IV. PAPERS SUBMITTED BY AUTHORS
(In Alphabetical Order by First Author)

DEVELOPMENT OF CHRONIC PRODUCTIVE COUGH AS ASSOCIATED WITH LONG-TERM AMBIENT INHALABLE PARTICULATE POLLUTANTS (PM10) IN NON-SMOKING ADULTS -- The AHSMOG Study

David E. Abbey[A], Naomi Nishino[A] and William F. McDonnell[B]

[A] Loma Linda University, Loma Linda CA

[B] EPA, University of N. Carolina, Chapel Hill, NC

ABSTRACT

Standardized respiratory symptoms questionnaires were completed by 3,091 nonsmoking California Seventh-day Adventists in 1977, 1987 and 1992. Ambient concentrations of air pollutants estimated included total suspended particulates (TSP), 1973-1987; particulates <10 microns in diameter (PM10), indirectly estimated from TSP, 1973-1987, and directly estimated from monitored PM10, 1987-1992; suspended sulfates (SO_4) 1977-1992; sulfur dioxide (SO_2) 1973-1992; 8-hour average of ozone (O_3) 1973-1992. Adjustments to ambient concentrations have been made for time spent indoors. Chronic bronchitis was defined as cough (cough type) or with sputum production (chronic productive cough) on most days, for at least three months/year, for two-years or more. Using indirect estimates of PM10 prior to 1987 and direct estimates after 1987, multiple logistic regression models adjusting for covariates have been used to study development 1977-1992 of chronic bronchitis (228 new cases) and chronic productive cough (191 new cases). For chronic productive cough, gender specific analyses indicated similar effects of PM10 and covariates for both genders. Hence, a pooled analyses was conducted using gender as a covariate. The relative risk for developing chronic productive cough was 1.57 (95% confidence interval: 1.08, 2.27) for an increase of 83 days/year in excess of 100 $\mu g/m^3$ of PM10, while no association was found with O_3, SO_4, or SO_2. Statistically significant associations were seen for development of chronic productive cough for days when PM10 exceeded 80 $\mu g/m^3$, a level far below the current U.S. standard. The estimated relative risk for developing definite symptoms of overall chronic bronchitis (either cough or sputum type) in 1992 as associated with an increase of 83-days/year when PM10 concentrations exceeded 100 $\mu g/m^3$ was 1.71 (95% confidence interval: 1.27, 10.53) ($p = 0.0007$).

KEY WORDS: (3-10 words, or short phrases) air pollution, chronic productive cough, inhalable particulates

INTRODUCTION

There have been few human population studies of the health effects of long-term ambient concentrations of inhalable particulates PM10, indeed, PM10 has been monitored on a statewide basis throughout California only since 1987. A cohort of 3,091 non-smoking California Seventh-day Adventists that has been followed since 1977 provides an ideal opportunity to study such health effects in the absence of confounding from active cigarette smoking. A previous paper reported on analyses conducted on this cohort which dealt with development of respiratory symptoms between 1977 and 1987 as associated with indirectly estimated ambient concentrations of PM10.[1] The indirect estimates of PM10 were based on site and seasonal specific regressions on TSP. This paper utilizes directly monitored PM10 data since 1987 along with the indirect estimates of PM10, available for the cohort, prior to 1987 to study associations of long-term ambient concentrations of PM10, 1973 to 1992 as associated with development of chronic productive cough and overall chronic bronchitis. Multipollutant analyses that replace PM10 with other pollutants, or add other pollutants one-at-a-time to health effects models are also conducted for suspended sulfates (SO_4), sulfur dioxide (SO_2) and ozone (O_3). The study uses different cutoffs for daily concentrations of PM10 to address the question "at what levels of ambient PM10 do statistically significant health effects begin to occur?"

METHODS

Enrollment and Follow-up of Cohort. Abbey, et al[2] details the enrollment and follow-up of the cohort and gives descriptive characteristics. Briefly, the cohort consisted of 3,091 baptized Seventh-day Adventist, non-Hispanic whites who completed a detailed lifestyle questionnaire in 1976 as part of the Adventist Health Study (AHS).[3] This questionnaire ascertained demographic, dietary, and lifestyle factors thought to be related to cancer and heart disease incidence. In addition, individuals completed standardized respiratory symptoms questionnaires in April: 1977, 1987 and 1992. This respiratory

questionnaire was originally developed by the British Medical Research Council and modified in 1971 for use in the U.S. by the National Heart and Lung Institute (NHLI) later renamed the National Heart, Lung and Blood Institute (NHLBI). The 1987 and 1992 questionnaires also contained additional questions from the standardized questionnaire of the American Thoracic Society (ATS) which was developed from the NHLBI questionnaire. In addition, to self-reported symptoms of chronic respiratory disease, the questionnaires ascertained occupational exposures, hours of freeway driving, time spent indoors, detailed smoking histories, and environmental tobacco smoke (ETS) exposures. Residence and work location history since 1966 was ascertained by month and zip code. The response fraction obtained at each of the mailings of the questionnaires in 1977, 1987 and 1992 was the same -- 87%. To be included in the prospective cohort, individuals must have been -- 25 or older in 1976, have lived at least the previous ten years within five miles of the present residence in 1977, and reside in one of three metropolitan areas - San Francisco, the South Coast Air Basin, (i.e., Los Angeles and eastward), or San Diego -- or be included in a random sample of 862 individuals residing in the rest of California.

In 1977, 60% of the cohort lived in the South Coast Air Basin. Ages in 1977 ranged from 27 to 87 with an average age of 54. No individuals who reported smoking on the 1977, 1987, or 1992 questionnaires were included in the study. Although abstinence from smoking is a prescription for church membership in the SDA church, some individuals had joined the SDA church later in life and had smoked prior to 1977. Descriptive characteristics of the cohort are given in Table I. These descriptive statistics are given gender specific where large gender differences occurred.

Definition of Health Outcomes. Computer algorithms developed by Hodgkin, et al.[4] were used to classify individuals in 1977, 1987, and again in 1992 as having overall chronic bronchitis or chronic productive cough. Individuals were classified as having "definite" chronic bronchitis symptoms if they

had cough (cough type) and/or sputum production (chronic productive cough) on most days for at least three months per year, for two years or more. Individuals having some symptoms but not meeting the criteria for definite were classified into a "possible" category. The primary analyses in this paper focuses on chronic productive cough as this was regarded as the more reliable and serious of the types of bronchitis. The primary outcome for analyses was taken as development of "definite" symptoms (191 new cases) between 1977 and 1992 in individuals who did not have definite symptoms in 1977. We also looked at development of overall chronic bronchitis (either cough or sputum type) (228 new cases). Table II shows gender specific prevalence, development and reversal rates of chronic productive cough and overall chronic bronchitis.

Methods for Estimating Concentrations of PM10. Simultaneously monitored PM10 and TSP occurring at different locations every sixth day throughout California between the years 1979 through 1986 were used to form site-seasonal regressions of PM10 on TSP for 1973-1987.[1] Indirect estimates of PM10 through 1987 and directly monitored PM10 since 1987 were interpolated by month to zip codes of home and work locations of study participants using methods described in Abbey et al..[5] No interpolations were made in excess of 31.25 miles (50 km). For 1973-1987 if any stations within 20 miles (32 km) of a zip code centroid had data, only such stations were used for interpolations. This distance was reduced to 10 miles (16 km) for 1987-1992. No interpolations were made across geographic obstructions in excess of 250 meters or across meteorological airflow barriers.

For those study participants who worked more than five miles from home, a weighted average of the monthly home location and work location statistics were computed from the number of working and nonworking days in each month, assuming that study participants were at their home locations except 9/24 (9:00 AM to 5:00 PM) of a 24-hour period on working days. In addition to monthly mean concentration of PM10, monthly exceedance frequency statistics for cutoffs of 40, 50, 60, 80, and 100

$\mu g/m^3$ were computed. Exceedance frequency was defined as number of estimated days in the month when the 24-hour PM10 mean concentration exceeded a cutoff. The precision of the interpolation methods was assessed by regarding fixed site stations with monitored PM10 as receptor sites and interpolating PM10 concentrations monitored at surrounding stations. Monthly values were first averaged over a two year period. High correlations were seen with a correlation with actual PM10 of 0.88 being seen for directly interpolated PM10 and a correlation of 0.86 for indirectly estimated PM10.[1]

Statistical Models. Multiple logistic regression models, fit using maximum likelihood, were used to study associations between development of new cases of disease between 1977 and 1992 and average annual days per year 1973-1992 when PM10 concentrations exceeded 100 $\mu g/m^3$ [PM10 (100)] adjusting for covariates. A "main effects" model was first formed using the covariates -- age; gender; years education; whether or not individual had asthma, bronchitis, or pneumonia before age 16; frequency of childhood colds on a 5-point Likert Scale; maximum of years dusts or fumes at work; years lived with a smoker; years worked with a smoker; and pack-years of past cigarette smoking prior to 1977. These main effect covariates were chosen from a review of the literature as those most likely to be related to development of chronic productive cough. Gender specific main effects models were also considered but not used as the effects of most covariates on development of chronic productive cough were similar and the interaction between gender and PM10 was not significant. Secondary candidate covariates were also included in the model if they made a substantial change in the PM10 (100) coefficient (>0.0001) or made a statistically significant improvement in the model according to the likelihood ratio test ($p < 0.05$). All p values given subsequently are from the likelihood ratio test unless otherwise stated. The secondary candidate covariates included -- years spent in hobbies that produced dusts or fumes; whether or not lived near a dusty road in 1992; person to bedroom ratio for

home in 1992; present type of home heating and years of two types of home heating as a child (up to age 18) and as an adult according to the two types -- 1) wood, coal, or oil, or 2) gas, kerosene. The likelihood ratio test was used to determine if there were interactions between PM10 (100) and hours spent outdoors in 1977 or whether or not home was air conditioned in 1977; also between PM10 main effects or retained secondary candidate covariates. In order to minimize the problem of multiple testing similar covariates were grouped together and a smaller number of chunks of covariates tested for interaction. No significant ($p < 0.05$) interactions were found. After testing for interactions a step-down procedure was used to remove nonsignificant main effect covariates that, when removed, did not change the PM10 coefficient by more than 0.0001. However, gender, age, and education were retained so that adjustment was made form these factors.

