# AIR QUALITY BENEFITS OF ALTERNATIVE FUELS

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## I. Introduction and Background

This paper summarizes EPA's current best estimates of the potential for air quality improvements available if various alternative fuels scenarios were implemented. Included in this evaluation are methanol (for current technology, flexible-fueled, and advanced technology vehicles), compressed natural gas (CNG), and blends of gasoline with ethanol, methanol, and methyl tertiary butyl ether (MTBE). The primary air quality concerns relate to ozone and carbon monoxide (CO); thus the motor vehicle pollutants which receive most emphasis here are volatile organic compounds (VOCs) and CO.

### A. Health and Welfare Impacts

The National Ambient Air Quality Standard (NAAQS) for ozone is presently a maximum one hour level of 0.12 ppm, not to be exceeded more than three times in a three-year period. Several factors were cited as a basis for setting the current ozone standard. Among these were: (1) ozone is a pulmonary irritant that affects respiratory mucous membranes as well as other lung tissue and impairs respiratory function at levels as low as 0.15 ppm for exercising persons, (2) elevated numbers of asthma attacks occur when peak hourly ozone concentrations reach about 0.25 ppm, (3) increased susceptibility to bacterial infection is noted in laboratory animals exposed to ozone plus a bacterial challenge, (4) premature aging symptoms have been reported in rabbits, (5) apparent synergistic effects exist on pulmonary function from exposure to 0.37 ppm ozone and 0.37 ppm sulfur dioxide, and (6) the adverse health effect threshold could not be identified with certainty. Since the ozone NAAQS standard was established in the late 1970s, a number of new health studies have been completed confirming these earlier findings and showing that alterations in pulmonary function can occur for healthy adults and children during exercise at levels as low as 0.12 ppm. EPA is presently reviewing these new studies and determining whether a change should be proposed for the ozone standard. The EPA Science Advisory Board has supported an EPA staff recommendation that a revised one hour ozone standard be somewhere in the range of 0.08 to 0.14 ppm based on this new information.

In addition to these health effects, ambient ozone has been shown to cause crop yield reductions, forest injury, and damage to materials such as rubber and dyes. There is no known threshold for these welfare effects.

The NAAQS for carbon monoxide (CO) was established in 1971 as a 9 ppm eight hour average and 35 ppm one hour average not to be exceeded more than once a year. In 1980, EPA proposed retaining the eight hour standard and tightening the one hour

standard to 25 ppm but this proposal has not been finalized. The 1971 NAAQS was based on work showing 2 to 3 percent carboxyhemoglobin (which is the product formed from reaction of carbon monoxide absorbed into the lungs and hemoglobin which carries oxygen in the blood stream) impairs the ability to discriminate time intervals by affecting the central nervous system. Since then, further work has shown this type of effect occurs only at higher carboxyhemoglobin levels.

However, newer studies (some of which have been recently disputed) show that 2.7 to 3 percent carboxyhemoglobin levels shorten the time before the onset of pain for heart patients with angina pectoris (a heart disease associated with decreased oxygen to the heart). A 1984 EPA reevaluation of available health information indicates that chronic angina patients (who are presently viewed as the most sensitive group for carbon monoxide exposure) show aggravation of pain at levels of 2.9 to 4.5 percent carboxyhemoglobin. Based on this evidence, EPA has decided to retain the present carbon monoxide NAAQS although high priority has been given to obtain additional data on angina patients so that the NAAQS can be reevaluated as needed; EPA expects to be able to propose any necessary changes in the carbon monoxide NAAQS based on the new angina studies in about 3 years.

Other groups at risk for carbon monoxide exposure include fetuses and young infants, the elderly (especially those with compromised cardiopulmonary functions), younger individuals with severe cardiac or respiratory disease, and individuals with genetically unusual forms of hemoglobin that affect oxygen carrying capacity.

B. Contribution of Mobile Sources

Since alternative fuels strategies would primarily affect motor vehicle emissions, it is important to estimate the magnitude of the emissions contribution of motor vehicles to overall urban emissions under likely scenarios. For CO, more than two thirds of all emissions today come from motor vehicles (see Figure 1); for many urban areas the percentage is 80 percent or more. This fraction will continue to decrease in the foreseeable future as the current Federal Motor Vehicle Control Program (FMVCP) reduces CO from motor vehicles, since newer vehicles emit CO at much lower levels than the older vehicles they are replacing.

For VOC (and hence ozone), motor vehicles play a significant role, although somewhat smaller than for CO (see Figure 2). As Table 1 shows, the mobile source VOC emission fraction also varies considerably from city to city. As with CO, the VOC mobile source fractions will decline in the near term as newer, cleaner cars replace older ones. In any event, the contribution of vehicles to overall VOC at any given time is a key factor in assessing the potential impact of local or national fuel-based emission control programs.

## II. Attainment Status and Projections

Non-attainment of the National Ambient Air Quality Standards (NAAQSs) for ozone and CO is widespread and, especially for ozone, is likely to remain so into the foreseeable future. Figures 3 and 4 show how average ambient CO and ozone levels, respectively, have exceeded the NAAQSs in recent years. For ozone, 76 areas are currently in violation of the standard, 11 of which are in California. For CO, 81 areas are in violation, 10 in California. Tables 2 and 3 present for the 30 most seriously affected cities for each pollutant the 1983-1985 air quality design values and the emission reduction required for attainment.

EPA has projected how emission control programs currently in place or in the process of being implemented will affect the status of ozone and CO non-attainment. Table 4 shows projected numbers of non-attainment areas for the base year and for 1990, 1995, 2000, and 2010.

It can be seen from Table 4 that while the magnitude of the non-attainment problem is similar for both ozone and CO at the present time, projections of future non-attainment for the two pollutants diverge substantially. The CO attainment situation is expected to improve significantly through the end of the century due primarily to fleet turnover, as new vehicles with low CO emission factors continue to displace older vehicles with much higher emission factors. We expect that by 1995 approximately 80 to 90 percent of the urban areas now in non-attainment for CO will move into attainment. In comparison, less than half of the urban areas now in non-attainment for ozone are projected to comply by 1995. Of even greater concern is the fact that, as Table 4 shows, the ozone non-attainment problem is expected to begin to worsen in the late 1990s. There are several reasons for this: 1) the relative improvement due to newer vehicles is less for VOC than for CO; 2) vehicle miles traveled in urban areas is expected to continue to increase in the future, thus offsetting and at some point overwhelming the reduction in the emissions of each car; and 3) it is expected that VOC emissions from stationary sources will increase due to general economic growth.

## III. Air Quality Impact of Alternative Fuels

### A. Vehicle Emissions Effects

Table 5 presents EPA's estimates of potential reductions in VOC, CO, and NOx emissions for several types of light-duty vehicles operating on methanol, CNG, and oxygenated blends. These are per-vehicle estimates and appear as comparisons to gasoline vehicles meeting current EPA emission standards. For this analysis EPA assumes that light-duty gasoline trucks operated on alternative fuels would exhibit similar emission reductions.

1. Methanol

Methanol has long been considered to be an excellent motor vehicle fuel. Its simple molecular structure, high octane, wide flammability limits, high flame speed, and low flame temperature result in a fuel that can be burned in a very clean and efficient way relative to petroleum fuels. Interest in methanol vehicles has grown significantly in the last few years, and several vehicle demonstrations are in progress throughout the United States. Because methanol is such a different (and generally superior) fuel than gasoline, it is helpful to distinguish between two types of methanol vehicles -- current technology and advanced technology methanol vehicles.

