et al.* estimate for this low-sulfur fuel, but we do not adopt the increased engine life and lower aromatic content of fuel that they postulate will accompany this level of sulfur, pending verification of their results.

Our estimates of diesel-bus emissions average the emissions of seven buses taken from operating service in Houston and San Antonio, as reported in Alson (1985, Table 5) except that, based on Alson *et al.* (1989) and a conversation with Jeff Alson and Tom Baines, we assume that formaldehyde emissions are 7.5% of hydrocarbons. We assume that all particulates are less than 10 microns in diameter, making them part of a class known as PM10, and that all but 0.16 grams/mile are carbonaceous (see Small, 1988, p. 18). We have combined hydrocarbon, methanol, and formaldehyde emissions into a single index of reactive organic gases (ROG) using relative weights 1.00 for hydrocarbons, 0.43 for methanol, and 4.8 for formaldehyde (from Alson *et al.*, 1988, p. 7). Sulfur emissions are calculated assuming that two percent of the fuel's sulfur is emitted as sulfuric acid (a sulfate) and the rest as sulfur dioxide (SO_2).

Our low-aromatic fuel strategy postulates a diesel fuel with the same low sulfur content as in our baseline, but with a lower portion of aromatics (chemicals with benzene rings). Note that we are *not* analyzing the effects of lowering the sulfur content. Both the

Table 1. Assumptions

Annual mileage	34,115	Real interest rate	8.0%
Bus life (years)	12	Capital recovery factor	0.1296

	Baseline	Low-aromatic fuel	Particulate traps	Methanol with catalyst
Extra vehicle cost				
Capital ($)	0	0	1,100	5,200
Maint. ($/yr)	0	0	315	582
Fuel quality				
% Sulfur	0.05	0.05	0.05	0.00
% Aromatics	28.70	17.00	28.70	NA
Fuel economy (mpg)	3.81	3.81	3.70	1.81
Fuel price ($/gal)	0.78	0.791	0.78	0.55
Emissions (g/mi)				
Carbonaceous PM10	5.360	3.752	0.536	0.240
SO_4	0.026	0.026	0.080	0.000
SO_2	0.836	0.836	0.809	0.000
ROG	4.550	3.868	1.365	1.310
NO_x	26.100	23.229	26.100	13.600

costs and effects of the clean-fuel approach are somewhat speculative, but from Weaver *et al.*'s analysis it appears that substantial reductions in particulates, ROG, and No_x—we assume 30%, 15%, and 11%, respectively—are possible at quite modest cost.

Our particulate-trap strategy is based upon the analysis in Weaver *et al.* of a ceramic monolith trap-oxidizer followed by a catalytic afterburner. It costs $1,100, requires a $350 maintenance every 45,500 miles, degrades fuel economy by 3%, and reduces carbonaceous particulates by 90% (Small, 1988, p. 18) and NO_x by 70% (roughly in the middle of a range of 50%–90% suggested by conversations with Alson and Baines). There is a slight rise in the portion of sulfur emitted as sulfates because of oxidation of SO_2 in the afterburner.

Our methanol strategy follows the assumptions in Small (1988): extra initial cost of $5,200 per bus, extra engine wear of $582 per year compared to the low-sulfur baseline fuel (but no change from the higher-sulfur diesel now in use in most areas of the United States), and fuel economy of 1.81 mpg, making it seven percent more efficient than a diesel engine. We are aware that these assumptions are optimistic and omit some additional costs such as more frequent fueling, but we also believe that methanol engines will be improved. Emissions data are speculative because there are so few in-use engines, most have been measured only at low mileage, and there is enormous variation from one engine to the next. To not be too optimistic, we assume that emission of each pollutant is equal to the higher of (i) the early in-use chassis measurements for the M.A.N. Golden Gate Transit bus (Alson, 1985, Table 5); and (ii) the engine test of the Detroit Diesel engine planned for Los Angeles (Alson *et al.*, 1989), with the standard conversion factor of 3 brake-horse-power-hours per mile. These assumptions entail reductions in PM10, ROG, and NO_x of 96%, 71%, and 48%, respectively, from our baseline. Methanol combustion produces smaller amounts of reactive hydrocarbons and formaldehyde than does diesel, but it gives off 1.16 grams per mile of unburned methanol where diesel gives off none; we do not address the health effects specific to these particular members of the ROG class of chemicals, but preliminary assessment suggests that unburned methanol will not pose a serious hazard (Alson *et al.*, 1989).

Fuel prices are very important in comparing methanol with other strategies. We adopt highly uncertain assumptions that make methanol 56% more expensive on an energy-content basis: namely, low-sulfur diesel at 75 cents per gallon and methanol at 55 cents per gallon. Frederick *et al.* (1987) and Small (1988) discuss the effects of other price assumptions.

3. SEVERITY INDEX

Small (1988) considered three alternative ways of combining particulates (P) and sulfur oxides (SO_x) into a single index of pollution. The index that gave lowest relative weight to SO_x was total particulates, including sulfate particulates formed in the atmosphere. The index that gave highest was mortality, based upon regression estimates of relative effects of the two pollutants on mortality. The third index, representing something of a middle ground, was called the "severity index" and was based upon ambient air quality standards.

The severity index weights a given emission according to its role in causing a pollutant's concentration to reach the relevant air quality standard, a concept introduced by Babcock (1970). Based on California's ambient standards, it is analogous to the federal Pollutants Standards Index. The idea is simply to assume that all relevant effects have been taken into account in the setting of these standards and that damage is proportional to concentration. Hence, for each pollutant of interest, the ratio of ambient concentration to the standard is calculated, and total damage is measured by summing the ratios.

This idea was implemented by Small for just two pollutants (particulates and SO_x) and three ambient standards (particulates, sulfates, and a joint standard involving sulfur dioxide and particulates). The three standards were those in effect in July 1983, with particulates measured as total suspended particulates (TSP) and all concentrations measured as 24-hour averages. The joint standard for SO_2 and TSP was taken to be a limit on the product of the two concentrations.

In this paper, we update that index by using 1985 data and by replacing the TSP standard with the new standard for particulates of less than 10 microns (PM10), which went into effect in August 1983. We also extend the index by considering standards for nitrogen dioxide (NO_2) and ozone (O_3). The evidence suggests that virtually all NO_x emitted becomes NO_2 eventually, so we assume NO_2 concentrations to be proportional to basin-wide emissions of all NO_x, just as sulfate and SO_2 concentrations are each assumed proportional to basin-wide emissions of all SO_x. Ozone is modeled in more detail, as described in the next section. Full details of the severity index are given in Appendix A.