The Hosmer-Lemeshow Goodness of Fit test[6] indicated adequate fit of the multiple logistic regression models. Logistic linearity of continuous variables used in the model was checked using two dummy variables to represent three categories of the continuous variable as outlined by Hosmer & Lemeshow.[7] There was no evidence of departure from logistic linearity for the continuous variables used in final models. Separate multiple regression models were fitted for mean concentration, as well as exceedance frequencies for each of the cutoffs. Formulas given by Abbey et al,[8] were used to estimate increased relative risk (RR) for a k unit incremental increase in a continuous valued variable.

RESULTS

The multiple logistic regression model for development of chronic productive cough see Table III, showed that an increase of 83-day/year when PM10 exceeded 100 $\mu g/m^3$ was associated with an elevated RR of 1.57 [95% confidence interval (CI), 1.08, 2.27] with males being more likely than females to develop chronic productive cough. A statistically significant association was also seen between development of chronic productive cough and days when PM10 exceeded 80 $\mu g/m^3$ but not for

lower cutoffs of PM10 nor for PM10 mean concentration. Regression coefficients and RR for selected increments are shown for cutoffs of 80 and 100 of PM10 as well as mean concentration in Table IV. Multiple logistic regression models were run for development (1977-1992) of overall chronic bronchitis (either cough or sputum type) using the same set of covariates as Table III. The estimated RR for developing definite symptoms of overall chronic bronchitis (either cough or sputum type) in 1992 as associated with an increase of 83-days/year when PM10 concentrations exceeded 100 $\mu g/m^3$ was 1.71 (95% CI: 1.27, 10.53) ($p = 0.0007$).

Multipollutant Analyses. To ascertain whether or not the observed associations between PM10 and development of chronic productive cough might be due to surrogate relationships with other monitored air pollutants, other air pollutants were each in turn substituted for PM10 in the multiple logistic regression model and the statistical significance of their regression coefficients ascertained using the likelihood ratio test. Also, these other pollutants were added to the PM10 model one-at-a-time to form two-pollutant models. Models with more than two-pollutants were not considered in order to avoid problems of multicollinearity. Table V shows pairwise correlations for average estimated ambient concentrations of air pollutants over the study time period - (1973-1992) except for SO_4, which was only monitored since 1987. None of the pollutants SO_4, SO_2, or O_3 were significantly associated with chronic productive cough when they were used instead of PM10 in the multiple logistic regression model. When these other pollutants were added one-at-a-time to form two-pollutant models, the PM10 coefficient remained stable or increased.

DISCUSSION

The strength and limitations of this study have been discussed extensively in previous papers.[1,2] We will discuss only those points pertinent to the 1992 updated analyses.

1. **Stability of chronic productive cough.** Although our algorithm required individuals to have productive cough for at least two-years before being classified as definite, the majority of individuals who had developed chronic productive cough between 1977 and 1987 no longer met the criteria of definite in 1992 (see Table II). The multiple logistic regression of Table III was rerun using, as an alternative outcome, development of persistent prevalence of chronic productive cough, i.e., 57 new cases in 1987 who still had definite symptoms in 1992. The regression coefficient for PM10 for development of persistent prevalence remained positive but was only about 1/3 the magnitude of that in Table IV. The estimated standard error of the coefficient increased, due to the smaller number of new cases and the regression coefficient was no longer statistically significant ($p > 0.69$). The coefficients for age and education remained nonsignificant. The coefficients for gender, frequency of childhood colds and smoke pack years remained statistically significant and positive with the coefficient for gender increasing. Thus the effects of past smoking and childhood colds appeared to be more persistent than those of ambient concentrations of PM10. It is possible that the most persistent cases of chronic productive cough, as associated with PM10, had already developed chronic productive cough by 1977 and were thus excluded from this analysis. The multiple logistic regression model for PM10 (100) was repeated, for prevalence of chronic productive cough in 1992, not excluding individuals with chronic productive cough in 1977. This model had 236 prevalent cases out of 2,741 with nonmissing data. The coefficients of all regression coefficients in this model were similar to those in the development of chronic productive cough model. The PM10 coefficient was more statistically significant ($p = 0.008$).

2. **Inclusion of 1977 Possible Symptoms Variable.** To determine if individuals who had possible symptoms of chronic productive cough in 1977 were more susceptible to elevated PM10 levels, we added a dummy variable for 1977 possible symptoms to the model of Table

III as well as an interaction term with PM10 (100). The interaction term was not statistically significant ($p = 0.36$) indicating that this was not the case. However those with possible symptoms in 1977 were much more likely to develop chronic productive cough by 1992 [RR = 3.25, 95% CI: = 2.32, 4.46 ($p < 0.001$)]. The other coefficients in the model remained virtually unchanged when this variable was added.

3. **Asthma, Bronchitis, Pneumonia Before Age 16.** A stratified analysis indicated that association with development of chronic productive cough and PM10 (100) in 127 individuals who had asthma, bronchitis, or pneumonia before age 16 was 10 times as strong ($p = 0.01$) then for individuals without these childhood diseases. The PM10 regression coefficient for the later group was close to that for everyone. We considered adding an interaction term of this risk factor with PM10 in the model but since it failed to reach statistical significance in our chunk tests for interaction and because of the small number of cases (only 10) we did not.

4. **Using Direct PM10 (1987-1992) Only.** Measurement error due to estimating PM10 indirectly from TSP for the years 1977-1987 could be biasing our results. To assess this possibility the multiple logistic regression model for development of chronic productive cough 1977-1992 of Table III was rerun using directly monitored PM10 (100) 1987-1992. The PM10 coefficient increased by 26% and remained statistically significant at the 0.05 level.

5. **Restriction of Interpolation Distance.** The multiple logistic regression of Table III was rerun restricting study participants to those who had 80% of monthly interpolations within 10 miles (16 km) of a monitoring station for direct PM10 (100) 1987-1992. This left 1,450 study participants in the model, of whom 89 developed chronic productive cough between 1977 and 1992. The coefficient for PM10 (100) was reduced by 15% and its level of statistical significance to $p = 0.109$. This reduction may reflect lack of contrasting concentrations of PM10 since those individuals who lived within 10 miles of a monitoring station experienced

significantly more days/year when PM10 concentrations exceeded 100 $\mu g/m^3$ than those who did not meet the criteria -- 37.5 days/year vs. 20.2 days/year ($p < 0.01$ from a t-test). This is due to the fact that monitoring stations tend to be placed in the more polluted areas.

6. **Indoor PM10**. Indoor sources of PM10 were considered as candidate covariates for the multiple logistic regression model but failed to enter the model. A major source of indoor PM10 is ETS. Previous analyses on this cohort have found ETS at home and at work to be significantly associated with development of respiratory diseases.[1,2,9] However, exposures to ETS at home and at work have declined markedly in our cohort in recent years. This may account for the lack of statistical significance of ETS exposure in current analyses. Although approximately half of our cohort had lived or worked with a smoker at sometime in the past (see Table I), by 1992 only 4.0% of females and 1.5% of males were currently living with a smoker, and only 1.5% of females and 2.4% of males were currently working with a smoker.

Adjusted estimates of ambient mean concentrations of PM10 were formed for study participants by applying an indoor penetration factor of 0.7 to time spent indoors as reported by study subjects by season in 1977, 1987, and 1992. The indoor penetration factor of 0.7 was obtained from work by Winer, et al..[10] This factor is in agreement with more recent indoor/outdoor studies of PM10 in California where an indoor penetration factor of 2/3 was found.[11] The multiple logistic regression model of Table III was rerun using adjusted mean concentration of PM10. The regression coefficient for adjusted PM10 and its standard error were very close to that for unadjusted PM10, accounting for the scale factor of 0.7. Personal exposures to PM10 in Southern California correlate more highly with indoor ($R^2 = 0.47$) than outdoor concentrations of PM10 ($R^2 = 0.14$).[11] Since estimating personal exposures in a long-term epidemiological study is not possible, we have instead considered factors that are major sources of PM10 personal exposure as covariates in health effects models.

7. **Occupational Dusts and Fumes.** Years exposed to occupational dusts and fumes was not found to be statistically significantly associated with development of chronic productive cough or overall chronic bronchitis in these analyses. This may be the result of diminishing exposures in more recent years in our cohort. In 1992 only 5.4% of females and 11.7% of males were currently exposed to dusts or fumes at work, whereas past exposures were much greater than this (see Table I).

Comparison of Current Results with Previously Published Findings from this Cohort. Abbey et al,[1] reported a statistically significant association between indirectly estimated PM10 (100) and development of overall definite symptoms of chronic bronchitis between 1977 and 1987 in this cohort (RR = 1.17; 95% CI = 1.01, 1.35) as associated with 41.7 days/year in excess of 100 $\mu g/m^3$ of PM10. However, the strength of the association was weaker than what has been currently observed for 1977-1992, using direct measures of PM10 since 1987 (RR = 1.31; 95% CI = 1.13, 1.53). This may be due to bias toward the null when using indirect measures as discussed above. A strong association was reported between development of overall chronic bronchitis 1977-1987 (RR = 1.74; 95% CI = 1.11, 2.72) as associated with 41.7 days/year directly estimated TSP in excess of 200 $\mu g/m^3$ a level roughly equivalent to 100 $\mu g/m^3$ of PM10[2] No associations have previously been found in this cohort between development of chronic bronchitis 1977-1987 and O_3, SO_2, SO_4, or nitrogen dioxide (NO_2).[2,12,13] Estimates of NO_2 concentrations included indoor sources.[13]

Development of chronic productive cough between 1977 and 1987 was found to be statistically significantly ($p < 0.03$) associated with mean concentration of particulates <2.5 microns in diameter (PM2.5) indirectly estimated from airport visibility data available on about 50% of the cohort.[14,15] These estimates of PM2.5 were for the years 1966 through 1986 but have not yet been updated through 1992.

Comparison of our Results with Others Studies of Development of Chronic Respiratory Disease and Long-Term Ambient Concentrations of PM10. Comprehensive reviews have recently been conducted of epidemiological findings as related to ambient particulate pollutants.[16,17] Jedrychowski[18] has conducted a similar review for Eastern European studies. We will focus here only on studies relating development or prevalence of chronic respiratory disease to long-term concentrations of PM10.