Current technology methanol vehicles utilize engines that are very similar to engines used in today's gasoline vehicles, with modifications to allow the engine to operate well, but not optimally, on a blend of 85 percent methanol, 15 percent gasoline (M85). These are the types of methanol vehicles currently involved in demonstration programs, and these vehicles would have emissions and efficiency characteristics very similar to flexible fuel vehicles (FFVs) when FFVs are operated on M85. Such vehicles generally have emissions similar to those of gasoline vehicles, with the primary difference being the reactivity of the VOC emissions (this will be discussed in much greater detail later in this section). Because methanol has a low flame temperature, it has been claimed that current technology methanol vehicles should reduce NOx emissions. We do not believe this to be the case. Vehicle emissions are a function of many factors, including fuel type, engine type, emission control system design, etc. With current NOx emission standards, manufacturers will likely trade off methanol's low-NOx characteristic to gain other benefits such as fuel economy, performance, or a less expensive catalytic converter.

Advanced technology methanol vehicles would be designed and optimized specifically for methanol fuel. Such an engine should include, at minimum, high compression, lean combustion, advanced fuel injection, and an emission control system optimized for formaldehyde reduction. From an emissions perspective, the most interesting aspect of methanol fuel is its potential to operate successfully under lean burn conditions. Gasoline engines cannot operate at high air-fuel ratios both because of engine misfire and because of high NOx emissions (the NOx reduction function of the catalytic converter cannot operate well at high exhaust oxygen levels). Methanol's fuel properties suggest that methanol can operate at very lean conditions while still maintaining good driveability. If NOx emissions can be maintained below EPA standards without catalytic reduction, then significant benefits in terms of fuel consumption and CO emissions can be achieved through lean combustion. Again, however, we believe

that there will not be NOx benefits with advanced technology methanol vehicles, although it is methanol's low-NOx characteristic that permits lean combustion with corresponding CO and fuel economy benefits.

With regard to VOC emissions, estimating potential reductions is complicated by the fact that the composition of methanol VOC emissions is fundamentally different from that of gasoline vehicles. In its August 29, 1986 Notice of Proposed Rulemaking to establish emission standards for methanol vehicles, EPA has considered this problem in detail and determined that the organic compounds of primary concern are methanol, formaldehyde, and non-oxygenated hydrocarbons (HC). With gasoline vehicles, the organic compounds are primarily non-oxygenated HC. As mentioned previously, the various components of methanol vehicle organic emissions have differing tendencies to form ozone; that is, different degrees of reactivity. Thus, in order to evaluate the potential ozone reductions associated with methanol vehicles, EPA has developed a methodology for estimating the relative reactivities of the different emission components and using them to weight the expected emissions. The weighted sum can be compared to the organic emissions of a gasoline vehicle. The resulting ratio represents the ozone-producing potential of a methanol vehicle relative to that of a gasoline vehicle.

This type of analysis has two key elements. First, atmospheric photochemistry modeling is required in order to develop reactivity factors for the relevant emissions. Second, emission factors must be estimated. The following discussion presents these analyses as well as important caveats that must accompany their use. It must be emphasized that this methodology presented below is still under development and should be considered very preliminary at this time.

a. Relative Reactivity

Various methods can be used to characterize the relative reactivities of organic compounds. One common approach is to measure the reaction rates of the organic compound of interest when introduced into a chamber containing one or two common atmospheric reactants such as hydroxyl radicals or NOx. This approach is often useful for ranking organics in terms of reactivity but may be less reliable when attempting to establish a more absolute quantification of a compound's reactivity in a real urban environment.

Within the last few years several computer simulation studies have been performed to model the effect of mobile source methanol fuel substitution in a number of urban areas. These studies simulate the air chemistry and transport within the urban area, and account for the entrainment and dilution of local pollutant inventories into the airshed. It should be noted that none of the studies actually utilized emission

factors for methanol vehicles which EPA believes to be wholly consistent with the available data base, nor did they model substitution scenarios for methanol vehicles similar to the scenarios developed for this document, as presented in Section IV.B. below.

It is also extremely important to bear in mind that the reactivity factors will vary not only city-to-city, as the data in Table 6 demonstrate, but also within a given city from day to day. This is because the chemistry of ozone formation is sensitive to changes in the makeup of the pollutant inventory, variations in the boundary and aloft conditions (related to inter-city transport), shifts in the wind field (the less wind, the less flushing of the airshed, the higher the local pollutant concentrations, and the more ozone is likely to be formed), solar intensity, and numerous other transient factors.

EPA is not confident that the available modeling base is sufficient to adequately characterize methanol and formaldehyde reactivity on a national average; a city specific characterization is even more problematic. Great care must be exercised in using the results from past modeling as the basis for regulatorily binding decisions which presume environmental effects that may or may not actually occur.

Nevertheless, the data may and should be used to evaluate relative policy options. EPA therefore has analyzed the results of these studies and developed reactivity factors from them for each relevant pollutant. To date, three modeling studies have been performed which can be used in the desired fashion. One study modeled the Los Angeles airshed, under contract for ARCO and other companies. Another study modeled the Philadelphia airshed, under contract for EPA. Finally, Ford Motor Company did a study which, though less detailed than the previous two, considered methanol substitution in 20 ozone non-attainment cities. Using the results of these studies, EPA has developed reactivity factors for methanol and formaldehyde relative to non-oxygenated HC. These factors are presented in Table 6.

b. Emission Factors--Current Technology Methanol Vehicles

Emission testing has been performed on many different prototype methanol vehicles by a number of different organizations. EPA has evaluated the methodology used by these organizations and, as appropriate, included the data in a master data base. This data base was discussed briefly in the Regulatory Support Document that accompanied the Notice of Proposed Rulemaking (NPRM) for methanol vehicle emission standards. Since that time the data base has been expanded and will be presented in detail with the Final Rule. For the purposes of this discussion only the relevant details of the analysis are presented.

To develop exhaust emission factors for methanol vehicles, the following approach was used. EPA proposed and will likely finalize CO and NOx standards for methanol vehicles equivalent to those for gasoline and diesel vehicles. Therefore, the data base was sorted to pull out all emission tests where CO and NOx met the current standards. For these vehicles, a typical ratio of methanol to formaldehyde was established. A methanol to hydrocarbon ratio was developed using much more limited data (the HC measurement techniques used by most researchers to date were found inappropriate for EPA's purposes). These ratios were then applied to the proposed organic emission standards for methanol vehicles in order to determine likely certification exhaust emissions of methanol, formaldehyde, and HC (the proposed standards allow roughly the same amount of carbon to be present in methanol vehicle VOC emissions as is allowed by current standards for gasoline and diesel HC emissions). In order to estimate in-use exhaust emissions of methanol vehicle VOCs, offsets were developed based on the in-use performance of gasoline vehicles. A similar procedure was used with regard to evaporative emissions, both for carbureted and fuel-injected engines. The resulting in-use exhaust and evaporative emission factors are given in Table 7.

The present analysis accounts for refueling emissions based on very recent and as yet unpublished test data performed by EPA's Office of Research and Development (ORD). This testing involved only one methanol vehicle. The results are given in Table 7 and assume that no refueling controls are applied to 49-state methanol vehicles and that refueling controls in California are as effective for methanol vehicles as for gasoline vehicles (86% control efficiency is assumed).

c. Emission Factors--Advanced Technology Methanol Vehicles

As mentioned earlier, there is great potential for emissions and efficiency benefits with engines designed and optimized for methanol fuel. The data for vehicles representative of this future technology are extremely limited. The one prototype vehicle that incorporates many of the potential design features of an advanced technology methanol vehicle is a Toyota Carina that is currently undergoing evaluation by EPA. Thus, the emission factors given in Table 7 are based on engineering judgment as well as preliminary test results from the Carina obtained by both Toyota and EPA. In general, methanol-optimized technology is expected to lower the overall mass of organic emissions and to selectively lower formaldehyde levels. Additionally, these vehicles must operate on pure methanol (M100) in order to achieve the maximum environmental benefit. With M100, relatively less HC (which is related to fuel gasoline content) and more methanol are found in the emissions. The numbers given in Table 7 reflect these effects. With regard to

evaporative emissions, the estimates are based on the assumption that all organic evaporants are methanol (since there is no HC in the fuel) and that the vehicles emit only the equivalent of 1.0 gram per test, which is half of the standard currently applicable. Refueling emissions are again based on the recent ORD test of only one vehicle, this time operated on M100.