The result is a revised measure of the severity of emissions from transit buses. A change in the index may be written as a linear combination of small changes in total basin-wide emissions by the four pollutants, namely carbonaceous particulates (ΔE_p), SO_x (ΔE_s), NO_x (ΔE_n), and ROG (ΔE_r):

$$\Delta D = D_p \Delta E_p + D_s \Delta E_s + D_n \Delta E_n + D_r \Delta E_r. \quad (1)$$

4. MODELING OZONE EXPOSURE

Ozone formation is a complex process that depends on many factors including temperature, sunlight, wind, and the ratio of ambient reactive organic gases to nitrogen oxides. Since these factors vary across the air basin, it is not possible to define the kind of simple relation between emissions and ambient ozone concentrations that we use for other pollutants. Instead, we use some results from a computer simulation model developed specifically for the Los Angeles basin by Systems Applications, Incorporated (SAI). The model assumes the existence of the climatic conditions that prevailed on two days in late June 1974, an episode chosen because of the detailed data available and because Los Angeles's well-known temperature inversion prevailed throughout.

Souten *et al.* (1981) used this airshed grid model to evaluate the effects of five different scenarios, each representing a unique percentage reduction in emissions of ROG and NO_x. Each of the five simulations predicted a maximum hourly average ozone concentration at each of 29 monitors distributed throughout the basin.

For our analysis, we select two scenarios whose deviations from a baseline scenario (in both emissions inventory and predicted ozone concentrations) provide the basis for a linear approximation of a highly nonlinear ozone formation process. The baseline is a rough approximation of current emissions. In the first scenario, ROG emissions are reduced by 1.8% and NO_x by 3.1%; in the second, the reductions are 2.1% and 2.5%. Hence, together the two scenarios define the model's sensitivity to small changes in each type of emission, and the derived linear approximation is suitable for the small percentage changes that could be expected from controlling transit buses. We use it to estimate changes in 1985 ozone concentrations at 23 of the 29 monitors (since we lack needed data at the other six monitors). We use the word "changes" rather than "decreases" because reductions in ROG and NO_x may actually increase ozone levels at some monitors, particularly in the west–central part of Los Angeles County.

To evaluate the impacts of these changes in ozone concentrations, we estimate the daytime population exposed to the measured level at each monitor, using maps, city populations, and census data on journeys to work. These and other details of our ozone exposure model are described in Appendix B.

This procedure permits us to describe the ozone levels prevailing at points throughout the air basin both before and after the adoption of any of our strategies, as well as the population exposed to each of those levels. In Appendix A, this information is used to add ozone to the list of pollutants in the severity index. In Appendix B, the same information is used to estimate changes in the extent of five specific ozone-related health conditions throughout the basin.

5. RESULTS

The results for the severity index are shown in Table 2. Below the row showing annual cost increase per bus are three panels, one for each of three versions of the index. The first version contains only the first two terms in eqn (1), and hence includes only the effects of particulate and SO_x emissions. The second version adds the direct effects of NO_x emissions on the NO_2 standard, but omits ozone (it includes the first two terms and part of the third term in the equation). The third version is the full index including ozone. Each panel compares the control strategies both in terms of reduction in that index and in terms of "cost-effectiveness" of that reduction, i.e. the annual cost divided by the reduction in that index, in this case with the index

Table 2. Severity index results

	Low-aromatic fuel	Particulate traps	Methanol with catalyst
Cost increase per bus ($/year)	98	674	4,638
Severity index including ambient standards for PM10, SO_4, SO_2			
Percent reduction†	16.9	49.4	97.5
Cost-effectiveness‡	1.80	4.21	14.69
Incremental cost-effectiveness	1.80	5.47	25.47
Severity index including ambient standards for PM10, SO_4, SO_2, NO_2			
Percent reduction†	14.7	30.6	78.6
Cost-effectiveness‡	1.28	4.21	11.29
Incremental cost-effectiveness	1.28	6.91	15.80
Severity index including ambient standards for PM10, SO_4, SO_2, NO_2, O_3			
Percent reduction†	15.1	37.1	81.7
Cost-effectiveness‡	1.30	3.63	11.35
Incremental cost-effectiveness	1.30	5.24	17.77

†This is the percentage reduction in the contribution of transit buses to the index.

‡Cost-effectiveness is expressed in $ per unit reduction in the normalized index (1986 prices), i.e. in $ per reduction in pollution that is equivalent (as measured by that index) to 1-kg particulates. The more pollutants are included in the index, the larger its value for any scenario; hence, percentage reductions may be smaller even though absolute reductions (as normalized) are larger.

normalized by dividing by D_p. (One may think of cost-effectiveness, then, as cost per kilogram of particulates removed, where all other pollutant reductions are converted to their damage equivalents in particulates.)

Table 2 shows that no matter which index is used, particulate traps achieve a greater reduction than low-aromatic fuel, and methanol achieves the greatest reduction of all. It also shows that going to successively more stringent control strategies involves a substantially higher cost per unit of reduction, again no matter what the index. This does not necessarily mean that the more stringent strategies are unwise, since the additional benefits might still be worth that higher cost; but it does mean that one would want first to investigate the possibilities for more widespread adoption of the cheaper strategies. In the present case, for example, adopting either fuel modification or particulate traps for all heavy vehicles might achieve the same benefits, at less cost, as adopting methanol just for buses.

The last row of each panel shows an incremental cost-effectiveness, which evaluates each strategy relative to the next most stringent one. If one knew the dollar benefits per unit reduction in the index, one would want to adopt the most stringent policy whose incremental cost-effectiveness fell below that benefit estimate.

We can now ask whether accounting for ROG and NO_x makes much difference in the relative cost-effectiveness of various strategies. Comparing the three panels in Table 2 shows that the cost-effectiveness of both low-aromatic fuel and methanol is improved substantially by including NO_2 in the analysis, but that including ozone makes virtually no difference.

There are two reasons for the negligible effects of ozone in these calculations. The first is that ROG emissions from heavy-duty diesel engines are so small relative to other sources in the basin—less than two percent according to South Coast Air Quality Management District (SCAQMD) (1988)—that controlling them has very little impact on ozone formation. This, of course, does not contradict the U.S. Environmental Protection Agency's (EPA) expressed belief that control of ROG "is generally the most promising strategy for reducing ozone levels" (Alson *et al.*, 1989), but only suggests that diesels are the wrong place to look for such control.