Prevalence of chronic bronchitis in children was found to have increased odds of 2.5 (95% CI = 1.1, 6.1) for annual mean concentrations of particulates <15 microns in diameter (PM15), elevated from 20 - 60 $\mu g/m^3$.[19] This is consistent with our findings regarding development of chronic bronchitis and PM10 in adults. Particulate pollutants have been found to be associated with emphysema, chronic bronchitis, and asthma in cross-sectional data from U.S. cities.[21,21] A prospective study in Cracow, Poland compared high (180 $\mu g/m^3$) versus low (109 $\mu g/m^3$) areas of Cracow for ambient particulates <20 microns in diameter (PM20) and found development of persistent chronic bronchitis to be significantly associated with high ambient PM20 as well as overall chronic airway disease.[22] These results are consistent with ours but cannot be directly compared as they measured PM20 rather than PM10. Also, areas of Cracow that were high in PM20 were also high in SO_2. SO_2 levels are low in our study and are not highly correlated with PM10 (see Tables I and V). There is some evidence that prevalence of chronic bronchitis is associated with ambient levels of SO_4 above 5.8 $\mu g/m^3$,[23] as well as overall COPD with SO_4 levels above 9 $\mu g/m^3$ if SO_2 and TSP were also high and above 15 $\mu g/m^3$ of SO_4 without high TSP.[24] For our cohort maximum levels of SO_4 were 10.2 $\mu g/m^3$ (see Table I) and SO_4 mean concentration was not highly correlated with days when PM10 exceeded 100 $\mu g/m^3$. We found no associations between SO_4 and development of chronic bronchitis in our cohort. This may be due to three factors -- low levels of SO_2 experienced by our cohort, particulates and SO_2 not necessarily being high when SO_4 was high, and low humidity in much of California. However, previous analyses from our study have found statistically significant associations between SO_4

and development of asthma 1977-1987.[12] Development of asthma 1977-1987 has also been found to be strongly associated with O_3 in males in our cohort.[25] Preliminary results from our data do not indicate strong associations between development of asthma 1977-1992 and particulates. A future paper will deal in-depth with development of asthma in this cohort.

There is considerable evidence that chronic bronchitis is associated with occupational air pollutants.[26] We found no significant associations with occupational dusts or fumes and development of chronic bronchitis in our cohort. As discussed above, this may be due to low current levels of occupational exposure in our cohort.

Lung function samples have been collected on a 50% sample of our cohort in 1993 and future papers will deal with decrements of lung function as associated with PM10 and other ambient air pollutants. Other papers will report on development of asthma and other respiratory diseases in this cohort and examine associations with indoor and outdoor NO_2.

In summary, we have found that development of chronic productive cough and overall chronic bronchitis was significantly elevated in non-smokers who experienced several years of many days per year when PM10 exceeded 80 $\mu g/m^3$. This level is far lower than the current 24-hour standard of 150 $\mu g/m^3$ of PM10 for the U.S.

Acknowledgements

Funding for this project provided by EPA Cooperative Agreement #CR819619-01-3 and the California Air Resources Board Contract #A933-160.

References

1. Abbey, D.E.; Hwang, B.L.; Burchette, R.J.: Estimated Long-Term Ambient Concentrations of PM10 and Development of Respiratory Symptoms in a Nonsmoking Population. Arch. Environ. Hlth. 50(2):139-150 (1995a).

2. Abbey, D.E.; Petersen, F.F.; Mills, P.K.; Beeson, W.L.: Long-Term Ambient Concentrations of Total Suspended Particulates, Ozone, and Sulfur Dioxide and Respiratory Symptoms in a Non-smoking Population. Arch. Environ. Hlth. 48(1):33-46 (1993a).

3. Beeson, W.L.; Mills, P.K.; Phillips, R.L.; et al.: Chronic Disease Among Seventh-day Adventists, A Low Risk Group. Rationale, Methodology and Description of the Population. Cancer. 64:570-81 (1989).

4. Hodgkin, J.E.; Abbey, D.E.; Euler, G.L.; Magie, A.R.: COPD Prevalence in Nonsmokers in High and Low Photochemical Air Pollution Areas. Chest. 86:830-838 (1984).

5. Abbey, D.E.; Moore, J.; Petersen, F.; Beeson, W.L.: Estimating Cumulative Ambient Concentrations of Air Pollutants: Description and Precision of Methods Used for an Epidemiological Study. Arch. Environ. Hlth. 46:281-287 (1991).

6. Hosmer, D.W.; Lemeshow, S.: Applied Logistic Regression. Goodness of Fit Test - pp. 140-145. John Wyley & Sons, Inc. NY. (1989).

7. Hosmer, D.W.; Lemeshow, S.: Applied Logistic Regression. Logistic Linearity - p.96. John Wyley & Sons, Inc. NY. (1989).

8. Abbey, D.E.; Euler, G.L; Moore, J.K,; et al.: Applications of a Method for Setting Air Quality Standards Based on Epidemiological Data. J. Air. Pollut. Contr. Assoc. 39(4):437-445 (1989).

9. Robbins, A.R.; Abbey, D.A.; Lebowitz, M.D.: Passive Smoking and Chronic Respiratory Disease Symptoms in Non-smoking Adults. Intl. J. Epidemiol. 22:809-817 (1993).

10. Winer, A.M.; Lurman; R.W.; Coyner, L.A.; et al.: Characterization of Air pollutant exposures in the California South Coast Air Basin: Applications of a New Regional Human Exposure (REHEX) Model. Final Report. Contract No. TSA 106-01-88. Statewide Air Pollution Research Center. University of California, Riverside, California (1989).

11. Ozkaynak, H.; Xue, J.; Spengler, J.; et al.: Personal Exposure to Airborne Particles and Metals: Results from the Particle Team Study in Riverside, California. JEAEE. 6(1):57-78 (1996).

12. Abbey, D.E.; Petersen, F.F.; Mills, P.K.; Kittle, L.: Chronic Respiratory Disease Associated With Long Term Ambient Concentrations of Sulfates and Other Air Pollutants. JEAEE. 3(S1):99-115 (1993b).

13. Abbey, D.E.; Colome, S.D.; Mills, P.K.; et al.: Chronic Disease Associated With Long Term Concentrations of Nitrogen Dioxide. JEAEE. 3(2):;181-202 (1993c).

14. Abbey, D.E.; Ostro, B.; Burchette, R.J.: Chronic Respiratory Symptoms Associated With Estimated Long-Term Ambient Concentrations of Fine Particulates Less Than 2.5 Microns in Diameter (PM2.5) and Other Air Pollutants. JEAEE. 5(2):137-159 (1995b).

15. Abbey, D.E.; Lebowitz, M.D.; Mills, P.K.; et al.: Long-Term Ambient Concentrations of Particulates and Oxidants and Development of Chronic Disease in a Cohort of Non-Smoking California Residents. Inhalat. Toxicol. 7:19-34 (1995c).

16. Pope, III, C.A.; Bates, D.V.; and Raizenne, M.E.: Health Effects of Particulate Air Pollution: Time for Reassessment? Envir. Hlth. Persp. 103:472-480 (1995).

17. Lebowitz, M.D.: Epidemiological Studies of the Respiratory Effects of Air Pollution. In press. (1996).

18. Jedrychowski, W.: Review of Recent Studies from Central and Eastern Europe Associating Respiratory Health Effects with High Levels of Exposure to "Traditional" Air Pollutants. Environ. Hlth. Persp. 103(S2):15-21 (1995).

19. Dockery, D.W.; Speizer, F.E.; Stram, O.; et al.: Effects of Inhalable Particles on Respiratory Health of Children. Am. Rev. Respir. Dis. 139:587-594 (1989).

20. Portney, P.R.; Urban Air Quality and Chronic Respiratory Disease. Regional Sci. Urban. Econ. 20:407-418 (1990).

21. Schwartz, J.: Particulate air pollution and Chronic Respiratory Disease. Environ. Res. 62:7-13 (1993).

22. Jedrychowski, W.; Krzyzanowski, M.: Ventilatory Lung Function and Chronic Chest Symptoms Among the Inhabitants of Urban Areas with Various Levels of Acid Aerosols: Prospective Study in Cracow. Environ. Hlth. Perspect. 79:101-107 (1989).

23. Leaderer, B.P.; Berman, M.D.; Stolwijk, J.A.J.: In: Proceedings of Fourth International Clean Air Congress, Tokyo: JUAPPA, pp. 1-4. Eds. S. Kasuga, et al. (1977).

24. WHO/EURO.: Air Quality Guidelines for Europe. Copenhagen & Geneva (1987).

25. Greer, J.R.; Abbey, D.E.; Burchette, R.J.: Asthma Related to Occupational and Ambient Air Pollutants in Nonsmokers. J. Occup. and Environ. Med. 35(9):909-915 (1993).

26. Burge, P.W.: Occupation and Chronic Obstructive Pulmonary Disease (COPD). Eur. Respir. J. 7:1032-1034 (1994).

Table I. Selected Descriptive Statistics for Cohort (Given Gender Specific Where There Were Large Differences Between Genders. Females = 1997, Males = 1094)

Mean		Max	Variable	% 0	
30.2		150.7	PM10, days/yr above 100 $\mu g/m^3$ ('73-'93)	0.3	
50.3		83.8	PM10 mean concentration '73-'92 in $\mu g/m^3$	0.2	
6.8		10.2	SO_4 mean concentration '77-'92 in $\mu g/m^3$	0.4	
4.9		10.5	SO_2 mean concentration '73-'92 in ppb	0.4	
46.5		74.9	O_3 mean 8-hr. average '73-'92 in ppb	0.7	
54		87	Age as of April 1, '77	N/A	
F 12.5[A]	M 18.6[A]	82.3	Pack-Years of Smoking	F 90.8	M 76.4
20.5[A]		72	Yrs Lived with Smoker through '92	47.8	
13.0[A]		57	Yrs Worked with Smoker through '92	55.4	
F 10.8[A]	M 21.5[A]	79	Yrs Dusts or Fumes at work through '92	F 86.5	M 54.8
14.2		19	Yrs of education through '77	--	
2.5		5	Frequency of Childhood Colds[B]	--	
N/A		N/A	Asthma, Bronchitis or Pneumonia before Age 16	91.7	

[A] Mean of non-zero values.
[B] 5 point Likert scale, 1= much less, 2= less, 3= same, 4= more, 5= much more frequent than other preschool children your age.