2. Compressed Natural Gas (CNG)

Compressed natural gas (CNG) consists primarily of methane but also contains smaller quantities of other compounds such as ethane and propane. Sharing many of the same fuel characteristics of methanol (simple molecular structure, high octane, ability to combust under lean conditions, etc.), CNG has long been considered to be an environmentally attractive engine fuel. Two characteristics of CNG distinguish it from methanol. The most obvious difference is that CNG is a gaseous fuel. This is an advantage in that it is an excellent cold starting fuel, and thus has the potential to significantly reduce cold start emissions. Its gaseous nature also poses problems, however, in that it is much more difficult to store large amounts of energy onboard the vehicle, and engine power output is usually decreased. A second difference is CNG's relatively high flame temperature, which results in relatively high NOx emissions. This characteristic makes the possibility of a high compression, lean burn CNG engine which could meet stringent NOx standards without the aid of a NOx reduction catalyst more difficult than for methanol (such catalysts are not effective under the high-oxygen regime of lean burn combustion). As with methanol, it is helpful to distinguish between near-term current technology and the potential for optimized, advanced technology CNG vehicles.

a. Relative Reactivity

Since methane is the primary component of CNG fuel, a high percentage of the fuel-related emissions from CNG vehicles is methane. Test data from EPA and Ford suggest that typically between 80 to 90 percent of the hydrocarbon emissions from CNG vehicles are methane, with small amounts of ethane, propane, and other hydrocarbon compounds. Methane has long been considered to be photochemically nonreactive, so much so that the California Air Resources Board permits vehicle manufacturers to certify to nonmethane hydrocarbon standards and EPA has proposed such standards in the past, though the change was not finalized. Since the relative reactivity of CNG vehicle exhaust has not been modeled as extensively as that of methanol vehicle exhaust, the practice has been to assume that the methane component of CNG emissions has zero reactivity while the remaining hydrocarbons have an overall reactivity similar to gasoline vehicle hydrocarbons. Though this methodology is rather simplistic, it has been widely practiced and will be used in this analysis as well.

b. Emission Factors--Dual-Fuel Retrofit CNG

Most of the vehicles currently operating in the U.S. on CNG fuel are gasoline vehicles retrofitted with a conversion kit to allow the vehicles to operate on either CNG or gasoline. It is very difficult to estimate the overall emissions impacts of such vehicles for several reasons: 1) there has been little reliable emission testing performed, particularly on conversions of recent computer-controlled vehicles, 2) the performance of conversion kits can vary greatly depending on kit manufacturer, the expertise of the installer, the quality of maintenance, etc., 3) the fact that the vehicle can operate on either CNG or gasoline means that overall emissions depend on the fuel that is actually used, and 4) the conversion process itself sometimes interferes with the gasoline combustion process leading to increased emissions on gasoline. It should also be obvious that an engine that must be able to operate on fuels as different as CNG and gasoline cannot be optimized for either fuel.

EPA has recently examined emissions data from dual-fuel CNG retrofit vehicles tested by the California Air Resources Board, Colorado Department of Health, the Canadian Ministry of Energy, Mines and Resources, as well as EPA. As might be expected in view of the caveats listed above, there is considerable scatter in the data. Nevertheless, there are clear trends in the data, and ranges for likely emission impacts compared to gasoline vehicles are shown in Table 5. Nonmethane exhaust HC were typically 40 to 60 percent lower, which when combined with zero evaporative and refueling HC, lead to overall VOC reductions of 50 to 80 percent compared to gasoline vehicles. CO emissions were typically between 50 and 90 percent lower on CNG. NOx emission impacts of CNG fueling are quite variable, ranging from small decreases to large increases. It should also be noted that there are usually significant power losses associated with dual-fuel CNG retrofit kits, as well as somewhat reduced energy efficiencies.

c. Emission Factors -- Advanced Technology CNG

As with methanol, it is possible to take full advantage of CNG's fuel properties only by utilizing it in engines dedicated and optimized for its use. To date there has been very little work done to identify the proper design of such an engine, so the overall emissions impacts are very speculative. The primary issue that needs to be addressed is whether it will be possible to reap the CO and efficiency benefits of high-compression/lean-burn operation while maintaining NOx emissions within acceptable levels (again, the NOx reduction part of the catalytic converter would not be effective under lean burn conditions). CNG, with a high flame temperature, faces a more difficult task in this regard than methanol (which is an inherently low-NOx fuel due to its low flame temperature).

Ford Motor Company has built several dedicated CNG Ranger pickup trucks that are being operated by gas utilities throughout the country. Ford's zero-mile emission data showed a 40 percent decrease in exhaust HC, a 99 percent decrease in CO, and an 80 percent increase in NOx emissions. The NOx emissions level of 1.96 grams per mile was still slightly below the applicable light-duty truck NOx standard of 2.3 gpm, but beginning in 1988 the light-duty truck NOx standard drops below that value. Thus, it is still unclear whether NOx levels will be a problem on advanced technology CNG vehicles. EPA is currently in the process of obtaining a high-mileage CNG Ranger for emissions testing. For the time being, Table 5 assumes that advanced technology CNG vehicles will have emissions characteristics similar to those of dual-fuel vehicles when the latter are operating on CNG.

3. Oxygenated Blends

EPA's estimates of the emissions effects of switching gasoline vehicles to various oxygenated blends are also presented in Table 5. The VOC estimates were developed by modeling all the factors that affect hydrocarbon emissions (evaporative emissions, exhaust emissions, and the effect of commingling of fuels when straight gasoline and a blend are mixed in-use), summing these factors, and applying reactivity weightings so that all results appear on an ozone-equivalent basis.

CO reductions are estimated on the basis of tests collected on vehicles tested at low and high altitudes. The high-altitude data on non-catalyst and open-loop vehicles, developed by the Colorado Department of Health, are similar to the available low-altitude data; EPA has chosen values representative of the entire body of data. The closed-loop reduction presented, however (10 percent for a 3.7 percent oxygen fuel), is smaller than the test data indicated. Colorado and EPA have reduced that value to reflect the fact that theory and very limited test data indicate that the CO benefit may actually be less if the recently-introduced "adaptive-learning" closed-loop technology becomes widespread (such systems are designed to optimize the individual vehicle's operation for the specific climate, altitude, and usage patterns, theoretically reducing the potential CO reductions available from blends to near zero). While older vehicles operated at high altitude emit more CO than at lower altitudes (the thinner air causes a richer air-fuel mixture), the percentage reduction available through blend usage is about the same regardless of altitude.

The estimated NOx increases were developed by EPA from the low-altitude data that were a part of the data base for the CO reduction estimates above.

The potential improvement in CO emissions over straight gasoline that would result from the use of blends in current gasoline vehicles could be significant. However, ozone-equivalent VOC emissions show no clear benefit; if they were sold at a Reid Vapor Pressure (RVP) higher than for gasoline during the summer, ethanol and methanol blends would cause considerably more ozone. NOx emissions are worse in all scenarios.