The second reason for the unimportance of ozone in our results is the local scavenging effects of NO_x emissions on ozone that are modeled in the underlying SAI simulations. Nitric oxide, the main component of NO_x emissions, initially reacts with ozone, only later producing new ozone through secondary reactions. Hence, ozone may be reduced at sites near NO_x sources even while increased (after several hours' lag) further downwind. A more detailed look at the results by air monitor reveals that, in fact, the increases at some monitors (mainly in the coastal areas) are roughly balanced by decreases at others (mainly inland), resulting in very little net effect. (Our results imply that population-weighted average ozone concentration has an elasticity with respect to ROG emissions of 0.47; but the elasticity with respect to NO_x emissions is only -0.11, a value so small that we regard it as effectively indistinguishable from zero.)

This result must be regarded as tentative pending improved ability to simulate the effects of NO_x on ozone. Indeed, preliminary results of a new simulation model developed at Carnegie-Mellon University, just being released at time of writing, are said by Tom Cackette of the California Air Resources Board (CARB) to show a less important scavenging effect than previous models, including SAI's. Unfortunately, there has not yet been time for the scientific community to evaluate these results, and CARB has already taken a strong regulatory position that relies heavily upon the belief that NO_x's scavenging effects are relatively unimportant (see CARB, 1985). The published descriptions of the first stages of the modeling effort itself seem consistent with the SAI findings that ozone is mainly ROG-limited in central Los Angeles County (Russell and Harris, 1988, pp. 5–6).

We have not taken into account that the geographical distribution of the NO_x emissions of buses differs from that of other vehicles. Buses are concentrated where daytime populations are high. If reducing NO_x emissions does increase ozone locally, buses are located where any such increase will do the most harm. Santini and Schiavone (1988) argue that this factor reduces the case for stringent NO_x controls on transit buses.

As for NO_2 concentrations themselves, our results suggest that the effects of low-aromatic fuel or of methanol conversion are significant in relation to the California NO_2 standard, and that this may be the chief advantage of NO_x reductions. The California NO_2 standard is based upon human responses to short-term exposures (one-hour average), as summarized in SCAQMD (1986, p. 29).† There is some controversy about these short-term health effects, and the federal government has declined to set a short-term standard for NO_2 exposures (Bureau of National Affairs, 1984), although it does use short-term exposure information to trigger declaration of stage-1, stage-2, and stage-3 "episodes" (SCAQMD, 1986, p. 33). But had we instead used the federal long-term standard of 0.053 parts per million annual average (SCAQMD, 1986, p. 32), we would have obtained virtually identical results. This is because the federal standard was exceeded in downtown Los Angeles in 1985 by a ratio nearly identical to the ratio by which the California standard was exceeded

†The source for that evidence was inadvertantly omitted in the SCAQMD publication, but an earlier version shows it to be U.S. EPA (1978).

(1.13 vs. the value of 1.08 shown in Table A1), and it is only this ratio that affects the calculation.

Two alternatives to our severity index deserve comment. One, suggested to us by Danilo Santini, would assume that each ambient standard represents a threshold below which there is no damage. This has a certain consistency with the rationale behind the standards, although we believe that the scientific evidence is mainly against the existence of thresholds (see Appendix A). In most cases, ambient standards were set near the lowest concentrations at which any adverse effects were found (see Appendix B for examples of such studies); but this need not imply that smaller effects, below the experiments' statistical abilities to discriminate, are not present at lower concentrations. Even if there are thresholds at the individual exposure level, they will tend to be blurred by averaging over time and place. Nevertheless, calculating such an index would provide a useful indication of how important the assumption of linear damage functions is to our results. In our case, the concentration as we measured it—namely, the maximum 1985 concentration in downtown Los Angeles (or, in the case of ozone, the maximum concentration at each of 23 monitoring stations)—exceeded the standard in every case except sulfates. Hence, performing this calculation would simply eliminate the role of sulfates, something already studied in Small (1988). A far better way to assess the possibility of thresholds is to perform month-by-month location-specific calculations using nonlinear damage functions, which we do for ozone in Appendix B.

Another alternative to our index is to use relative severities to allocate the costs of a pollution-control strategy to various pollutants. This is the approach taken by Moyer *et al.* (1989), who use allocation formulae incorporating thresholds. Aside from our reservations about thresholds, we believe that the cost-allocation approach is inferior to our severity-index approach because there is no economic principle to justify attributing portions of a joint cost to the individual ends for which that cost is undertaken. Furthermore, the cost-allocation approach has a couple of strange properties. By this measure, a project reducing a given emission appears least favorable precisely when pollutant levels are high, and it becomes extremely favorable when the initial concentration is just slightly above the standard. Also, the cost-allocation approach, despite initial appearances, does not really provide cost-effectiveness information specific to each pollutant; in fact, it ranks all strategies in exactly the same order no matter which pollutant is being considered.

6. CONCLUSION

Our study illustrates some of the difficulties confronting cost-effectiveness analysis when each air pollution control strategy provides a different mix of pollution reductions. We have attempted to discover the importance of ozone and nitrogen dioxide in assessing the relative merits of clean fuel, particulate traps, and methanol conversion as strategies for dealing with pollution from diesel transit buses in the Los Angeles area.

Ozone itself seems not very important in comparing these strategies. Diesel emissions of reactive organic gases are sufficiently small that reducing them has little effect on total emissions in the basin. In contrast, diesels are heavy emitters of nitrogen oxides (NO_x), and reducing these emissions through low-aromatic fuel or methanol conversion would have a sizable effect on total NO_x emissions; but our methodology does not demonstrate much resulting ozone benefit because reducing NO_x has ambiguous impacts on ozone concentration, reducing it in some areas and increasing it in others. This result is specific to the Los Angeles basin and depends on atmospheric modeling which is still under intense study; it is not at all certain that these effects are accurately portrayed by any existing model. We do show in Appendix B that if ozone were purely NO_x-limited, i.e. if reducing NO_x were to reduce ozone proportionately, then either low-aromatic diesel fuel or methanol would create substantial benefits in the form of reduced acute symptoms from ozone.

Perhaps our most surprising result is that the benefits of reducing concentrations of nitrogen dioxide (NO_2), a brown gas contributing to smog, may be greater than the benefits of reducing ozone. This conclusion is based upon our "severity index" which considers emissions in relation to the ambient air quality standard for the pollutants to which they contribute. Most NO_x emissions are in the form of nitric oxide, which is readily oxidized to NO_2 and hence contributes directly to undesirable levels of this pollutant. Although we have not attempted to model the health effects of NO_2 explicitly, our results suggest that more attention might be directed there in the future.

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REFERENCES

Alson J. (1985) The Environmental Protection Agency view of methanol as a transit bus fuel. Paper presented at the Headway '85 Transit Conference, Kansas City, Missouri, November. Emission Control Technology Division, Environmental Protection Agency, Ann Arbor, Michigan.