Table II. Gender-specific Prevalence, Development and Reversal Rates for Chronic Productive Cough or Sputum, and Overall Chronic Bronchitis (1,997 Females, 1,094 Males)

Variable	Chronic Productive Cough		Overall Chronic Bronchitis	
	Females %	Males %	Females %	Males %
1977 Prevalence	6.63	9.56	9.01	11.59
1987 Prevalence	7.31	10.69	11.27	14.08
1992 Prevalence	6.96	10.88	11.77	14.63
Development 1977-1992 out of Those without it in 1977	4.95	7.83	8.30	10.20
Development of Persistent Prevalence (Yes in '87 and '92) out of Those without it in 1977	1.45	3.05	3.26	4.27
Reversal of 1987 Development (No in '77, yes in '87, no in '92) / (no in '77 and yes in '87)	63.01	53.85	49.14	53.42

Table III. Multiple Logistic Regression for Development of Chronic Productive Cough (1977-1992) with Average Annual Days in Excess of 100 $\mu g/m^3$ of PM10 1973-1992). (n = 2,515, cases = 154)

Variable	Regression Coefficient	Increment[A]	Relative Risk[B]	95% C.I. for Relative Risk
PM10 (days/yr. when PM10 above 100 $\mu g/m^3$)	5.8128 E-3*	83	1.57	1.08, 2.27
Pack years of past smoking	0.0240**	10	1.25	1.10, 1.43
Frequency of childhood colds	0.2719**	1	1.29	1.07, 1.56
Gender	0.4340*	1=M, 0=F	1.50	1.08, 2.07
Age	0.0109	10 yrs	1.11	0.96, 1.28
Education	-0.0272	4 yrs	0.90	0.71, 1.14
Constant	-4.0648			

[A] Increment for computations of relative risks. For childhood colds, the increment is 1 point on a 5 point scale. (1 = much less, 2 = less, 3 = same, 4 = more, 5 = much more frequent than other preschool children your age.)

[B] Relative risk of increase in exposure of one increment, holding other variables in model constant.

* $p < .05$;

** $p < .01$; for statistical significance of regression coefficient

Table IV. Regression Coefficients and Estimates of Relative Risk for Development of Chronic Productive Cough for Different Incremental Increases of PM10.

Cutoff level	Regression Coefficient (S.E.)	Increment[A] in Days/Yr	Percent Population above this increment	Relative Risk Estimate	95% CI
80 $\mu g/m^3$	3.2544 E-3[B]	10.4	82	1.03	1.00 - 1.06
	(1.5984 E-3)	20.8	68	1.07	1.00 - 1.13
		41.6	55	1.14	1.00 - 1.28
		83.3	27	1.29	1.01 - 1.64
100 $\mu g/m^3$	5.8128 E-3[B]	10.4	63	1.06	1.01 - 1.11
	(2.4912 E-3)	20.8	47	1.12	1.02 - 1.23
		41.6	26	1.26	1.04 - 1.52
		83.3	14	1.57	1.08 - 2.27
		$\mu g/m^3$			
Mean	0.0054	40	69	1.22	0.84 - 1.75
Concentration	(0.0051)	50	51	1.28	0.81 - 2.00
		60	29	1.35	0.77 - 2.28
		70	17	1.41	0.74 - 2.59

[A] Increment for computing relative risk. Annual days per year 1973-1992, expressed also as hours/year. Units for mean concentration are $\mu g/m^3$.

[B] Regression coefficients from which relative risks are calculated are statistically significant ($p < .05$) by likelihood ratio test.

Table V. Pairwise Correlations of Estimated Long-Term Ambient Concentrations of Air Pollutants for Cohort[A]. (n ranges from 1,735 to 1,999)

	PM10 (100)	PM10 mc	Ozone	SO_2	SO_4
PM10 (100)	1.00	.85	.68	-.08	.36
PM10 mc	. . .	1.00	.88	.31	.75
O_3	. . .	. . .	1.00	.25	.72
SO_2	. . .	. . .	. . .	1.00	.68
SO_4	. . .	. . .	. . .	. . .	1.00

[A] 80% non missing data required on both pollutants for pairwise correlation. Each pollutant first averaged over years 1973-1992 except SO_4 for which data available only 1977-1992.

Figure 1. Distribution of PM10 Mean Concentration ($\mu g/m^3$) Averaged 1973-1992 for Study Cohort

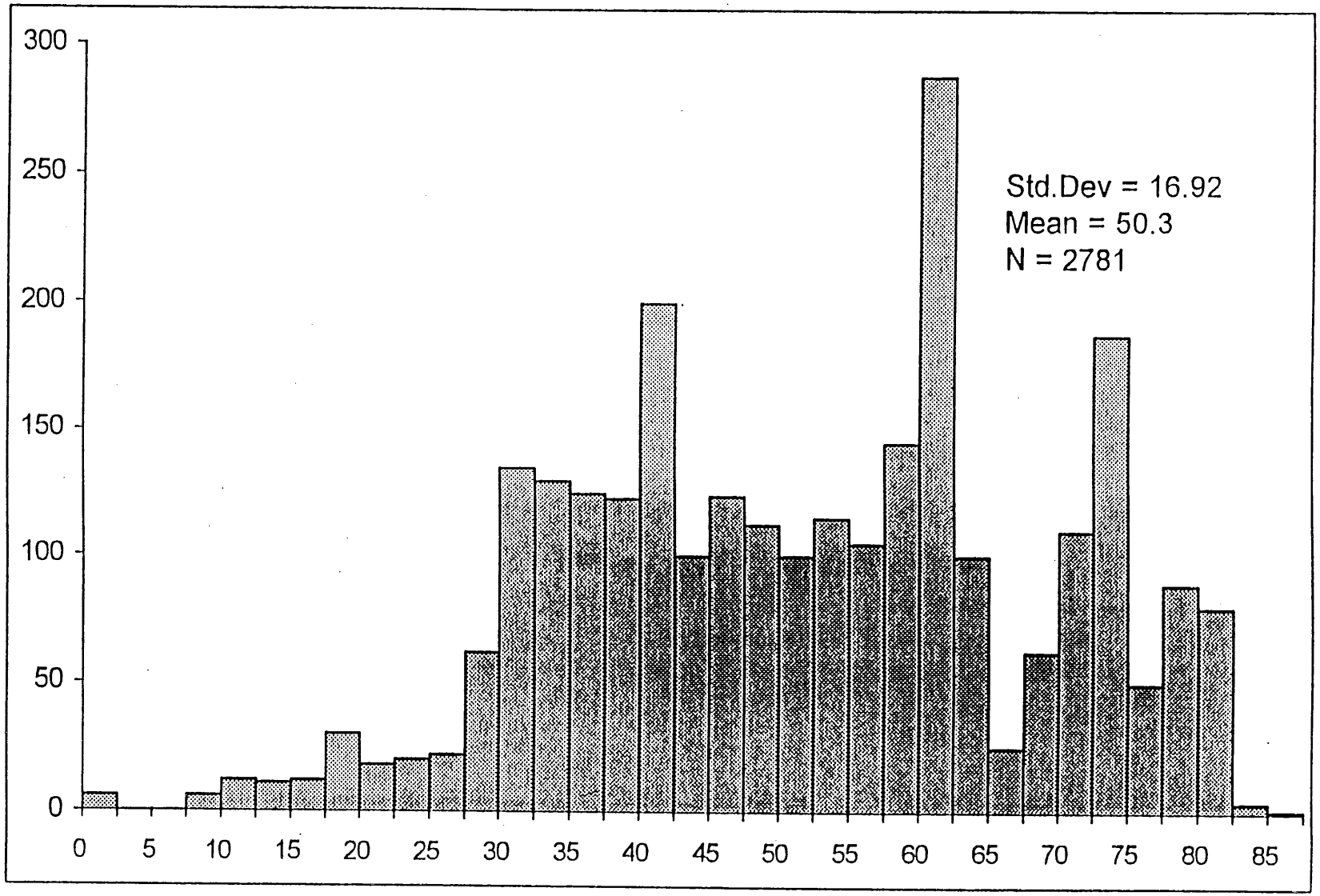
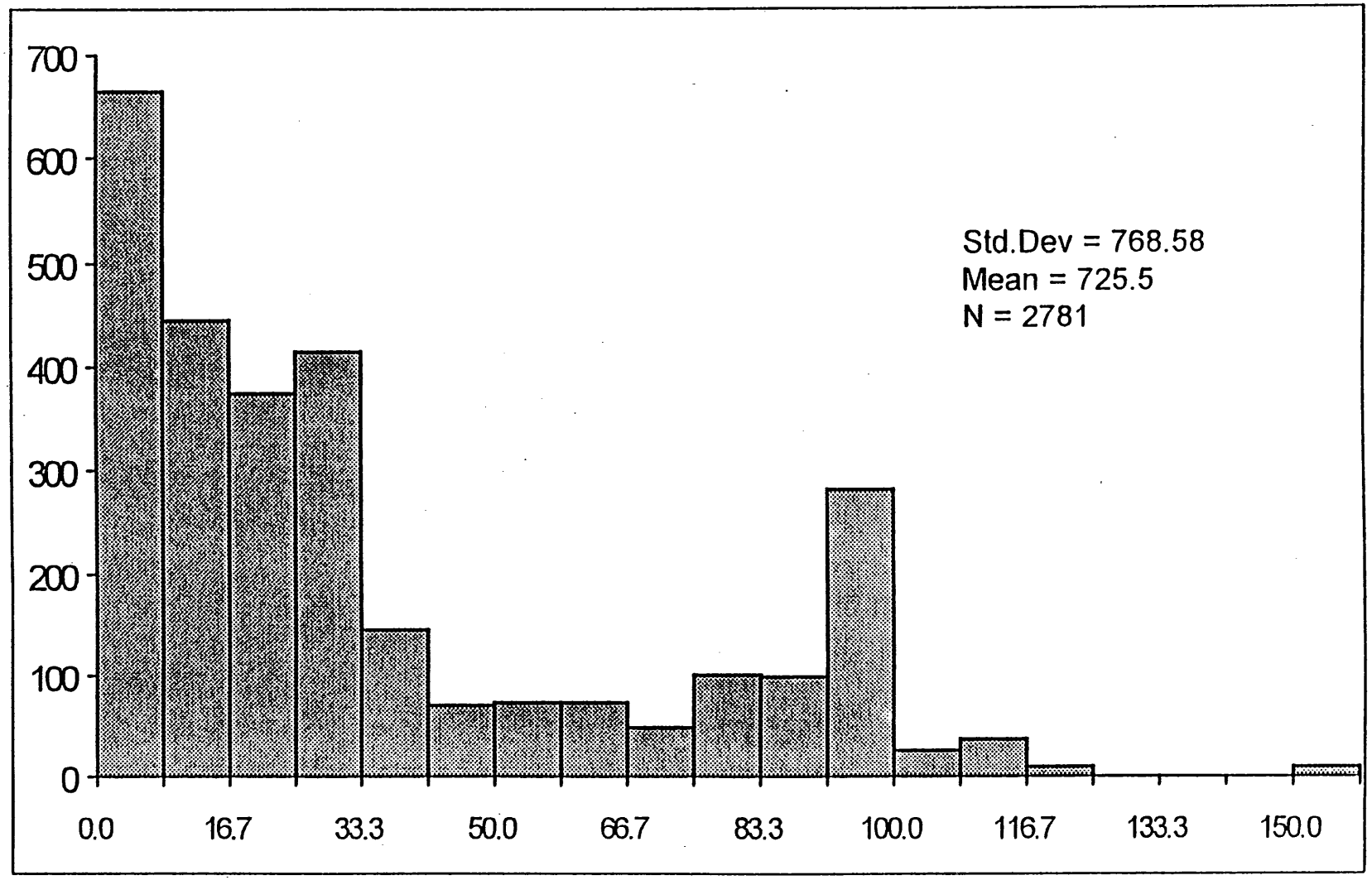