B. Potential Impact on Attainment

1. Overview

In order to analyze how the current ozone and CO attainment situation might respond to large scale alternative fuel initiatives, EPA has defined and evaluated three major scenarios.

a. Methanol Scenarios

In the first two scenarios a significant number of methanol-fueled passenger cars and light-duty trucks would be introduced into three urban ozone non-attainment areas--Los Angeles, New York, and Washington, DC. Fleetwide composite emission estimates are made for the short-term (assuming current technology and flexible-fueled vehicles) and in the longer term (assuming advanced methanol technology). The analysis then translates the emission estimates into overall estimates of how ambient ozone levels would be affected. Since, as stated above, compressed natural gas (CNG) vehicles would probably have ozone-equivalent VOC emissions falling somewhere between current technology/FFV methanol vehicles and advanced technology methanol vehicles, qualitative conclusions about CNG vehicles can also be drawn from this analysis.

In these methanol-vehicle scenarios, the effect on ozone is roughly proportional to the number of methanol vehicles assumed to be introduced in an area. Therefore, in order to translate potential per-vehicle VOC reductions into percent ozone reductions in an area, EPA needed to make assumptions of how many methanol vehicles might be introduced into an area each year, and for how many years this replacement would occur. In the current technology/FFV scenario, introduction of vehicles begins in 1991 (the earliest projected date for vehicle availability of any manufacturer) and continues through 1995; advanced technology vehicles would require additional time for development and are assumed to be introduced from 1995 through the year 2000. Methanol vehicles are assumed to replace all fleet vehicles first, with the remainder replacing vehicles in general use. Table 8 presents the numbers of vehicles assumed to be replaced each year and the total number reached by 1995 and by 2000.

Based on a comprehensive 1978 study of U.S. vehicle distribution, EPA has estimated that there are 80,000 vehicles sold each year in Los Angeles for use in vehicle fleets of 10 or more vehicles (very small fleets would not be any more likely to utilize alternative fuels than general public vehicles). For the New York and Washington areas, fleet vehicles assumed to be replaced annually are estimated as proportional fractions of the total vehicle populations of those cities.

b. Blend Scenarios

The final scenario evaluated by EPA for this paper estimates the effect of mandating oxygenated blend usage in two CO non-attainment areas, one at low altitude (Phoenix) and one at high altitude (Denver). Unlike the methanol vehicle scenarios, in which only a segment of the vehicle population is assumed to use an alternative fuel, in this blend scenario all gasoline vehicles in an area would switch to blend fuels. This distinction means that the per-vehicle CO reductions in Table 5 can be converted into estimates of fleetwide effect much more directly than could the methanol vehicle reduction numbers. The single analytical step required is to estimate the fleet mix of emission control technologies that would exist in a given year (non-catalyst, open-loop, and closed loop) and weight the effect of each type of vehicle if it were to switch to oxygenated blends.

Another way in which the blends scenario differs from the methanol vehicle scenarios is that while new fuels could be introduced sooner than could new vehicles, the largest CO reductions would only be available for the relatively near future. As suggested in the discussion of CO attainment above, most of the reduction in CO would come from older vehicles, which are steadily being replaced by much cleaner closed-loop vehicles. For this reason, the blends scenario focuses on CO reductions in the near term -- 1990 and 1995 (as well as for a future date when the entire fleet has turned over to closed-loop vehicles -- probably between 2000 and 2010).

2. Methanol Vehicles Replacing Gasoline Vehicles-(1991-1995)

In this scenario, significant numbers of current technology methanol light-duty vehicles and light-duty trucks (and/or flexible-fueled vehicles operating on M85) would replace gasoline fleet vehicles as well as some vehicles in general use in the New York, Washington, DC, and Los Angeles metropolitan areas. EPA's analysis of the effect on ozone levels of such a replacement of gasoline vehicles with methanol vehicles is calculated from composite projections for future years of gasoline vehicle emission performance and of expected annual vehicle miles traveled (VMT). Fleet vehicles are

assumed to average 33,000 miles per year (taken from DOT figures that are on the high end of available estimates), while vehicles in general use are assumed to average 13,000 miles per year (from EPA's MOBILE3 data base for light-duty vehicles and light-duty trucks in their early years of use).

An estimate of the VOC effect of using methanol vehicles to replace gasoline vehicles can be derived for each city from the per-vehicle emission factors in Table 7, and the city-specific reactivity factors from Table 6 (New York's value was approximated by averaging the values for Boston, Philadelphia, Baltimore, Scranton, and Allentown). The VOC effects for each city can be applied to city-specific VOC inventory projections for a specified year--1995 in this case. In this way different motor vehicle VOC fractions are incorporated for each city. For the New York and Washington area estimates, fleetwide composite emission factors and projected VOC inventories developed by EPA were used; for the Los Angeles area estimate, EPA used California Air Resources Board emission factors and inventories. The results presented in Table 9 are percent reductions in total mobile-plus-stationary source VOC emissions on an ozone-equivalent basis. Because of the ozone equivalence, these estimated reductions can also be interpreted as percent reductions in the ambient ozone levels expected in each city. For comparison, Table 9 presents results for the case in which the only vehicles that were replaced with methanol vehicles were those sold to centralized fleets of 10 or more vehicles, as well as for a broader case including a number of general use vehicles.

### 3. Methanol Vehicles Replacing Gasoline Vehicles (1995-2000)

The second scenario is similar to the first but is designed to evaluate the effect of advanced technology methanol vehicles. The same annual replacement rates for fleet vehicles used for the first scenario are used here as well. Table 8 summarizes the numbers of fleet and general use vehicles in use by the year 2000. Again, every new vehicle in a fleet of 10 or more vehicles is assumed to be a methanol vehicle; additional methanol vehicles in general use are also included in the analysis.

Table 9 presents the percent reduction in ozone-equivalent VOC emissions--and hence in ambient ozone--for each city in the year 2000. As for the first scenario, results are presented both for a case of fleet-vehicle only replacement as well as the case in which general-use methanol vehicles are also included.

4. Methanol Vehicles Replacing Gasoline Vehicles - Steady State

The foregoing discussions have considered the effect that methanol vehicles might have on ozone levels after a few years of implementation. For two reasons, however, this sort of analysis gives an incomplete understanding of the potential impact of a methanol program.

The first is related to the fact that methanol vehicles are assumed to displace newer, cleaner gasoline vehicles, which are responsible for less than a proportional share of the VOC inventory. This is because their emission are less deteriorated with regard to their emissions standards and because they may have been certified to tighter emission standards than the "average" gasoline vehicle was. Thus after only five years of fleet integration, a significant portion of the fleet is still composed of older gasoline vehicles which account for a greater fraction of total emissions than their VMT percentage would suggest. As a result, methanol's impact on total fleet emissions is significantly less than it is on the emissions of each model year's fleet into which methanol vehicles have penetrated. By extension, it is also significantly less than it would be on the emissions of an entire fleet in which methanol vehicles are given an equivalent age distribution to gasoline vehicles. Carrying forward an implementation scenario until the entire gasoline fleet is able to turn over will provide a demonstration of the maximum potential impact of methanol on mobile source emissions.

The second factor that comes into play in a longer term implementation is related to the changing mobile and stationary source fractions of the VOC inventory. Since mobile source fractions are generally expected to decline over time (see Table 10) the potential impact of a methanol fleet on overall ambient ozone levels is decreased. This second factor will directionally work to offset the impact of the first.

In the steady state scenarios, therefore, it is assumed that sufficient time has elapsed to allow methanol vehicles to integrate with the fleet as a fraction of the sales of every model year that is present in the total fleet. According to National Highway Traffic Safety Administration scrap rate data, 98 percent of the gasoline vehicle fleet turns over in 20 years, 83 percent in 15. To calculate steady state impacts of the two methanol scenarios considered, therefore, one would ideally take the current technology program out 20 years to 2011 and the advanced technology program to 2015. (Admittedly, private fleets may turn over more quickly than the overall fleet, but the scenarios considered all involve a substantial portion of private vehicles as well.) Since the mobile source fractions shown in Table 10 remain fairly stable

after 2010, and since later data are unavailable, year 2010 data for New York and Washington and year 2000 data for Los Angeles are used to represent later years.