Alson J., Adler J. and Baines T. (1989) The motor vehicle emission characteristics and air quality impacts of methanol and compressed natural gas. In: *Alternative Transportation Fuels: An Energy and Environmental Solution* (edited by Sperling D.). Quorum/Greenwood Press, Westport Connecticut (in press).

Babcock L. R. Jr. (1970) A combined pollution index for measurement of total air pollution. *J. Air Pollut. Control Assoc.* **20**, 653–659.

Bureau of National Affairs, Inc. (1984) Data insufficient

to support short-term rule for nitrogen dioxide, science panel tells EPA. *Environ. Rep.* **15**, 460–461.
California Air Resources Board (1984) Status report: Diesel engine emission reductions through modification of motor vehicle diesel fuel specifications. Energy Section, CARB, Sacramento.
California Air Resources Board (1985) The effects of oxides of nitrogen on California air quality. Technical Support Division, CARB, Sacramento, November.
California Department of Finance (1986) California statistical abstract 1986. California Dept. of Finance, Sacramento, California.
Detels R., Rokaw S. N., Coulson A. H., Tashkin D. P., Sayre J. W. and Massey F. J. Jr. (1979) The UCLA population studies of chronic obstructive respiratory disease, I: Methodology and comparison of lung function in areas of high and low pollution. *Am. J. Epidemiol.* **109**, 33–58.
Detels R., Sayre J. W., Coulson A. H., Rokaw S. N., Massey F. J. Jr., Tashkin D. P. and Wu M. (1981) The UCLA population studies of chronic obstructive respiratory disease, IV: Respiratory effect of long-term exposure to photochemical oxidants, nitrogen dioxide, and sulfates on current and never-smokers. *Am. Rev. Respir. Dis.* **124**, 673–680.
Detels R., Tashkin D. P., Sayre J. W., Rokaw S. N., Coulson A. H., Massey F. J. Jr. and Wegman D. H. (1987) The UCLA population studies of chronic obstructive respiratory disease, 9: Lung function changes associated with chronic exposure to photochemical oxidants; a cohort study among never-smokers. *Chest* **92**, 594–603.
Dungworth D., Goldstein E. and Ricci P. F. (1985) Photochemical air pollution—part II. *West. J. Med.* **142**, 523–531.
Frederick S. J., Morrison J. L. C. and Small K. A. (1987) Converting transit to methanol: Costs and benefits for California's South Coast Air Basin. *Transpn. Res. Rec.* **1155**, 12–17.
Goldstein E., Hackney J. D. and Rokaw S. N. (1985) Photochemical air pollution—part I. *West. J. Med.* **142**, 369–376.
Horowitz J. L. (1982) *Air Quality Analysis for Urban Transportation Planning.* MIT Press, Cambridge, Massachusetts.
Krupnick A. J. (1986) *A Preliminary Benefits Analysis for the Control of Photochemical Analysis.* Resources for the Future, Washington, D.C.
Kulle T. J., Sauder L. R., Hebel J. R. and Chatham M. D. (1985) Ozone response relationships in healthy nonsmokers. *Am. Rev. Respir. Dis.* **132**, 36–41.
Lave L. B. and Seskin E. P. (1977) *Air Pollution and Human Health.* The Johns Hopkins University Press, Baltimore, Maryland.
Lipfert F. W. (1984) Air pollution and mortality: Specification searches using SMSA-based data. *J. Environ. Econ. Manage.* **11**, 208–243.
Moyer C. B., Unnasch S. and Jackson M. D. (1989) Air quality programs as driving forces for a transition to methanol use. *Transpn. Res.* **23A**, 209–216.
National Academy of Science and National Academy of Engineering (1974) Air quality and automobile emission control, vol. 2. U.S. Government Printing Office, Washington, D.C.
Raloff J. (1986) Lungs hurt most by ozone–acid synergy. *Sci. News* **130**, 85.
Rokaw S., Detels R., Coulson A. H., Sayre J. W., Tashkin D. P., Allwright S. S. and Massey F. J. Jr. (1980) The UCLA population studies of chronic obstructive respiratory disease, 3: Comparison of pulmonary function in three communities exposed to photochemical oxidants, multiple primary pollutants, or minimal pollutants. *Chest* **78**, 252–262.
Russel A. and Harris J. (1988) Quantitative estimate of the air quality effects of methanol utilization: Results of sensitivity analysis. Working paper, Carnegie-Mellon University, Pittsburgh, Pennsylvania.
Santini D. J. and Schiavone J. J. (1988) Technical problems and policy issues associated with the 1991 Bus Emissions Standards. *Transpn. Res. Rec.* **1164**, 5–14.
Small K. A. (1977) Estimating the air pollution costs of transport modes. *J. Trans. Econ. Policy* **11**, 109–132.
Small K. A. (1988) Reducing transit-bus emissions: Comparative costs and benefits of methanol, particulate traps, and fuel modification. *Transpn. Res. Rec.* **1164**, 15–22.
Souten D. R., Tesche T. W. and Oliver W. R. (1981) Use of the airshed grid model to evaluate the air quality impacts of alternative air pollution control policies on the South Coast Air Basin. Report no. 305R-EF81-7. Systems Applications Inc., San Rafael, California.
South Coast Air Quality Management District (1986) 1985 summary of air quality in California's South Coast Air Basin. SCAQMD, El Monte, California.
South Coast Air Quality Management District (1988) Draft appendix III-A: 1985 emissions inventory, South Coast Air Basin. In Draft air quality management plan, 1988 revision. SCAQMD, El Monte, California.
Sun M. (1988) Tighter ozone standard urged by scientists. *Science* **240**, 1724–1725.
U.S. Environmental Protection Agency (1978) Health effects for short-term exposures to nitrogen dioxide. Report 5-53. Office of Research and Development, EPA, Ann Arbor, Michigan.
Wachs M. and Levine N. (1985) Vehicle fleets in the South Coast Air Basin: Results of a survey performed for the South Coast Air Quality Management District. Urban Innovations Group, Los Angeles, California.
Weaver C. S., Klausmeier R. J., Erickson L. M., Gallagher J. and Hollman T. (1986) Feasibility of retrofit technologies for diesel emissions control. Technical Paper Series, No. 860296. Society of Automotive Engineers, Warrendale, Pennsylvania.