Figure 2. Distribution of Average Annual Days/Year (1973-1992) When 24-hr Mean Concentration of PM10 Exceeded 100 μ g/m^3



PM 10(100)

Assessment of the temporal relationship between daily summertime ultra-fine particulate count concentration with PM2.5 and Black Carbon Soot in Washington DC

George Allen, Eileen Abt, Petros Koutrakis
Harvard School of Public Health
665 Huntington Ave
Boston, MA 02115

INTRODUCTION

Several recent epidemiological studies have shown a significant relationship between ambient daily particulate mass concentrations and human health effects as measured by cardio-pulmonary morbidity and mortality (Schwartz, 1994). Much of the current research aimed at determining causal agents of these PM health effects focuses on fine mass (PM2.5), which is primarily the combustion-related component of PM10. Some studies have suggested that ultra-fine aerosols (typically defined as those particles that are less than 0.1 or 0.15μm in diameter) may be an important category of particulate matter to consider, as opposed to or in addition to other measures of fine particle mass (Ferin et al., 1990; Ferin, et al., 1992; Oberdorster et al., 1995). One of the postulated toxicological mechanisms for ultra-fine particles is that it is the number of particles which is most important, and not necessarily their composition or mass (Seaton et al., 1995; Chen et al., 1995). Some studies suggest that the count concentration could be important by overwhelming macrophages (Miller et al., 1995). Another possible particle metric that could be important in health-effect outcomes is particle surface area, which may serve as a condensation surface for gas phase components that are then deposited deep in the lung.

To provide more detailed temporal and size distribution data on ambient PM, the Harvard School of Public Health, in conjunction with the US EPA, has recently conducted intensive aerosol size characterization measurements in two east-coast cities during the summer peak pollution season: Washington DC during the summer of 1994, and Nashville TN during the summer of 1995. At this time, only a portion of the Washington DC data are available for analysis and presented here; the Nashville data are not currently ready for analysis.

During the summer of 1994, continuous ambient particle size distributions were measured in Washington, DC. Size distributions from 0.02 to greater than 10 μm were continuously measured using the TSI (St. Paul, MN) Scanning Mobility Particle Sizer (SMPS) and the TSI Model 3310A Aerodynamic Particle Sizer (APS). The SMPS samples particles from 0.02 to 0.7 μm, and the APS samples particles with sizes from 0.7 to greater than 10 μm. Data from the SMPS and the APS were merged to determine hourly total particle counts, total fine mass concentrations, and size distributions for count and mass concentrations. Continuous PM2.5 and black carbon soot (BC, a surrogate for elemental carbon) were measured at the same site with a Rupprecht and Patashnick (Albany, NY) TEOM and Magee Scientific (Berkeley, CA) Aethalometer respectively. Integrated low-volume 24 hour (9am to 9am local time) PM2.5 samples were also collected for method validation purposes. This paper presents the temporal relationship for 24 hour means between these pollutants and discusses the implications of these results.

METHODS

In July and August of 1994, aerosol measurements were made 4 km north northeast of downtown Washington, DC, in an urban residential area. The monitoring site was on the grounds of the McMillan Reservoir, located approximately two and one quarter miles north northeast of downtown Washington D.C. in an open clearing at the south-western end of McMillan Reservoir. Howard University borders the entire western side of the reservoir site, while three hospitals are situated along the north border. A moderately traveled road with 4 lanes (1st Street) runs parallel with the reservoir on the eastern side, approximately 200 meters from the monitoring site.

Samples were collected outdoors at 5 meters above ground level. For particle size methods, both the sampled air and the instruments were maintained at ambient temperature and relative humidity to prevent distortion of size distributions resulting from changes in the aerosol's particle bound water. All particle

data reported here for the calculation of temporal associations are 24 hour means of hourly datasets (midnight to midnight local standard time), since the relevant health effects epidemiology is based on that time interval.

Particle Size and Count Measurement Methods

SMPS. To characterize particle size distribution in the 0.02 to 0.7 μm range, a TSI Inc. Scanning Mobility Particle Sizer (SMPS) Model 3934L, consisting of a Model 3071A electrical mobility size classifier and a Model 3010 Condensation Particle Counter (CPC), was used to count particles and classify them by size based on electrical mobility characteristics. The nominal sample scan time was 6 minutes; the mean results of 2 scans were recorded. Sample aerosol for the SMPS is drawn through an impactor to remove particles above the measurement range and then exposed to a Kr-85 neutralizer to reduce the particle charge distribution to one described by the Boltzmann equilibrium. For a given particle size and rod voltage charge, the mobility is just enough to allow the measured particles to be swept past a charged rod and through a slit into a particle counter. All particles with greater mobilities than the desired size precipitate on the charged rod, and those with smaller mobilities are removed with the sheath air. Corrections are made in data processing for multiple charges on particles.

APS. The size distribution of particles in the range from 0.7 to greater than 10 μm was measured using the TSI, Inc. Aerodynamic Particle Sizer (APS) model 3310A, a laser-Doppler velocimeter. The principle of operation is the measurement of the time of flight in an accelerating air stream. Results are stored every 5 minutes in 58 channels of logarithmic size intervals between 0.5 and 30 μm. The aerodynamic particle size distribution is calculated from a previously stored calibration curve. Calibration is performed with standardized polystyrene latex spheres. Data below 0.7 and above 10 μm were discarded for this study.

Merging of data from particle sizing instruments. The SMPS and APS data were merged by analyzing the response from both methods in the size range where the measurement capabilities of the two instruments overlap at 0.7 μm. The APS measures aerodynamic diameter, which is a function of particle shape and particle density, while the SMPS measures electrical equivalent diameter, which is a function of particle size and shape, but not density. The electrical equivalent diameter was considered to be equal to the volume equivalent diameter, since a spherical particle shape was assumed (Peters et al. 1993).

Data from the SMPS and APS were each processed independently and then merged using a determined density. Data from the SMPS, which consisted of twelve minute samples, were charge corrected using TSI's SMPS software (version 1.1) and then averaged for each hour. APS data, which consisted of five minute samples, were averaged for each hour and corrected for coincidence using TSI Inc.'s APS Extra software. The SMPS data was then converted from volume equivalent diameter to aerodynamic diameter using an assumed spherical particle geometry and various aerosol densities, ranging from 1.0 to 1.4 g/cm^3. Particulate number and mass distributions were analyzed for multiple hours on various days to determine the density at which the data achieved the best fit in the overlapping region. The converted SMPS data was then merged with the APS data (Peters et al., 1995) to provide particle count and mass size distributions over the entire range of 0.02 to greater than 10 m.

Black Carbon (Aethalometer)

The Aethalometer (Magee Scientific Inc., Berkeley CA) measures "black carbon soot" (BC), a surrogate of elemental carbon (EC), in real time. The method is similar in principle and highly correlated to the

coefficient of haze (COH) parameter that has been monitored for several years in many urban areas (Allen et al., 1996). The Aethalometer BC data is much more sensitive and stable than COH, and it is scaled to elemental carbon atmospheric concentrations. Measurements are made every five minutes; the one hour LOD is 50 ng·m^{-3} BC. The method is based on the optical attenuation of light by particles collected on a 47 mm diameter pre-fired quartz fiber filter. The light source is an incandescent bulb, with an effective center wavelength of 820 nm. Using the internal, empirically determined conversion factor, the BC data from this instrument has agreed well with EC in previous comparisons. This instrument does *not* measure organic carbon or the *atmospheric* light absorption of the elemental carbon aerosol. The method is documented in detail elsewhere (Hanson et al).

Continuous Mass (TEOM®)

The Rupprecht and Patashnick (Albany, NY) model 1400a Tapered Element Oscillating Microbalance (TEOM®) is an EPA designated equivalent method for measuring PM_{10}. The TEOM provides continuous mass concentration by collecting particles on a small heated (50 °C) filter mounted on the end of a hollow tapered oscillating glass rod. The frequency of oscillation decreases as the mass on the filter increases. For fine mass measurements, a 2.5 µm size fractionating impactor replaces the 10 µm inlet. The fine mass inlet and impactor used in this study is the same design as is used in the Harvard Impactor (HI) integrated PM sampler (Marple et al., 1988). To accommodate the HI inlet, the total sample flow of the TEOM is increased from 16.7 to 20 l·min^{-1}, with the filter flow increased from 3.0 to 3.6 lpm. The performance of the TEOM for PM2.5 has been evaluated previously (Allen et al, 1995); for east-coast US sites in warm weather seasons when the aerosol composition is primarily non-volatile mass, the comparison with integrated 24 hour PM2.5 gravimetric samples collected with the Harvard Impactor sampler is very good.