For these scenarios it is assumed that methanol vehicles are responsible for 30 percent of all VMT. This assumption implies that methanol vehicles constitute 30 percent of the total urban fleet and, to reiterate the above discussion, are distributed through it in an age and use distribution identical to that for gasoline vehicles.

Table 11 shows the results of this analysis. Phoenix is included in this table along with the other cities which have been used throughout this analysis because it has a high mobile source VOC fraction into the early 21st century and represents an upper limit on the potential impact of methanol implementation. The effect of methanol on mobile source VOC is distinguished in this table from its effect on total urban VOC to provide an illustration of the relevance of the mobile/stationary breakdown to this analysis. In this table, current technology methanol vehicles are seen to provide a potential 3 to 4 percent impact on urban VOC and advanced technology could reduce urban VOC by 6 to 11 percent.

5. Oxygenated Blends Replacing Gasoline in All Vehicles

The final scenario evaluated in this paper projects the effect of a mandated oxygenated blends program on ambient CO levels. EPA's analysis is based on an assumption that fuels would contain about 3.7 percent oxygen. This approximate level of oxygen content can be achieved with gasohol (ten percent ethanol), Oxinol (4.75 percent methanol and 4.75 percent t-butyl alcohol), or the DuPont methanol blend (five percent methanol with 2.5 percent cosolvent alcohol). A blend of MTBE and gasoline would need to be 20 percent by volume to reach 3.7 percent oxygen; currently MTBE can only be used at levels up to 11 percent by volume (two percent oxygen). CO reduction estimates for a program mandating two percent oxygen fuel are assumed to be proportional to the 3.7 percent oxygen values.

With respect to a mandated fuel substitution program, the per-vehicle CO reductions in Table 5 essentially represent fleetwide CO reductions as well because all gasoline vehicles would in effect become blend-fueled vehicles. To convert this mobile source CO reduction to a total CO reduction, an assumption about the mobile source fraction of CO emissions is needed. We have used values of 90 percent for Phoenix and 75 percent for Denver (the Denver fraction is lower due to the extensive use of wood stoves there).

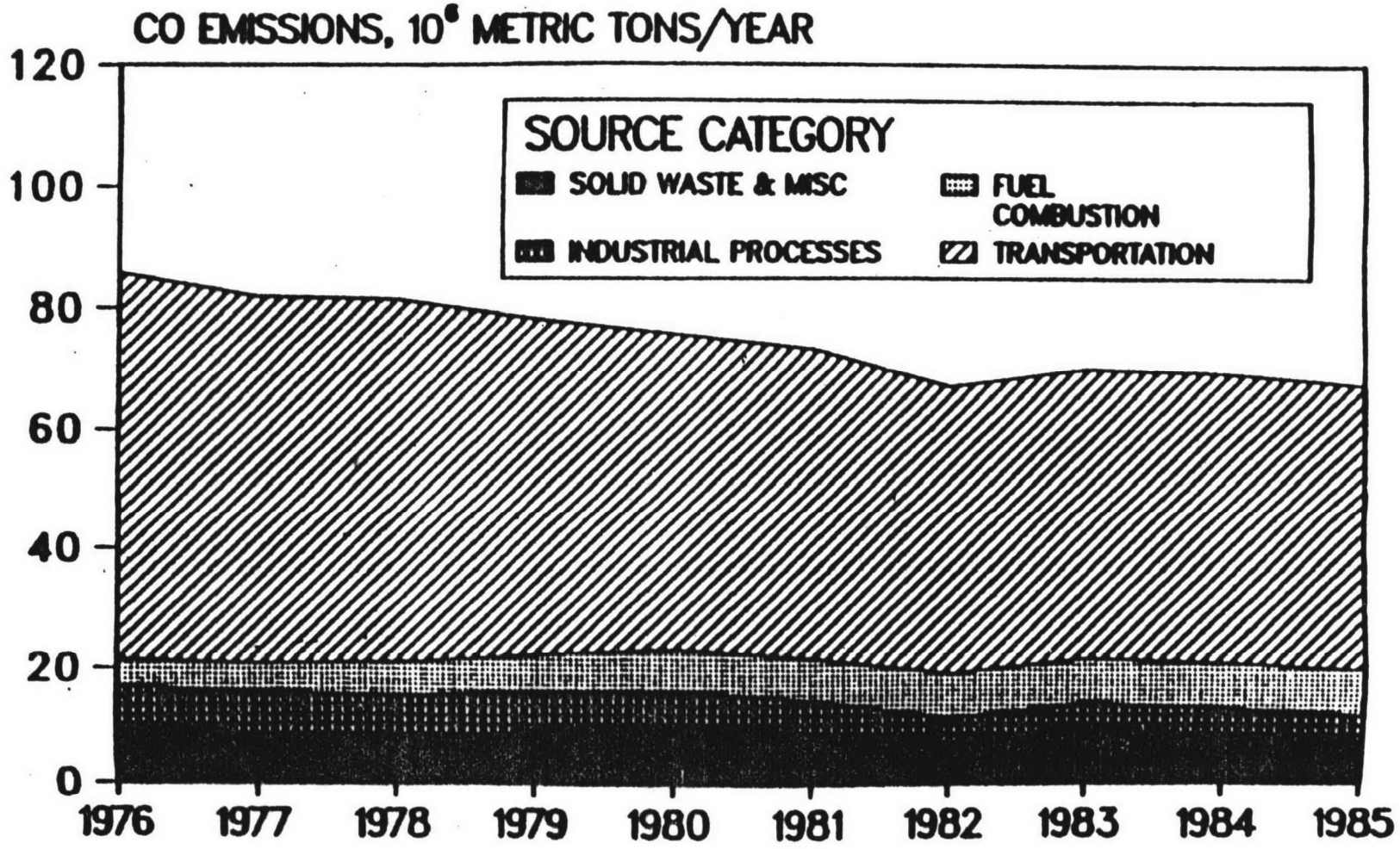
Table 9 presents both the percent reduction in mobile source CO that EPA would expect in 1990, 1995, and in a steady-state (100 percent closed-loop) year in an area

implementing an oxygenated fuels program. Specific total stationary-plus mobile-source CO reductions are presented for Denver and Phoenix.

Finally, EPA has calculated rough estimates of the CO effect of requiring a retrofit conversion of all vehicles in an area to operate on CNG. The estimates are based on the low end of the range of potential CO reductions presented in Table 5. Overall ambient CO reductions from a total fleetwide CNG retrofit program appear in Table 9.