APPENDIX A: SEVERITY INDEX

The severity index is based on California's ambient air quality standards, and is constructed somewhat analogously to the federal Pollutants Standards Index, as described in the U.S. Code of Federal Regulations (40 CFR Part 58, Appendix G). The idea is simply to assume that all relevant effects, such as health and visibility impairment or damage to plants and materials, have been taken into account in the setting of these standards. Hence, with respect to any one pollutant, the relative severity of an emission is measured by the fraction it contributes to the ambient concentration defining the air quality standard for that pollutant. For example, if all mobile sources contributed 0.07 ppm to a region's hourly average ozone, the California standard for which is .10 ppm, the severity of their combined ozone-producing emissions would be measured as 0.07/0.10 or 0.7. The total severity of an emission is found by summing its severities with respect to all the air pollutants to which it contributes.

Computing this index requires not only knowledge of the standards, but also a model of the relationship between emissions and ambient pollution concentrations. In this paper we consider four emitted pollutants and five ambient pollution standards. The emitted pollutants (with emissions E and severity D in parentheses) are:

$$\begin{aligned} P &= \text{Fine carbonaceous particulates} && (E_p, D_p) \\ SO_x &= \text{Sulfur oxides} && (E_s, D_s) \\ NO_x &= \text{Nitrogen oxides} && (E_n, D_n) \\ ROG &= \text{Reactive organic gases} && (E_r, D_r). \end{aligned}$$

The five ambient air quality standards apply to the following air pollutants (concentrations in parentheses):

PM10 = Fine particulate matter (C_p)
SO_4 = Sulfates (C_{so4})
SO_2 & PM10 = Sulfur dioxide and PM10 ($C_{pso2} = C_p \times C_{so2}$)
NO_2 = Nitrogen dioxide (C_{no2})
O_3 = Ozone (C_{o3}).

Note that the joint standard for SO_2 and particulates, based on a well-established synergism (Horowitz, 1982, p. 16), is accounted for in the same way as in the Pollutants Standards Index: by assuming that the standard establishes a degree of severity for the product of the two concentrations. However, for simplicity, we have used PM10 instead of total suspended particulates in this joint standard, reducing the assumed standard accordingly.

The specific assumptions are:

(i) Ambient concentrations of PM10 are proportional to the "total fine particulates" emissions as given by (E_p + $1.2E_s$); the rationale is that nearly all SO_x is emitted as SO_2, each gram of which produces 1.2 grams of particulate sulfate in the atmosphere (CARB, 1984, pp. 60–63). Hence:

$$C_p = a_p E_{tp} \tag{A1}$$

$$E_{tp} = E_p + 1.2E_s, \tag{A2}$$

where C_p is ambient PM10 concentration and E designates total emissions of a pollutant throughout the air basin.

(ii) Ambient concentrations of sulfates and of SO_2 are each proportional to SO_x emissions, with different proportionality constants:

$$C_{so4} = a_{so4}E_s \tag{A3}$$

$$C_{so2} = a_{so2}E_s. \tag{A4}$$

(iii) Ambient concentrations of NO_2 are proportional to all NO_x emissions:

$$C_{no2} = a_{no2}E_n. \tag{A5}$$

(iv) The ambient concentration of O_3 in each zone i is a function of basin-wide emissions of NO_x and ROG:

$$C^i_{o3} = f^i(E_n, E_r). \tag{A6}$$

Small changes ΔE_n, ΔE_r in these emissions produce changes in C_{o3} given by the linear term in a Taylor-series approximation:

$$\Delta C^i_{o3} = b^i_{o3,n}\,\Delta E_n + b^i_{o3,r}\,\Delta E_r. \tag{A7}$$

The two coefficients $b^i_{o3,n}$ and $b^i_{o3,r}$ are calculated by solving the equation with ΔE_n and ΔE_r set to the values used in each of two scenarios in Souten *et al.* (1981) (see Appendix B) and ΔC^i_{o3} set to the averaged resulting values for 26 and 27 June in that zone.

(v) For each pollutant, the concentration at every season or time of day rises or falls by the same proportion.

(vi) For each pollutant except ozone, the concentration at every location in the basin rises or falls by the same proportion.

(vii) The damage from an ambient concentration is proportional to the ratio of the concentration to the standard, for each of the following five standards: $\overline{C}_p$, $\overline{C}_{so4}$, $\overline{C}_{pso2}$, $\overline{C}_{no2}$, and $\overline{C}_{o3}$. Furthermore, the damages from these five ratios are additive, and in any particular location the amount of damage that occurs when any of the standards is reached is the same. In the case of ozone, this damage is allocated to zone i according to its fraction w^i of the basin's daytime population. This implies that total damage is proportional to:

$$D = \frac{C_p}{\overline{C}_p} + \frac{C_{so4}}{\overline{C}_{so4}} + \frac{C_p \cdot C_{so2}}{\overline{C}_{pso2}} + \frac{C_{no2}}{\overline{C}_{no2}} + \sum_i w_i \frac{C^i_{o3}}{\overline{C}_{o3}} \tag{A8}$$

By substituting eqns (A1)–(A7) into (A8), we can calculate relative severities as the partial derivatives of D with respect to emissions. Using (A1) and (A3)–(A5) to eliminate some of the proportionality constants, we can write these as:

$$D_p = \frac{1}{E_{tp}}\left(\frac{C_p}{\overline{C}_p} + \frac{C_p \cdot C_{so2}}{\overline{C}_{pso2}}\right) \tag{A9}$$

$$D_s = \frac{1.2}{E_{tp}}\left(\frac{C_p}{\overline{C}_p} + \frac{C_p \cdot C_{so2}}{\overline{C}_{pso2}}\right) + \frac{1}{E_{sox}}\left(\frac{C_{so4}}{\overline{C}_{so4}} + \frac{C_p \cdot C_{so2}}{\overline{C}_{pso2}}\right) \tag{A10}$$

$$D_n = \frac{1}{E_n}\left(\frac{C_{no2}}{\overline{C}_{no2}}\right) + \sum_i w_i\left(\frac{C^i_{o3}}{\overline{C}_{o3}}\right) b^i_{o3,n} \tag{A11}$$

$$D_r = \frac{1}{E_r}\sum_i w_i\left(\frac{C^i_{o3}}{\overline{C}_{o3}}\right) b^i_{o3,r}. \tag{A12}$$

Table A1 lists the data. The standards are those applying to California in 1985, using the averaging periods shown in the table. Ambient concentrations (of all but O_3) are taken to be the highest 24-hour or 1-hour average, as appropriate, observed at the downtown Los Angeles monitoring station during 1985. Emissions are those estimated for the South Coast Air Quality Management District for 1985.