RESULTS

At this time, 14 days of data from the summer of 1994 in Washington DC are available. One of those days had insufficient hours for the day, and was removed from the data set. For the remaining days, the coefficient of determination (R^2) for Pearson regression analysis was calculated for both PM2.5 and black carbon (BC) against particle counts. The results are shown in Table I. There was no relationship between PM2.5 and total count concentrations or BC and total count concentration. A scatter plot and a time-series plot of the mass and count data are shown in Figures 1 and 2 respectively. A significant relationship [p=.05] was observed between BC and PM2.5, with an adjusted R^2 of 0.25. This is shown in Figure 3.

Figure 4 shows a typical cumulative distribution for particle count and mass between .02 and 10 µm aerodynamic size. Ultra-fine particles dominate the total particle count, with about 85% having sizes less than 0.2 um diameter at ambient relative humidities. Only about 4% of the mass is in particles smaller than 0.2 µm. These data are from midnight to 1 AM on July 31, 1994, when humidity was high (85%) and particles would be expected to have had significant amounts of particle bound water (PBW) associated with them, causing them to grow in size. Even with the PBW in this example, 97% of particles by count are less than 0.5µm in diameter, and 99% are less than 0.6µm. Figure 5 shows mass distribution by size for 2 hours on August 3, 1994 (5 AM with an RH of 92%, and 2 PM with an RH of 57%), in Washington, DC. For both hours, the mass peaks at about 0.6µm. As expected, the size distribution is shifted slightly to the right (larger diameter) for the more humid early morning hour of data. The Y axis of this plot shows the relative mass, normalized to be independent of the size measurement interval (the "bin" width).

DISCUSSION

The lack of an observable association between particle count and mass concentration in ambient urban atmospheres as shown here is consistent with the characteristic lifetimes, sources, and sinks of the different types of ambient particles in urban areas. Ultra-fine particles are typically generated from local, ground level combustion sources. Their lifetime in the atmosphere is very short (typically less than 1-2 hours), thereby limiting the distance they can be transported to a few kilometers. This implies that their chemical composition reflects local source emissions of secondary aerosol precursors, and may not be similar to larger, aged, transported aerosols. The exception to this would be freshly formed sulfate aerosol when local SO_2 concentrations and humidities were high.

Ultra-fine particles are removed as they age, primarily by forming or combining with larger particles (agglomeration). The higher the concentration of ultra-fine particles, the more rapidly they grow out of that size category, since for a given particle size, the rate of coagulation is proportional to the square of the particle number concentration. In addition, the smaller the particle size, the higher the rate of coagulation (the rate of change in number concentration);given an identical particle number concentration, a .01 um particle will have an 8-fold higher coagulation rate than a .1 um particle. This short lifetime of ultra-fine particles also prevents the formation of high count concentrations during prolonged periods of poor dispersion conditions that cause elevated PM2.5 concentrations, as demonstrated by the lack of correlation between PM2.5 and particle counts.

Aerosol theory dictates that large (greater than 5μm diameter) particles *also* have much shorter lifetimes in the atmosphere (typically a few hours at most) as compared to particles between 0.3 and 1.0μm (with lifetimes of many days) where most of the combustion source-related mass is found. Particles larger than a few microns are rapidly removed by settling. A 10μm particle has a settling velocity of 0.3 cm/s or 10 meters in 5 minutes. By comparison, a 0.1 m. particle's settling velocity is .00025 cm/s (with the slip correction factor applied), or 10 meters in 46 days, slow enough for the other factors (discussed above) to account for removal of particles in the ultra-fine size range.

With the exception of black carbon which is discussed below, the size of a fine mode (combustion source-related) particle generally indicates its age. Particles larger than ultra-fines but smaller than about 0.4 to 0.5 m. are typically not fresh, but are also not usually older than about one day (eg., the sources in this size range could generally be expected to be from the regional urban area, but not from long-distance transport sources). The number concentration of particles in this size range is smaller than that of the ultra-fine size range. However these mid-size particles still have substantially higher count concentrations than the aged aerosols typical of long distance transport, and usually dominate the particle surface area measurement (surface area being proportional to the square of the spherical particle diameter). The aged aerosols (typically between .5 and 1 um) are primarily from long-distance transport sources (or regional sources during periods of severe stagnation), and are the size group of combustion aerosol particles that dominate the temporal variation of PM2.5 and PM10 measurements, driving the high episodic regional concentrations of PM2.5. Note that high ambient levels of relative humidity can increase the upper limit of all of the size classifications discussed here. A good discussion of particle sources, lifetimes and sinks as a function of size can be found elsewhere. (Wilson et al., 1996).

Black Carbon (BC) was included in this analysis since it is distinctly different from the other combustion aerosols. Unlike the other combustion aerosols, which are condensed from the gas phase and are products of post-emission chemical reactions, BC is a primary (directly emitted) pollutant with particle sizes usually peaking around 0.3 m.. Because of this, BC is useful as a surrogate for local fossil fuel

(vehicular and space heating sources) or biomass combustion sources. As a result of this difference in sources, temporal patterns of BC are distinctly different from the sulfate dominated aerosols. BC peaks with morning rush hour (Allen, 1996), while sulfates have a diurnal pattern similar to ozone, peaking in the mid-day (Wilson et al.,1991). Interestingly enough, the size range of BC aerosols is such that they may account for more of the aerosol surface area than their mean mass concentration (typically about 2 to 3 $\mu g/m^3$ in urban/sub-urban areas) suggests.

Although data for surface area were not available for this paper, the relationship between number, area, and mass (volume) is well established, and given number distributions, the surface area can be predicted. For a density of 1 and spherical shape, particulate surface area increases as the square of the particle diameter, and mass increases as the cube of the particle diameter. For example, a 1 um diameter particle has 400 times more surface area than a 0.05 um particle of similar composition and shape, and a 1 um particle weighs 8000 times more than a 0.05 um particle of similar composition and shape. If particle counts peak at approximately 0.1 um (and assuming a log-normal number concentration distribution), surface area peaks at 0.3 ($0.1^{0.5}$), and mass peaks at 0.5 ($0.1^{0.33}$) μm. The assumption above of spherical shape is reasonable for east-cost US areas where ambient humidity is usually higher than 40%, since there would be water associated with the aerosol most of the time. The density of combustion-related aerosols, including the associated water, is usually between 1.1 and 1.4, so the mass size distribution peak used in this example would actually be somewhat higher, at 0.6 to 0.7 μm.

Given these relationships between count, area, and mass, and that none of the various epidemiological models based on mass have shown any clear indication of a highly non-linear (cubic) dose-response relationship, it is unlikely that count concentration alone is the best indicator for the health effect response. However, particle surface area may still be an important parameter in this respect, since there is enough uncertainty in the dose-response curves to allow a second-order particle-related effect to account for the observed response.

CONCLUSIONS

No temporal association was observed between particle count and mass concentrations for 24 hour mean measurements. These results are consistent with the characteristics of ultrafine particles, which are generated primarily by local sources, have short lifetimes, and show small day-to day temporal variability. In contrast, the larger sub-micron particles that dominate PM2.5 concentrations in east coast urban areas (0.3 to 1 μm) are primarily from regional or long-distance transport sources and have lifetimes of many days.

If the lack of an association between count and mass concentration holds over other seasons and locations, this may make it more difficult to postulate theories of causal mechanisms for PM health effects which rely on a temporal association between particle counts or ultra-fine mass and measures of PM2.5 or PM10. However, particulate surface area may still be an important metric in assessing the health effect response to particles.

It should be noted here that these are preliminary, limited results from a single site and single season. With the small sample size used here, the ability to detect a significant association is weak. To make definitive statements on the associations presented here, additional data would be needed that includes broader seasonal and spatial measurements. However, there is nothing to suggest that these results would not be representative at least for population-based monitoring sites in east-coast urban areas during the warm weather seasons.

Further work on this topic will include data from additional days and two additional sites (Nashville TN and Boston, MA). Surface area is a particle measurement of interest with regard to health effects assessment, and will be examined along with the count and mass measurements.

ACKNOWLEDGMENTS

The authors would like to acknowledge Mark Davey from the Harvard School of Public Health (HSPH) for field site support. TSI Inc provided the SMPS used in Washington, DC. Rupprecht and Patashnick provided assistance and TEOM instruments used in this study (TEOM® is a registered trademark of Rupprecht and Patashnick Co., Inc.). The State of MD also provided TEOM monitors for this research. Thomas Peters of RTI developed the techniques used to merge the SMPS and APS data. We thank Costas Sioutas (HSPH) and William Wilson (NERL, USEPA) for their comments.

Disclaimer. The information in this document has been funded by the United States Environmental Protection Agency under cooperative agreement # CR822050-01-0 to the Harvard University School of Public Health. It has been subjected to agency review and approved for publication. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

REFERENCES

G.A. Allen, P. Koutrakis, R. Reiss et al., "Evaluation of the TEOM method for measurement of ambient particulate mass in urban areas," in Particulate Matter: Health and Regulatory Issues, VIP-49, Air & Waste Management Association, Pittsburgh, PA, pp 297-308 (1995).

G.A. Allen, P. Koutrakis, J. Lawrence, "Field validation for a real-time method for aerosol black carbon (aethalometer) and temporal patterns of summertime hourly black carbon measurements in southwestern PA," Atmos. Env., in press (1996).

L.C. Chen, C.Y. Wu, Q.S. Qu, and R.B. Schlesinger, "Number concentration and mass concentration as determinants of biological response to inhaled irritant particles," Inhal. Toxicol., 7:577-588 (1995).

J. Ferin, G. Oberdorster, D.P. Penney et al., "Increased pulmonary toxicity of ultrafine particles? 1. Particle clearance, translocation, morphology," J. Aerosol Sci, 21:381-84 (1990).

J. Ferin, G. Oberdorster, and D.P. Penney, "Pulmonary retention of ultrafine fine particles in rats," Am. J. Respir. Cell Mol. Biol., 6:535-542 (1992).

A.D.A. Hansen, H. Rosen, and T. Novakov, "The aethalometer - an instrument for the real-time measurement of optical absorption by aerosol particles," Sci. Total Environ., 36:191-196 (1984).

