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ABSTRACT

We extend a previous cost-effectiveness analysis of methanol versus 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.

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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 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. In order 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 non-desert parts of San Bernardino and Riverside Counties. These baseline conditions include use of low-sulfur diesel fuel (0.05 percent 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 8 percent per year. We assume the maintenance requirements that Weaver et al. estimate for this low-sulfur

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 (mi/gal)	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
SO4	0.026	0.026	0.080	0.000
SO2	0.836	0.836	0.809	0.000
ROG	4.550	3.867	1.365	1.310
NOX	26.100	23.229	26.100	13.600

fuel, but we do not adopt the increased engine life and lower aromatic content of fuel which 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. (1988, Table 9) and a conversation with Jeff Alson and Tom Baines, we assume that formaldehyde emissions are 7.5 percent 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. 11). 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 <u>not</u> analyzing the effects of lowering the sulfur content. Both the 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 percent, 15 percent, and 11 percent, 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 percent, and reduces carbonaceous

particulates by 90 percent (Small, 1988, p. 12) and NO_x by 70 percent (roughly in the middle of a range of 50-90 percent 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 U.S.), and fuel economy of 1.81 mi/gal, making it 7 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. In order not to be too optimistic, we assume that emission of each pollutant is equal to the higher of (a) the early in-use chassis measurements for the M.A.N. Golden Gate Transit bus (Alson, 1985, Table 5); and (b) the engine test of the Detroit Diesel engine planned for Los Angeles (Alson et al., 1988, Table 11), with the standard conversion factor of 3 brake-horsepower-hours per mile. These assumptions entail reductions in PM10, ROG, and NO_x of 96 percent, 71 percent, and 48 percent, 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., 1988, p. 12).

Fuel prices are very important in comparing methanol with other strategies. We adopt highly uncertain assumptions that make methanol 56 percent 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 effect 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 is 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 ratio 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):

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

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 equation (1), 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

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 (a)	16.9	49.4	97.5
Cost-Effectiveness (b)	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 (a)	14.7	30.6	78.6
Cost-Effectiveness (b)	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 (a)	15.1	37.1	81.7
Cost-Effectiveness (b)	1.30	3.63	11.35
Incremental Cost-Effectiveness	1.30	5.24	17.77

(a) This is the percentage reduction in the contribution of transit buses to the index.

(b) 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.

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 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.)

The table 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 the table shows that the gap incremental

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 2 percent according to 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 expressed belief that control of ROG "is generally the most promising strategy for reducing ozone levels" (Alson et al., 1988, p. 3), 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).[1] There is some controversy about

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

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 (1.13 versus the value of 1.08 shown in Table A-1), and it is only this ratio which 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. (1988), 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.

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:

P	Fine Carbonaceous particulates	(E_p, D_p)
SO_x	Sulfur oxides	(E_s, D_s)

NO_x	Nitrogen oxides	(E_n, D_n)
ROG	Reactive hydrocarbons	(E_r, D_r)

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 particulate" 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:

$$C_p = a_p E_{tp} \qquad (A.1)$$

$$E_{tp} = E_p + 1.2E_s \qquad (A.2)$$

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:

$$C_{so4} = a_{so4}E_s \tag{A.3}$$

$$C_{so2} = a_{so2}E_s \tag{A.4}$$

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

$$C_{no2} = a_{no2}E_n \tag{A.5}$$

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

$$C^i_{o3} = f^i(E_n, E_r). \tag{A.6}$$

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

$$\Delta C^i_{o3} = b^i_{o3,n}\Delta E_n + b^i_{o3,r}\Delta E_r. \tag{A.7}$$

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:

$$D = \frac{C_p}{\overline{C_p}} + \frac{C_{so4}}{\overline{C_{so4}}} + \frac{C_p \cdot C_{so2}}{\overline{C_{pso2}}} + \frac{C_{no2}}{\overline{C_{no2}}} + \sum_i w_i \frac{C^i_{o3}}{\overline{C_{o3}}} \qquad (A.8)$$