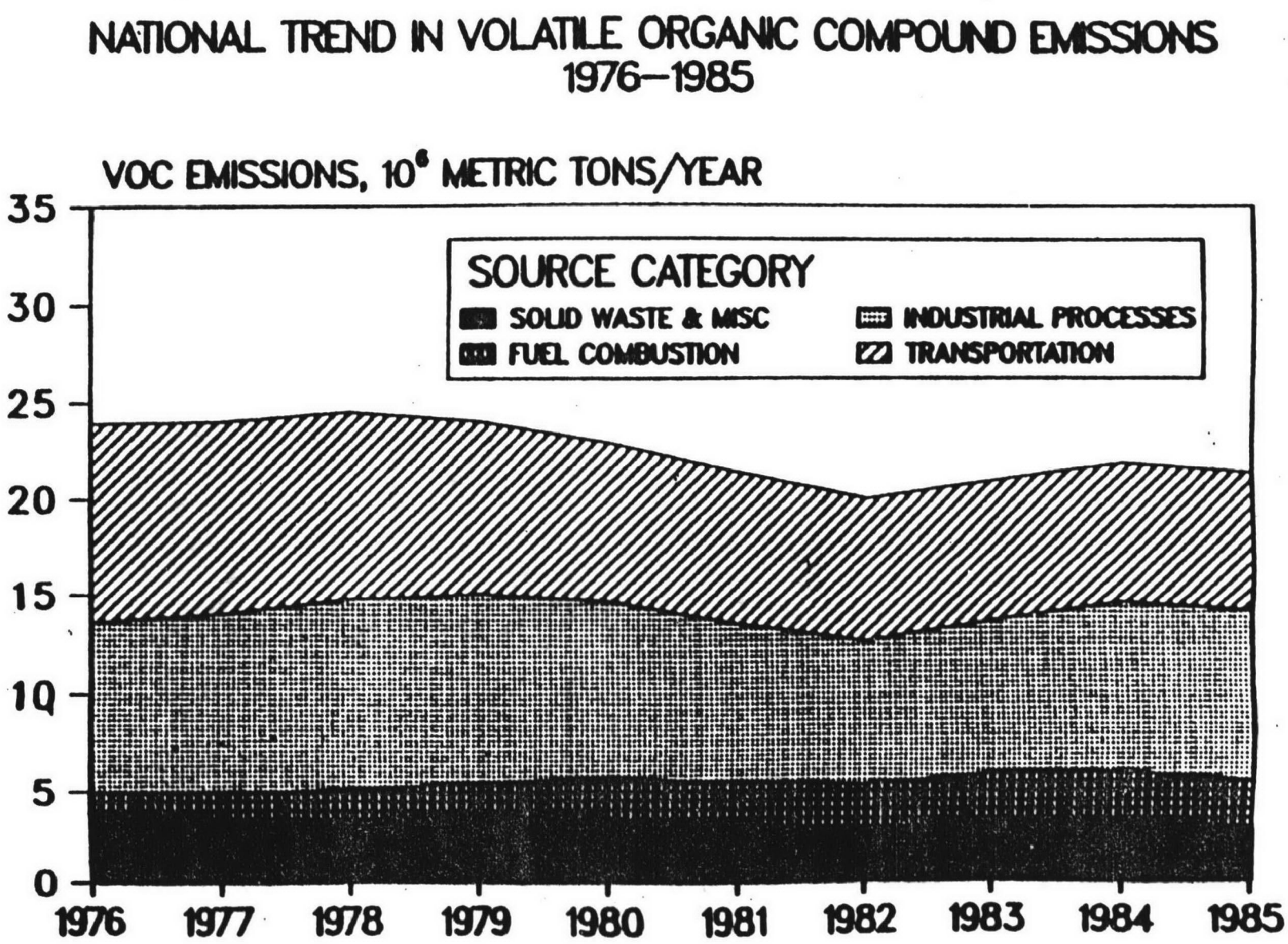
Figure 1

# NATIONAL TREND IN CARBON MONOXIDE EMISSIONS 1976–1985



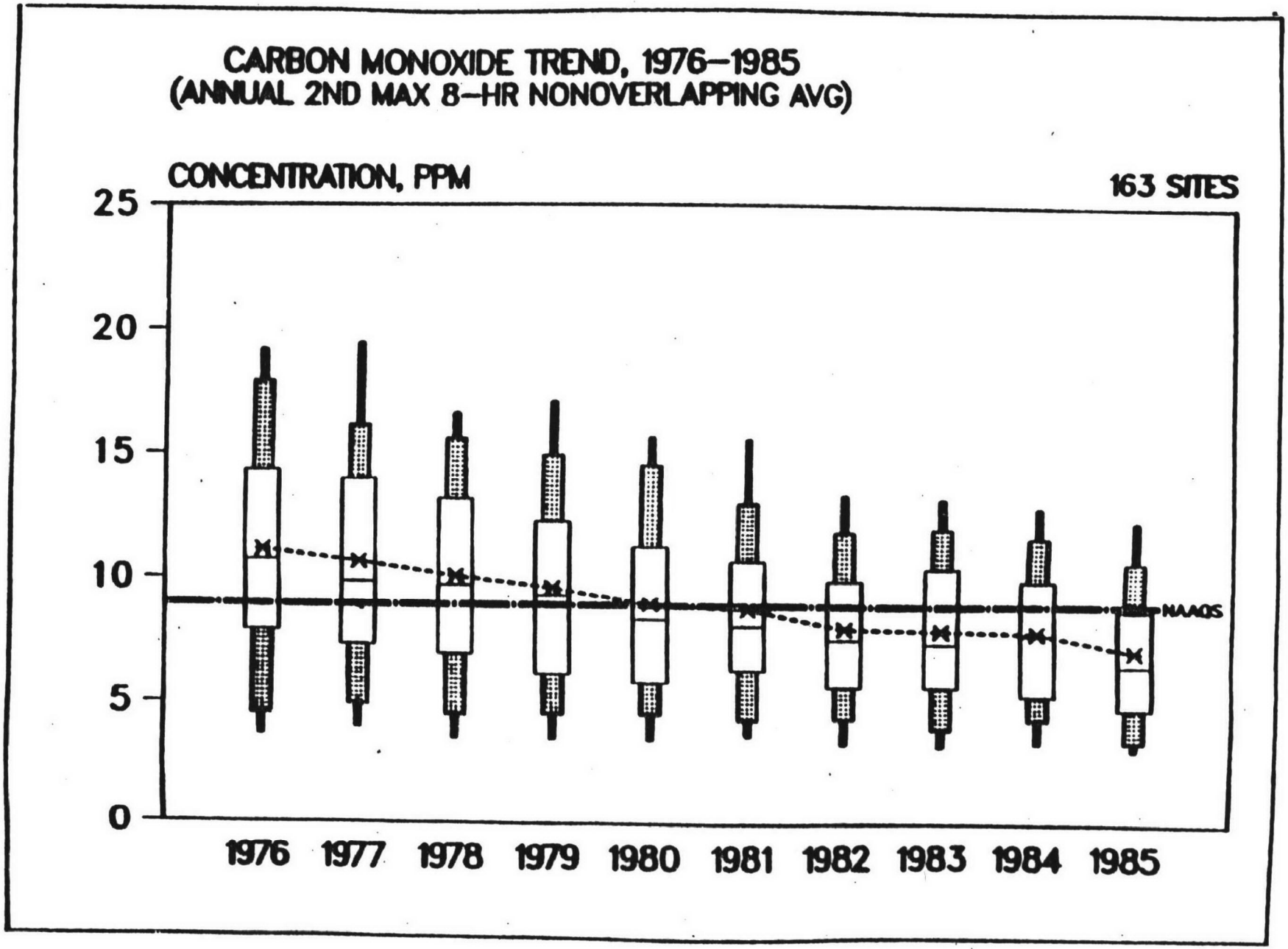

Source: EPA 1985 Trends Document

Figure 2

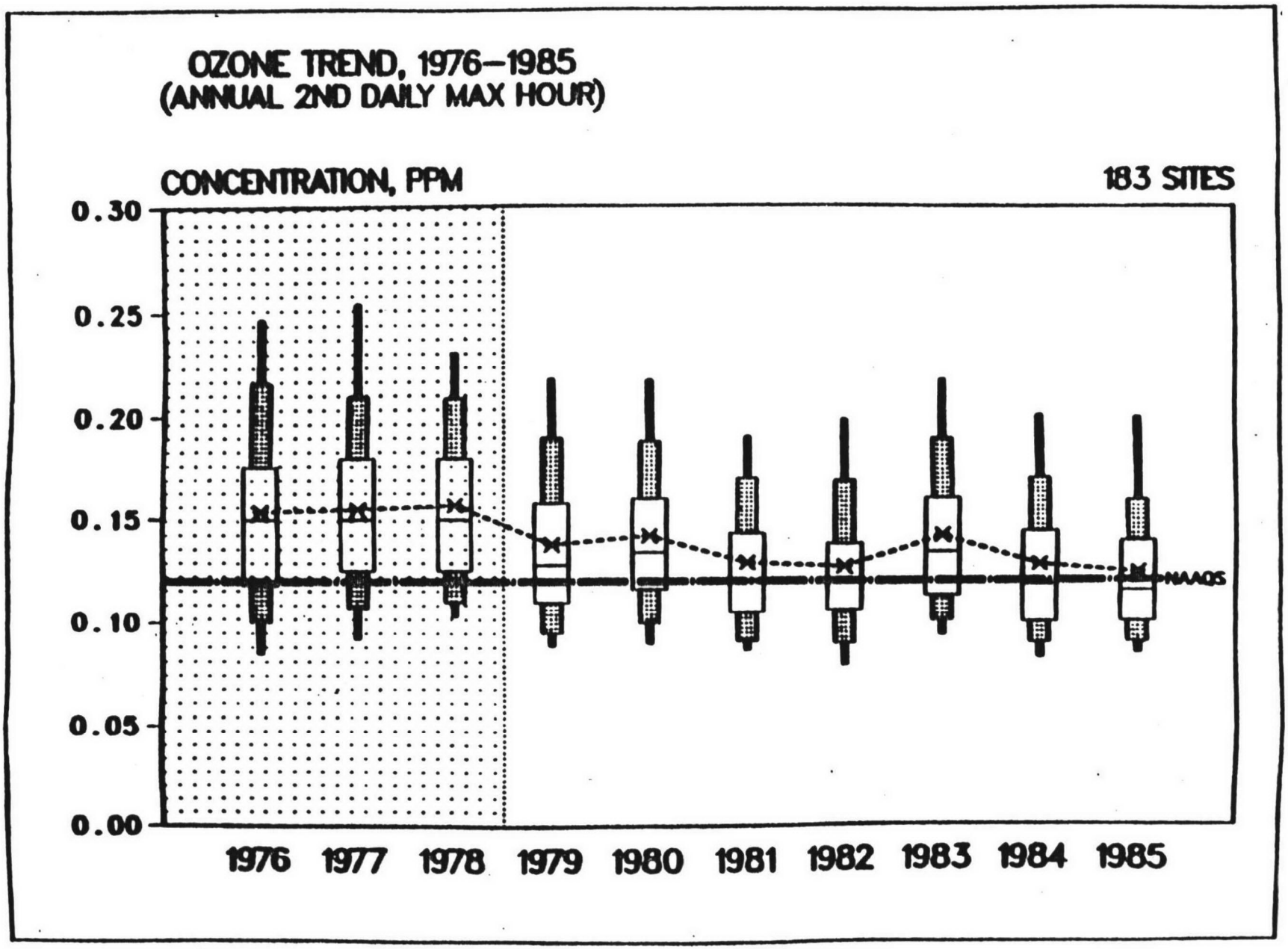



Source: EPA 1985 Trends Document

Figure 3




Source: EPA 1985 Trends Document




Source: EPA 1985 Trends Document

Table 1

Mobile Source VOC Fractions
for Selected Non-Attainment Areas (Percent)

| City | 1983 | 2000 |
|---|---|---|
| Houston | 35 | 16 |
| Chicago | 43 | 25 |
| New York City | 49 | 28 |
| Milwaukee | 51 | 29 |
| Washington, DC | 55 | 32 |
| Phoenix | 64 | 40 |

Table 2

Top 30 Ozone Non-Attainment Areas

| Area Name | 1982-84 Design Value (ppm) | Percentage Reduction Required |
|---|---|---|
| Beaumont-Port Arthur, TX | .21 | 87 |
| Los Angeles, CA | .36 | 72 |
| Houston, TX | .25 | 70 |
| Greater Connecticut | .23 | 64 |
| Oxnard-Venturi, CA | .21 | 60 |
| Chicago Metro. | .20 | 60 |
| Baton Rouge, LA | .17 | 59 |
| San Diego, CA | .20 | 59 |
| Atlantic City, NJ | .19 | 57 |
| Boston Metro., MA | .19 | 57 |
| El Paso, TX | .17 | 56 |
| Sacramento, CA | .18 | 54 |
| Baltimore, MD | .17 | 51 |
| Galveston, TX | .17 | 51 |
| Dallas-Ft.Worth, TX | .16 | 51 |
| San Francisco, CA | .17 | 50 |
| Atlanta, GA | .17 | 48 |
| Bakersfield, CA | .16 | 46 |
| Fresno, CA | .16 | 46 |
| Birmingham, AL | .15 | 38 |
| Allentown-Bethlehem, PA | .15 | 38 |
| Modesto, CA | .15 | 38 |
| Cincinnati, OH | .15 | 32 |
| Denver-Boulder, CO | .14 | 29 |
| Brazoria, TX | .14 | 27 |
| Detroit, MI | .14 | 27 |
| Huntington, WV | .14 | 27 |
| Vallejo-Fairfield, CA | .14 | 27 |
| Santa Barbara, CA | .14 | 25 |
| Kansas City, MO | .14 | 23 |

Table 3

## Top 30 CO Non-Attainment Areas

| Area Name | 1983-85 Design Value (ppm) | Percentage Reduction Required |
|---|---|---|
| Los Angeles-Long Beach | 27.4 | 67 |
| Denver | 24.0 | 63 |
| Phoenix | 20.3 | 56 |
| Provo-Orem | 19.1 | 53 |
| Anchorage | 18.0 | 50 |
| Ft. Collins, CO | 17.8 | 49 |
| Fairbanks | 17.7 | 49 |
| Newark | 17.7 | 49 |
| Albuquerque | 17.2 | 48 |
| Raleigh-Durham | 16.6 | 46 |
| Medford, OR | 16.3 | 45 |
| Sacramento | 16.3 | 45 |