V. Marple, K.L. Rubow, W. Turner et al., "Low flow rate sharp cut impactors for indoor air sampling: design and calibration," J. Air Pollut. Control Assoc., 37:1303-1307 (1987).

F.J. Miller, S. Anjilvel, M.G. Menache et al., "Dosimetric issues relating to particulate toxicity," Inhal. Toxicol., 7:615-632 (1995).

G. Oberdorster, R.M. Gelein, J. Ferin, and B. Weiss, "Association of particulate air pollution and acute mortality: involvement of ultrafine particles?" Inhal. Toxicol., 7:111-124 (1995).

H. Patashnick and E.G. Rupprecht. "Continuous PM-10 measurements using the tapered element oscillating microbalance," J. Air Waste Manage. Assoc., 41:1079-1083 (1991).

T.M. Peters, H. Chein, and D.A. Lundgren, "Comparison and combination of aerosol size distributions measured with a low pressure impactor, differential mobility particle sizer, electrical aerosol analyzer, and aerodynamic particle sizer," Aerosol Science and Technology, 19:396-405 (1993).

T.M. Peters, R.W. Vanderpool, R.M. Burton et al., "Combination of aerodynamic particle sizer and scanning mobility particle sizer data in measuring ambient aerosols," in Proceedings of the American Association for Aerosol Research 1995 Annual Meeting (1995).

J. Schwartz, "Air pollution and daily mortality: a review and meta-analysis," Envir. Res., 64:36-52 (1994).

A. Seaton, W. MacNee, K. Donaldson, and D. Godden, "Particulate air pollution and acute health effects," Lancet, 345:176-178 (1995).

W.E. Wilson and H.H. Suh, "Fine and coarse particles: concentration relationships relevant to epidemiological studies," J. Air Waste Manage. Assoc., in press (1996).

W.E. Wilson, P. Koutrakis, J. Spengler, G.J. Keeler, "Diurnal Variations in Atmospheric Acidity, Sulfate, and Ammonia." Paper # 91.89.9, Presented at the 1991 Annual Meeting of the Air and Waste Management Association, June, 1991.

W.E. Wilson, L.L. Spiller, T.G. Ellestad et al., "General Motors Sulfate Dispersion Experiment: Summary of EPA Measurements". J. Air Pollution Control Assoc., 27:46-51 (1977).

Table 1. Pearson regression analysis.

N=13 days

	adj R^2
#/cc vs FM:	-0.09 [Regression slope not significant at p=.05]
BC vs FM:	0.25
#/cc vs BC:	0.06 [Regression slope not significant at p=.05]

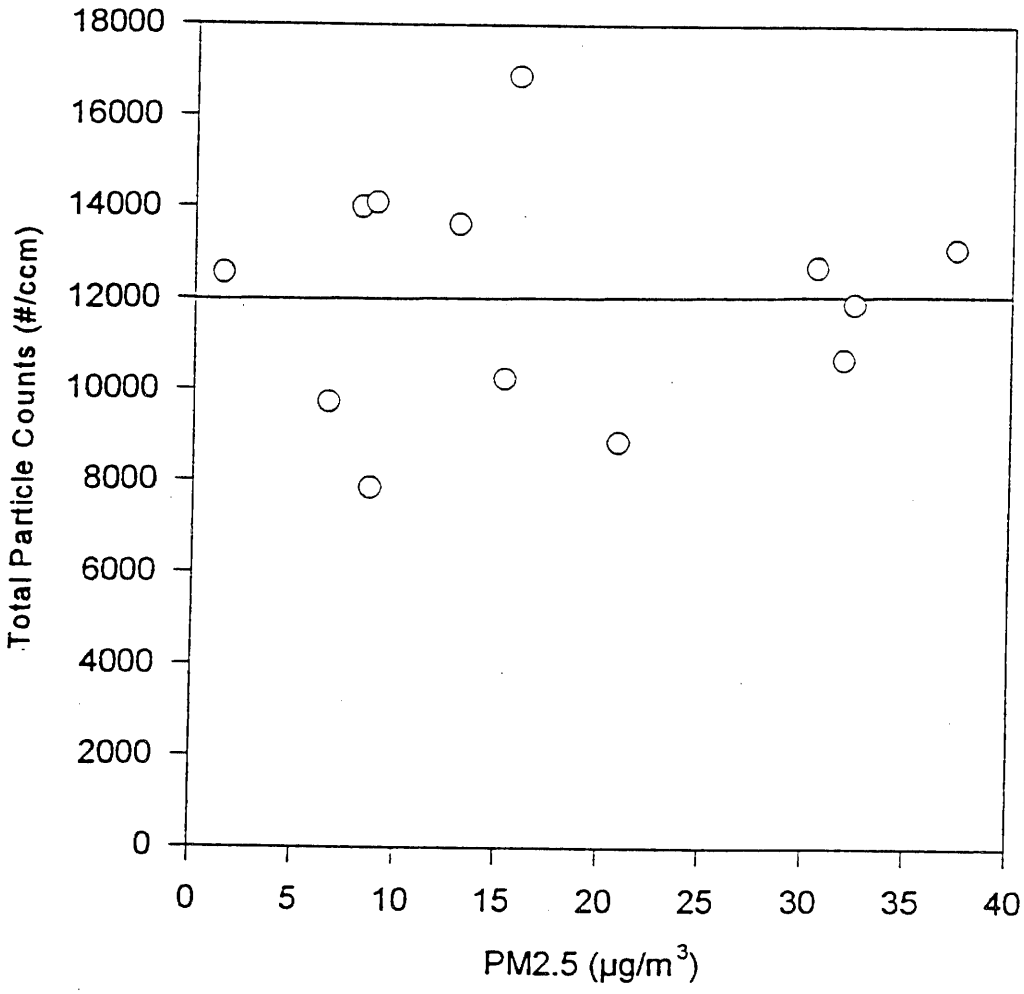


Figure 1. PM2.5 vs. Total Counts per ccm; 24 hour Means, Washington DC, Summer 1994 (13 days).

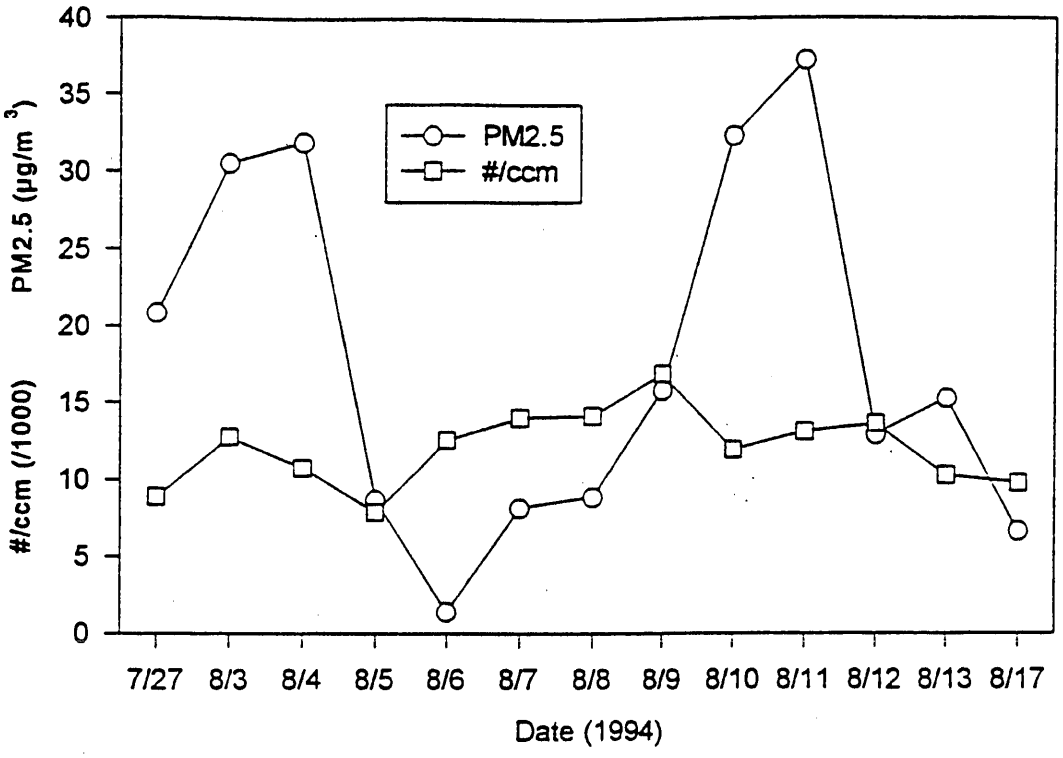


Figure 2. PM2.5 and Total Particle Counts per ccm; 24 hour means, Washington DC, Summer 1994

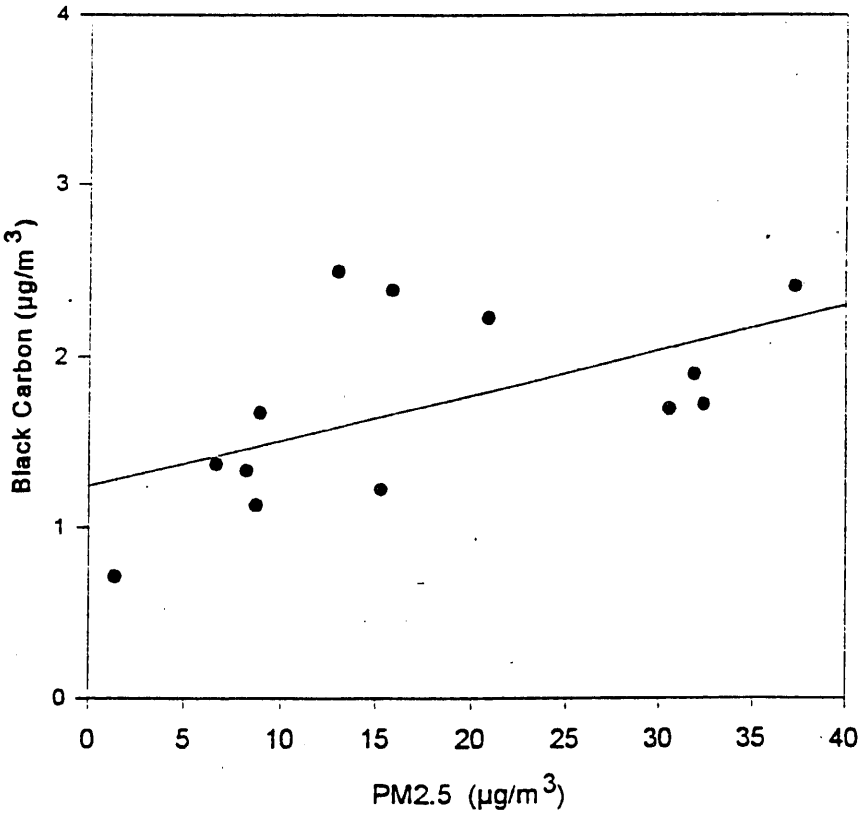


Figure 3. Black Carbon vs. PM2.5 (Regression line is shown).