By substituting equations (A.1) - (A.7) into (A.8), we can calculate relative severities as the partial derivatives of D with respect to emissions. Using (A.1) and (A.3) - (A.5) to eliminate some of the proportionality constants, we can write these as:

$$D_p = \frac{1}{E_{tp}} \left(\frac{C_p}{\overline{C_p}} + \frac{C_p \cdot C_{so2}}{\overline{C_{pso2}}} \right) \qquad (A.9)$$

$$D_s = \frac{1.2}{E_{tp}} \left(\frac{C_p}{\overline{C_p}} + \frac{C_p \cdot C_{so2}}{\overline{C_{pso2}}} \right) + \frac{1}{E_{sox}} \left(\frac{C_{so4}}{\overline{C_{so4}}} + \frac{C_p \cdot C_{so2}}{\overline{C_{pso2}}} \right) \qquad (A.10)$$

$$D_n = \frac{1}{E_n} \left(\frac{C_{no2}}{\overline{C_{no2}}} \right) + \sum_i w_i \left(\frac{C^i_{o3}}{\overline{C_{o3}}} \right) b^i_{o3,n} \qquad (A.11)$$

$$D_r = \frac{1}{E_r} \sum_i w_i \left(\frac{C^i_{o3}}{\overline{C_{o3}}} \right) b^i_{o3,r} \qquad (A.12)$$

Table A-1 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 one-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).

Table A-1. Data for Severity Index

Ambient Concentrations:	Averaging Time	Standard[a] ($\bar{C}$)	Actual[b] (C)	Ratio ($C/\bar{C}$)
Fine Particulates (PM10)	24 hr.	50 ug/m^3	146 ug/m^3	2.92
Sulfates (SO_4)	24 hr.	25 ug/m^3	20 ug/m^3	0.80
Particulates and SO_2 (pso2)	24 hr.	(100 ug/m^3) x(.050 ppm)	(146 ug/m^3) x(.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	(d)	(d)

Emissions:[c]	(E)
Fine particulates (p)	247.1 x 10^6 kg/year
Sulfur oxides (s)	40.1 x 10^6 kg/year
Nitrogen oxides (n)	344.3 x 10^6 kg/year
Reactive organic gases (r)	412.7 x 10^6 kg/year

[a]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 ug/m^3 PM10 instead of 100 ug/m^3 total suspended particulates.

[b]Maximum reading for downtown Los Angeles monitoring station in 1985. Source: SCAQMD (1986), pp. 40, 41, 43, 45.

[c]Source: SCAQMD (1988), p. IV-5; PM10 were provided by the California Air Resources Board.

[d]Varies by zone.

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, pp. 6, 190, 366-7, 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

$$\text{Severity Index} = P + 4.80(SO_x) + 0.16(NO_x) + 0.23(ROG) \quad . \quad (A.13)$$

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 <u>decreases</u> 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-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 undesirable physical symptoms in humans, especially those engaged in heavy exercise (Goldstein, 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 very 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, 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, 1987, p. 86). Ozone impairs the antibacterial defences of rats, which resemble those of humans (Dungworth, 1985, p. 527).

In order to determine the short-term ozone health effects induced by each of our emission-control strategies, we estimated pre- and post-control 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 six 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).

Post-Control 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 percent and 3.1 percent, respectively. Scenario 2, called "1992 Baseline + SIP," reduced them by 2.1 percent and 2.5 percent. 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 Alan 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 resricted 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 post-control ozone levels, for each month in 1985, in order 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 notes to Table B-1.

Results. The health-effects results of the three control strategies are shown in Table B-1. 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.

TABLE B-1. Health Effects Results

	Low-Aromatic Fuel	Particulate Traps	Methanol with Catalyst
Expected Annual Change in Acute Incidence[a]			
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 ($millions/year)[b]	-0.041	0.255	-0.409
Cost of Control Strategy ($millions/year)[c]	.434	2.987	20.556

[a]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.

[b]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.

[c]Calculated from Table 2, top row, assuming 4,432 buses.

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 ppmn 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 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.

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