Note that neither of the standards applying to sulfur was violated, though both were violated at monitoring stations further inland. Hence the proportionality assumption (ii), which implies that a given increase in concentration is just as damaging whether or not any particular threshold has been reached, is important. This assumption is supported by several lines of evidence. First, most epidemiological studies have failed to find thresholds (e.g. Lave and Seskin, 1977, p. 51), though some possible evidence is noted by Lipfert (1984, p. 208). Second, hypotheses of threshold existence have failed to hold up under scrutiny by four separate panels of the National Academies of Sciences and Engineering for four separate pollutants (NAS–NAE, 1974, pp. 6, 190, 366–367, 400). Third, even if thresholds exist for individuals, averaging over time, space, and people with varying sensitivities will tend to remove the threshold effects from aggregate population responses. See Small (1977, pp. 111–112) for further discussion.

The resulting index is

$$\text{Severity Index} = \text{P} + 4.80(\text{SO}_x) + 0.16(\text{NO}_x) + 0.23(\text{ROG}). \tag{A13}$$

Excluding the terms related to ozone, it would be P + $4.80(SO_x)$ + $0.22(NO_x)$; and excluding the terms related to NO_2 or ozone, it would be just P + $4.80(SO_x)$. Note that accounting for ozone *decreases* the coefficient of NO_x, indicating that on balance NO_x emissions decrease weighted ozone concentrations according to these simulations, though only slightly.

APPENDIX B: ASSESSING THE HEALTH BENEFITS DUE TO LESSENED AMBIENT OZONE

Emissions control strategies can be evaluated and compared in several ways. In the main body of the paper we rated them by using a severity index based on government-

Table A1. Data for severity index

Ambient concentrations	Averaging time	Standard† ($\bar{C}$)	Actual‡ (C)	Ratio ($C/\bar{C}$)
Fine particulates (PM10)	24 hr	50 μg/m³	146 μg/m³	2.92
Sulfates (SO_4)	24 hr	25 μg/m³	20 μg/m³	0.80
PM10 and SO_2 (pso2)	24 hr	(50 μg/m³)×(.050 ppm)	(146 μg/m³)×(.021 ppm)	1.23
Nitrogen dioxide (NO_2)	1 hr	0.25 ppm	0.27 ppm	1.08
Ozone (O_3)	1 hr	0.10 ppm	§	§

Emissions:‖	(E)
Fine particulates (p)	247.1 × 10⁶ kg/year
Sulfur oxides (s)	40.1 × 10⁶ kg/year
Nitrogen oxides (n)	344.3 × 10⁶ kg/year
Reactive organic gases (r)	412.7 × 10⁶ kg/year

†California ambient standard for 1985; except that for the particulate portion of the joint particulate and SO_2 standard, we have made the same substitution as was made in August 1983 for the particulate standard itself: namely, 50 μg/m³ PM10 instead of 100 μg/m³ total suspended particulates.

‡Maximum reading for downtown Los Angeles monitoring station in 1985. Source: SCAQMD, 1986, pp. 40, 41, 43, 45.

‖Source: SCAQMD, 1988, p. IV–5; PM10 were provided by the California Air Resources Board.

§Varies by zone.

mandated air pollution standards. Here we are more explicit about actual pollution effects: we estimate and place values on some of the health improvements that could be attributed to the lower ozone levels of each control strategy. This provides an alternative way to assess the effects of the ozone changes predicted by our exposure model; this is particularly important because recent research suggests that ozone may be more damaging than was suspected when the ozone standard was set.

Although the literature does not implicate ozone directly in mortality (in contrast to particulates and sulfates, the pollutants addressed in our earlier work), it does show that ozone elicits undersirable physical symptoms in humans, especially those engaged in heavy exercise (Goldstein *et al.* 1985). These symptoms include decrement in lung capacity, cough, chest discomfort, nose and throat irritation, headache, shortness of breath, and increased risk of asthma attack.

A brief sampling of recent laboratory and epidemiological studies illustrates some of the findings. Human responses of the kinds just mentioned have been observed in laboratories at ozone concentrations as low as 0.15 ppm (Kulle *et al.*, 1985, p. 36). Bonnet monkeys, exposed to ozone levels of 0.60 to 0.65 ppm for extended periods, developed such lung problems as inflammatory cells, narrowed bronchiolar airways, and permanent tissue stiffness, all changes known to be associated with fibrotic lung disease in humans (Raloff, 1986, p. 86). Ozone impairs the antibacterial defenses of rats, which resemble those of humans (Dungworth *et al.*, 1985, p. 527).

To determine the short-term ozone health effects induced by each of our emission-control strategies, we estimated pre- and postcontrol ozone levels at 23 points across the basin. We used these "before" and "after" levels in concentration–response functions that gave changes in the incidence of several health endpoints. Once we had estimated the changes in health status, we assigned values to these changes. Our data and methodology are discussed below.

Baseline ozone concentrations

For each month of 1985, the monthly average of daily one-hour maximum ozone readings was obtained from the South Coast Air Quality Management District for each of 23 stations (the other 6 stations used in the computer simulation discussed below are not maintained by SCAQMD, and are not essential to full coverage of the highly populated areas of the air basin).

Postcontrol ozone concentrations

We employ the same results of computer simulation of ozone formation, reported by Souten *et al.* (1981), that were used to extend the severity index. These results suggest that the combined reductions in NO_x and ROG emissions due to our control strategies would raise ozone levels in some locations and lower them in others. Because of this variation, it is necessary to calculate health benefits and disbenefits at many places and add them.

We chose two scenarios from Souten *et al.* (1981) to represent two quite different mixes of ROG and NO_x reductions, both from a baseline called "1987 Baseline + SIP" which was a projection of what emissions would be in 1987 with anticipated growth rates and controls envisioned in the State Implementation Plan, taking into account imperfect implementation. Scenario 1, called "Alternative Development Plus SIP," reduced baseline ROG and NO_x emissions by 1.8% and 3.1%, respectively. Scenario 2, called "1992 Baseline + SIP," reduced them by 2.1% and 2.5%. Souten *et al.* report the percentage change in maximum ozone reading predicted for each scenario at each monitor for each day of the two-day episode. We assume that these changes (averaged over the two days) follow a function, specific to that monitor, relating ozone reading to aggregate basin-wide emissions of ROG and NO_x. Since the changes are small, a first-order approximation to that function is adequate. It is linear in two unknown parameters, namely the elasticities of ozone reading at that monitor with respect to basin-wide ROG and NO_x emissions. By using our two observations on the resulting ozone changes, we can solve two linear equations for these two unknowns, yielding elasticities $a_R = -(1.240a^2 - a^1)/.804$ with respect to ROG, and $a_N = -(a^1 - .857a^2)/.957$ with respect to NO_x, where a^1 and a^2 are the percentage changes in ozone levels at that monitor from scenarios 1 and 2, respectively. These elasticities are then applied to the changes in basin-wide emissions of ROG and NO_x resulting from each of the pollution-control scenarios that we are studying, to obtain the predicted change in ozone concentration at that monitor. Note that this procedure does not account for the differences in geographical distribution of emissions between the various scenarios.