| Las Vegas | 16.3 | 45 |
| Reno | 16.2 | 44 |
| Greeley, CO | 16.2 | 44 |
| Nashua, NH | 16.0 | 44 |
| New York City | 16.0 | 44 |
| Boise City, ID | 15.5 | 42 |
| Spokane, WA | 15.4 | 42 |
| Syracuse, NY | 14.7 | 39 |
| San Jose, CA | 14.3 | 37 |
| Boston | 14.1 | 36 |
| Baltimore | 13.9 | 35 |
| Yakima, WA | 13.9 | 35 |
| Washington, DC | 13.8 | 35 |
| Jersey City, NJ | 13.7 | 34 |
| Chicago | 13.3 | 32 |
| El Paso | 13.3 | 32 |
| Charlotte, NC | 13.2 | 32 |
| Colorado Springs | 13.2 | 32 |

Table 4

## Nationwide Non-California Attainment Projections

| | 1983 | 1990 | 1995 | 2000 | 2010 |
|---|---|---|---|---|---|
| **Ozone Non-Attainment** | | | | | |
| Non-California Areas | 61[1] | 40 | 38 | 41 | 51 |
| California Areas | 11 | -- | -- | -- | -- |
| **CO Non-Attainment** | | | | | |
| Non-California Areas[2] | 71[3] | 17-23 | 7-15 | 6-14 | -- |
| California Areas | 10 | -- | -- | -- | -- |

[1] Projections based on 1982-84 data; recent 1983-85 data show 65 non-California areas and 11 California areas

[2] Ranges show sensitivity to assumptions about temperature and driving patterns

[3] Unchanged for 1983-85

Table 5

Emission Reduction Potential of Light-Duty Vehicles
(Per-vehicle basis compared to gasoline vehicles meeting current standards)

| | VOC | CO | NOx |
|---|---|---|---|
| **Methanol** | | | |
| FFV (M85) | -(20 to 50)% | 0 | 0 |
| Current Technology | -(20 to 50)% | 0 | 0 |
| Advanced Technology* | -(85 to 95)% | -(30 to 90)% | 0 |
| **CNG** | | | |
| FFV/Retrofit* | -(50 to 80)% | -(50 to 90)% | -20% to +80% |
| Advanced Technology* | -(50 to 90)% | -(50 to 90)% | -20% to +80% |

OXYGENATED BLENDS

| VOC | Ethanol | Methanol | MTBE |
|---|---|---|---|
| Constant RVP | -2% to +5% | -5% to +5% | -1% |
| 1 Psi Higher | +15% to +35% | +9% to +30% | Not Applicable** |

| CO | Ethanol, Methanol (3.7% Oxygen) | MTBE (2.0% Oxygen) |
|---|---|---|
| Non-Catalyst | -18% | -10% |
| Open Loop | -30% | -16% |
| Closed Loop | -10% | -5% |
| **NOx** | | |
| Open Loop | +5% | +3% |
| Closed Loop | +6% | +3% |

* Projections based on very small data bases; CNG vehicles would also typically experience a loss in power and performance.