Concentration–Response functions

We estimate the effect of ozone concentrations on short-term ("acute") health problems. To do this we draw on the work of Krupnick (1986, pp. 5-39–5-45), who has developed from the ozone exposure literature a series of concentration–response functions that permit estimation of health end-points given ambient ozone levels. The health end-points we consider are asthma attacks, headaches, cough, chest discomfort, eye irritation, and restricted activity days. Each function is based on laboratory or epidemiological evidence, and each is nonlinear and hence capable of approximating threshold effects if the underlying data so warrant. We apply each function separately to the pre- and postcontrol ozone levels, for each month in 1985, to estimate the annual change in incidence of each health condition at each monitor.

Daytime populations

The 1985 resident populations of all incorporated cities and unincorporated areas in the basin are taken from the California Department of Finance (1986). Within each county, the population in unincorporated areas is first assigned equally to all the cities in that county. Then each South Coast Air Basin city, with the exception of Los Angeles, is assigned to the nearest monitor. For Los Angeles, portions of the population are assigned to nearby monitors within or outside the city by crude estimates from maps. In addition, we identified four areas of Los Angeles that were intermediate between the downtown Los Angeles monitor and another monitor, and assigned each of them the average between the two readings; populations of these areas are also estimated from maps.

The daytime population around each monitor is estimated from the resident populations assigned as just described and adjusted by the percentage net commuting inflow (on the basis of census journey-to-work data) for the largest city assigned to that monitor. The resulting assignments of Los Angeles's daytime population to monitors are: Downtown Los Angeles (1,347,080); Burbank (1,037,570); West Los Angeles (500,000); Long Beach (75,000); and averages between the Downtown Los Angeles monitor and the following four monitors: Lynwood (129,000), Lennox (125,000), West Los Angeles (250,000), and Pasadena (100,000).

Target populations as a percentage of daytime populations (e.g. number of people suffering from asthma) are as given in Krupnick (1986, p. 6–11).

Monetary values

The suggested amount that a typical individual would pay to avoid being afflicted by each health condition, taken from Krupnick (1986, p. 8-19), is listed in the footnotes to Table B1.

Results

The health effects results of the three control strategies are shown in Table B1. As with the severity index, this method of aggregating the effects of varying ozone changes across the basin leads to a tiny net *disbenefit* from the two control strategies that reduce NO_x emissions (methanol and cleaner diesel). The magnitude is small compared either to the control costs (shown in the last row) or, in the case of the methanol strategy, to the estimates we presented in earlier work of the value of mortality reductions due to particulate and sulfate removal, namely $21 million to $113 million.

It should be noted that taking into account long-term health effects might alter this calculation. A UCLA study of residents of high-oxidant Glendora and low-oxidant Lancaster in Southern California showed significantly more symptoms (cough, sputum production, wheezing, and chest illness) and weaker lung functions in the high-oxidant community (Detels *et al.*, 1979, 1981, 1987; Rokaw *et al.*, 1980). Great care was taken to minimize such confounding variables as prior respiratory illness and socioeconomic differences. A follow-up study five years later revealed markedly greater lung capacity decrements in the Glendora than in the Lancaster residents. This work is important not only because it followed subjects over time but also because it combined both laboratory and epidemiological analysis of human response to ambient ozone levels.

As a result of these and similar investigations, analysts have recently questioned the suitability of the federal ozone standard, which assumes that it is short-term exposure to high concentrations of ozone that causes damage. Now researchers are suggesting that long-term exposure to levels of ozone below the 0.12 ppm federal standard is harmful and cumulative (Sun, 1988).

Since our uninspiring results on ozone reduction depend on the rather uncertain simulation modeling of geographically varied ozone chemistry, we wondered whether the sheer magnitude of possible reductions would be significant if ozone were more simply related to NO_x emissions. To find out, we considered the following extreme example: suppose all ozone formation in the basin were strictly NO_x-limited, and that ozone concentrations were everywhere proportional to total NO_x emissions. Recalculating under

Table B1. Health effects results

	Low aromatic fuel	Particulate traps	Methanol with catalyst
Expected annual change in acute incidence†			
Asthma attacks	14	−88	141
Headache	357	−1,973	3,429
Days of coughing	277	−1,525	2,635
Days of chest discomfort	40	−226	40
Days of eye irritation	1,539	−6,509	13,574
Respiratory restricted activity days	1,637	−11,266	17,111
Value of acute incidence changes ($ millions/year)‡	−0.041	0.255	−0.409
Cost of control strategy ($ millions/year)§	.434	2.987	20.556

†Symptom days and restricted activity days were computed independently of each other. To avoid double counting, we considered each symptom day to result in a restricted activity day and valued it as such. Only symptom days in excess of the number of restricted activity days were valued as symptom days.

‡Each incident is valued at the middle of the three alternative valuations suggested by Krupnik (1986). These values are: asthma attack $25; headache $5; day of coughing $4; day of chest discomfort $6; day of eye irritation $5; respiratory restricted activity day $18.

§Calculated from Table 2, top row, assuming 4,432 buses.

these assumptions led to a value of ozone reduction of $1.3 million for clean fuel and $5.4 million for methanol. By far the dominant component was restricted activity days, of which the majority were caused by eye irritation. These numbers are significant in comparison with the costs of these control strategies; they could be decisive in a cost–benefit comparison (depending, of course, on the other measured benefits). Hence, the potential for substantial benefits from ozone reduction is there, but it will be realized only if there is a more direct relationship between ozone and NO_x than the one assumed in the SAI simulations used here.

Methanol Versus Gasoline: The Case of the Moving Target, by Linda Cohen. (Preliminary draft, January 30, 1989).

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Methanol Versus Gasoline: The Case of the Moving Target

Linda Cohen

Methanol is the current cornerstone of the federal government's alternative transportation fuel program. Methanol proponents claim that it satisfies a range of policy goals that underly federal energy policy. From an environmental perspective, early estimates projected that methanol use would result in considerably less smog in affected areas like the Los Angeles basin, owing to fewer NOx emissions and a favorable mix of carbon emissions. In addition, methanol made from natural gas feedstocks -- the current focus of research -- results in less carbon emissions than petroleum-based fuels, and hence favorable results for greenhouse pollution. While more careful modeling has to some extent moderated enthusiasm for methanol on environmental grounds, it remains a popular option for addressing pollution from automobile emissions.

Second, methanol is viewed as a step in the right direction towards satisfying energy disruption and security goals. One of the main lessons learned in the 1980s for energy policy was that considerable substitutes for petroleum products exist -- and are utilized -- for electricity generation, and direct industrial, manufacturing, and residential applications. Alternate fuels as well as conservation combined to decrease consumption of petroleum in the United States for these purposes from 17.019 quads in 1973 to 12.020 quads in 1987 -- a decrease of 30% over a time period when real GNP grew by nearly 40%.[1] On the other hand, demand for petroleum products for transportation purposes remains relatively inelastic. Despite impressive gains in average fuel efficiency of automobiles, petroleum use in the transportation sector increased from 17.821 quads in 1973 to 20.606 quads in 1987, and has increased as a percent of energy used in the transportation sector from 95.8% to 97.4%.[2] Transportation is viewed as the hardest oil-use nut to crack. Natural gas, the current preferred feedstock for methanol, is far less geographically concentrated in Persian Gulf countries than is petroleum. Indeed, large gas reserves exist in North America, as well as gratifying quantities within the United States. Furthermore, methanol can be made -- at both an environmental and pecuniary cost -- out of any carbon-based feedstock, including peat, garbage, and coal. In theory, the United States could be energy independent with a methanol-based transportation economy.

Finally -- and this is the focus of this paper -- methanol is viewed as the most economical of all the alternate transportation fuels currently under investigation, with total costs of a methanol-based transportation system projected to be only slightly in excess of the current gasoline-based

1. U.S. Department of Energy, Energy Information Administration, Monthly Energy Review, December 1987, p. 29 (consumption of energy by residential and commercial sector), p. 35 (energy input at electric utilities), p. 18 (GNP).
2. Ibid., p. 33.

system, and with relatively minor modifications needed for delivery systems and automobile engineering. Consequently, it is considered to be the most feasible short-term solution to the transportation problem.

This paper, however, argues, that methanol economics have been estimated in a vacuum: that projected costs vis-a-vis gasoline prices have in large part ignored energy economics and energy prices, and in particular those features of the energy market that give root to energy policy: that oil is an exhaustible resource, and that the cheapest supplies, as well as by far the most supplies, are found in the Persian Gulf. Specific attributes of oil markets suggest that public investment in alternative fuels, and in methanol in particular, may well yield large social benefits. However, it may need to be an extraordinarily expensive public undertaking.

Consider first the typical efficiency calculations for the development of methanol.[3] The first step is to calculate production costs for methanol, then, considering extra vehicle costs and distribution costs, to calculate a "break-even cost" for gasoline. The final step is to impute backwards the price of oil that supports the break-even gasoline price. At that price, methanol becomes a competitive fuel. For the policy analyst, this is where the problem begins: when is oil likely to reach the critical price? Calculating future oil price

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3. See, e.g., Barry McNutt, Jeffery Dowd, John Holmes, "The Cost of Making Methanol Available to a National Market," in <u>Methanol - Promise and Problems</u>, SP-726, Society of Automotive Engineers, Inc., February, 1988.

trends is recognized to be a risky business; nevertheless, some estimates are necessary to calculate a reasonable time-frame for the introduction of methanol. The time-frame usually used is mid- to late- 1990s, by which time current non-OPEC free world oil supplies will have declined to the point that OPEC will once more have an upper hand in pricing.

The problem however, is worse than the usual energy analysts' joke that they should never mention a price and date in the same publication. Attributes of petroleum markets imply that in deriving a methanol commercialization date, government policy is faced with a rock versus a hard place.

First, the rock: petroleum and natural gas are substitute inputs for a wide range of products and applications. We expect, and have observed over the past ten years, that as oil prices rise (and fall), so do natural gas prices. Thus, as oil approaches the break-even price, methanol feedstock prices, and hence methanol costs, rise as well. While substitutability is not particularly unique to energy inputs, the specific problem to consider is how large the elasticity of substitution is likely to be when oil prices increase. For example, a response to higher oil prices may be the development of CNG vehicles in Europe and Canada, where such programs are well underway. If succesfully commercialized, they will create enormous demand for natural gas, driving up prices substantially. Thus, the price of oil is a moving target for methanol commercialization, and the issue to consider is whether it moves too fast.

The hard place alternative derives from an apparently inconsistent possibility: that oil prices may be persistently too low. This observation relates to the way oil prices are set. Suppose for simplicity that OPEC has a monopoly on world oil supplies, as is presumed to some degree as a prerequisite for oil prices reaching interesting levels for methanol use. Marginal production costs for OPEC oil are trivial -- currently believed to be from less than a dollar per barrel for short-run marginal costs to two dollars a barrel when exploration and development costs are prorated and included (long-run marginal production costs). The price that a monopolist sets for its exhaustible resource depends, then, on the user costs, which depend on what demand for the resource is now and in the future. The introduction of an alternative fuel increases demand elasticity for oil, and hence changes the optimal price path. Faced with such a "backstop technology", a monopolist is expected to limit price, or to lower prices (depending on production costs and demand elasticity) so that its stock of oil is depleted before the new technology is introduced. In other words, we can ask the following reasonable question: will the Saudis sit there like a lox while their market power is eroded by the introduction of methanol, or will they take action and lower prices?

The answer to this question is extremely complicated. Factors that enter into the calculation include the extent to which methanol threatens the market share of OPEC: the U.S. currently consumes one-third of the free world's oil and onehalf of its gasoline. OPEC response depends on the fraction of the U.S. market lost, and potential development in other countries as well. In addition, it depends on pricing respoonse given the threat of a backstop technology (methanol), which may be nil if short-run elasticity is low, versus pricing response given the actuality of a backstop technology. In the worst case for commercial development of methanol, OPEC might limit price for years, so that the breakeven price isn't reached until well into the next century. However, they would only do so if the investment in methanol vehicles, plants, and distribution looks very imminent. Otherwise, there is no need to limit price oil. The investment in alternate fuel technology is always economical ex ante, and never ex post. The conclusion is that methanol commercialization is a moving target for oil prices.

While private commercialization is impossible under these circumstances, as private benefits are zero, social benefits are large (and do not accrue solely to the United States): if the alternate technology isn't available, OPEC won't bother to limit price. The public expense is consequently enormous, with government bearing the entire cost of making the alternative fuel a viable, but never-used, alternative.