** Unlike ethanol and methanol, MTBE does not appear to significantly increase the RVP of gasoline.

Table 6

Reactivities[1]
of Methanol and Formaldehyde

| Study | City | Methanol Reactivity | Formaldehyde Reactivity |
|---|---|---|---|
| SAI | Los Angeles[2] | 0.02 | 2.95 |
| SAI | Philadelphia | 0.413 | 5.873 |
| Ford | Allentown | 0.350 | 4.600 |
| | Atlanta | 0.375 | 4.125 |
| | Baltimore | 0.533 | 3.867 |
| | Boston | 0.232 | 5.180 |
| | Chicago | 0.310 | 5.138 |
| | Cincinnati | 0.600 | 4.600 |
| | Dallas | 0.563 | 4.625 |
| | Detroit | 0.372 | 4.490 |
| | El Paso | 0.563 | 4.938 |
| | Ft. Worth | 0.417 | 5.972 |
| | Houston | 0.625 | 4.375 |
| | Milwaukee | 0.500 | 3.833 |
| | Nashville | 0.520 | 4.120 |
| | Philadelphia | 0.441 | 4.588 |
| | Phoenix | 0.421 | 5.158 |
| | Pittsburg | 0.258 | 5.560 |
| | Scranton | 0.375 | 4.875 |
| | St. Louis | 0.215 | 5.754 |
| | Washington D.C. | 0.444 | 4.611 |
| | Youngstown | 0.500 | 5.500 |
| | Average [3] | 0.430 | 4.826 |

[1] Amount of ozone produced relative to gasoline hydrocarbon emissions.

[2] Methanol reactivity appears low compared with other data. Further modeling required to verify.

[3] Average of 20 non-California cities (uses average values from SAI and Ford for Philadelphia).

Table 7

In-Use Emission Factors and Relative Ozone Potential
for Gasoline and Methanol Vehicles[1]

| | Exhaust (g/mi) | | | Evap (g/mi)[6] | | Refueling (g/mi) | |
|---|---|---|---|---|---|---|---|
| | HC | MeOH | Form | HC | MeOH | HC | MeOH |
| **49-State** | | | | | | | |
| Gasoline[2] | 1.00 | - | - | 0.51 | - | 0.21 | - |
| Current Methanol (M85)[3] | 0.09 | 1.08 | 0.09 | 0.14 | 0.39 | 0.14 | 0.04 |
| Advanced Methanol (M100)[3] | 0.04 | 0.44 | 0.01 | - | 0.20 | - | 0.05 |
| **California** | | | | | | | |
| Gasoline[4] | 0.58 | - | - | 0.07 | - | 0.03 | - |
| Current Methanol (M85)[5] | 0.25 | 0.70 | 0.06 | 0.04 | 0.10 | 0.02 | 0.01 |
| Advanced Methanol (M100)[3] | 0.04 | 0.44 | 0.01 | - | 0.20 | - | 0.01 |

[1] Based on 50,000 mile performance. Assumes carbon based standards apply to current technology methanol vehicles.

[2] Based on 1990+ model year vehicle EPA data. No volatility or refueling controls in place.

[3] Based on testing of prototype venicles. Assumes in-use versus standards offsets of 2.2 and 2.3 for exhaust and evaporative emissions. Offsets estimated based on performance of current technology gasoline vehicles. Refueling data based on test of only one methanol vehicle.

[4] CARB projections for future exhaust, evap emission factors. EPA estimate of refueling emissions assumes 85 percent control efficiency of LA Stage II vapor recovery systems.

[5] Based on testing of prototype venicles. Assumes in-use versus sstandards offsets of 1.42 and .58 for exhaust and evaporative emissions offsets estimated ased on performance of current technology California gasoline vehicles.

[6] Converted to g/mi by assuming 3.05 trips and 31.1 miles per day.

Table 8

## Methanol Vehicle Distributions Used in Analysis (Vehicles/Year)

| | 1991 | 1995 (1991-1995) | 2000 (1995-2000) |
|---|---|---|---|
| Total Vehicles Replaced Per Year | | | |
| New York | 200,000 | 1,000,000 | 1,200,000 |
| Washington | 25,000 | 125,000 | 150,000 |
| Los Angeles | 275,000 | 1,375,000 | 1,650,000 |

## Distribution With Maximum Fleet Vehicle Replacement (Vehicles/Year)*

| | | | |
|---|---|---|---|
| New York | | | |
| Fleet Vehicles | 130,500 | 652,500 | 783,000 |
| General Use | 69,500 | 347,500 | 417,000 |
| Washington | | | |
| Fleet Vehicles | 24,000 | 120,000 | 144,000 |
| General Use | 1,000 | 5,000 | 6,000 |
| Los Angeles | | | |
| Fleet Vehicles | 80,000 | 400,000 | 480,000 |
| General Use | 195,000 | 975,000 | 1,170,000 |

* Based on estimated maximum Los Angeles area fleet vehicle replacement of 80,000 vehicles per year (for fleets of 10 or more vehicles). New York and Washington fleet replacement estimates are proportional to the Los Angeles ratio of 80,000 fleet vehicles to 7,300,000 total vehicles.

Table 9

## Estimated Percent VOC Reduction for Methanol Strategies

| | Fleets Only | | Fleets Plus Some General Use Vehicles | |
|---|---|---|---|---|
| | %VOC Reduction | %VMT | %VOC Reduction | %VMT |
| **Current (1995)** | | | | |
| New York City | 1.1 | 15 | 1.4 | 18 |
| Washington | 1.2 | 14 | 1.2 | 15 |
| Los Angeles | 0.9 | 14 | 1.7 | 26 |
| **Advanced (2000)** | | | | |
| New York City | 3.5 | 16 | 4.2 | 20 |
| Washington | 3.8 | 16 | 3.8 | 16 |
| Los Angeles | 2.3 | 15 | 4.5 | 29 |

## Estimated Fleetwide Percent CO Reductions for Blends Strategies

| | 2% Oxygen | | | 3.7% Oxygen | | | CNG Retrofit |
|---|---|---|---|---|---|---|---|
| | 1990 | 1995 | Steady State | 1990 | 1995 | Steady State | |
| Mobile Source CO | 12 | 10 | 5 | 22 | 18 | 10 | 50 |
| Total CO (Denver) | 9 | 8 | 4 | 17 | 14 | 8 | 38 |
| Total CO (Phoenix) | 11 | 9 | 5 | 20 | 16 | 9 | 45 |

Table 10

## Mobile Source Percentages of Total VOC Inventories

| City* | Year | | | |
|---|---|---|---|---|
| | 1983 | 1995 | 2000 | 2010 |
| New York | 49 | 30 | 28 | 28 |
| Washington | 55 | 34 | 32 | 33 |
| Los Angeles | 50 | 27 | 24 | NA** |

* Data for New York and Washington from EPA. Data for Los Angeles based on information provided by California Air Resources Board.

** NA indicates data not available.