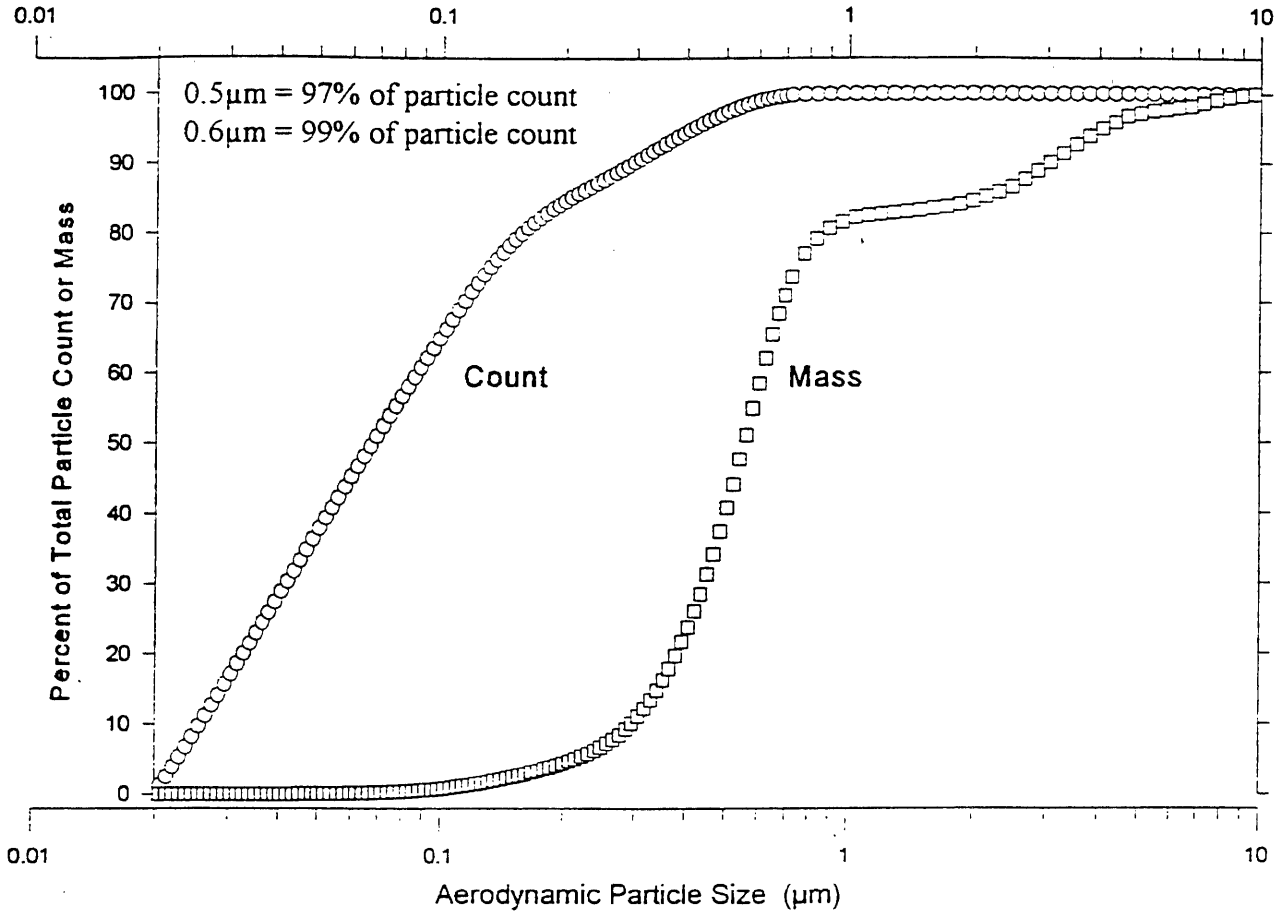


Figure 4. Typical Distribution of Cumulative Particle Counts and Mass, 0.02 to 10 m.

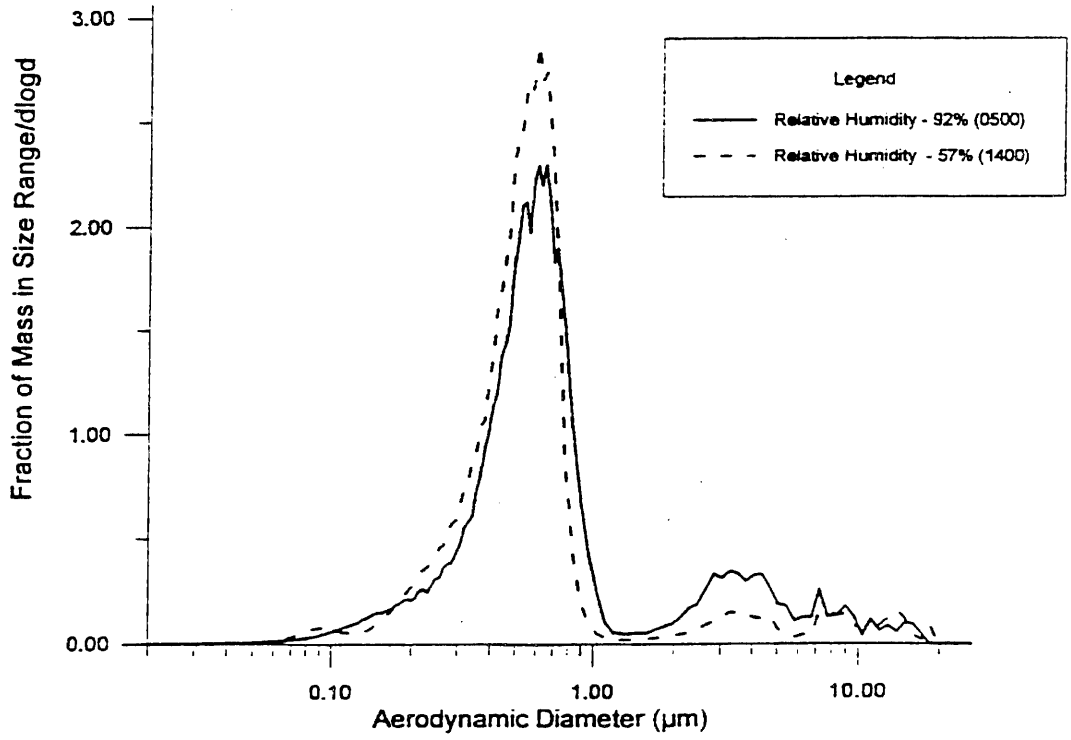


Figure 5. Typical Distribution of Particle Mass, 0.02 to 10 m. Aerodynamic Diameter

Daily Mortality and Exposure to PM_{10} Air Pollution: A Comparison of Salt Lake and Utah Counties, 1985-1993.

Kenneth M. Boucher, Joseph L. Lyon, Dean R. Lillquist. University of Utah School of Medicine, Salt Lake City, Utah 84112

Abstract

Earlier studies found an association between exposure to high levels of PM_{10} air pollution and all causes mortality (minus accidental deaths) among residents of Utah County, Utah between 1985-1993. We examined the same association for Salt Lake County, Utah, the adjacent urban county, and compared the results to the Utah County results. There is substantial air mixing between the two counties and the number of days when PM_{10} levels were 75 $\mu g/m^3$ or greater was similar, 9.6% and 12.4% respectively. For all causes mortality (estimated from a Poisson regression model for PM_{10} levels of 50 $\mu g/m^3$) the RR for Salt Lake County was 0.99 and for Utah County was 1.05. RR for cause specific mortality were also estimated for the two counties. For Salt Lake County the RR for deaths from respiratory, cardiovascular, and other causes were 0.99, 1.01, and 0.97 respectively. For Utah County they were 1.05, 1.08, and 1.02. RR by place of death were also examined. For Salt Lake County the RR was 0.99 for hospital deaths, 0.96 for nursing home deaths, and 1.02 for deaths at home. For Utah County the RR were all 1.05. Despite similar indices of air pollution, the two adjacent counties demonstrate a difference in mortality pattern for exposure to PM_{10} during the same period.

Introduction

Daily exposure to fine particulate air pollution of less than 10 microns in diameter has been associated with increased mortality within a few days of exposure in many locations within the United States, including Utah County, Utah. The association remains even after controlling for weather variables (Lyon, et al. 1995; Schwarz & Marcus, 1990; Schwarz 1991, 1993; Schwarz & Dockery 1992a, 1992b; Pope, et al. 1991, 1992, 1995).

The greatest excess mortality in most of the studies was from those dying of cardiopulmonary disease, and was present at levels of air pollution below the current National Ambient Air Quality Standard. In contrast, Lyon, et al. (1995) found the excess mortality in Utah County occurred primarily among those who did not die of respiratory causes, raising questions about the causal nature of the association between particulate air pollution and mortality. These studies have assumed a uniform exposure within a city or county, regardless of where death occurred, from pollution measurements taken at one monitoring site.

A study in Salt Lake County, Utah, which is immediately north of Utah County and shares similar weather patterns, found no association between PM_{10} and daily mortality from 1985-1990 (Styer, et al. 1995).

It has been suggested that excess mortality estimates from Poisson regression models are not robust, and are therefore particularly sensitive to measurement errors and the particular regression model used (Moolgavkar, et al. 1995). Measurement error may result from using one monitor to infer personal exposure over a large area, and may particularly be a problem in Salt Lake County, where the longest series of available PM_{10} data is from a monitor located in a corner of the county away from the majority of the population. The particular way in which time trends and weather are controlled for in the model may also lead to different results. Earlier studies of particulate air pollution in Salt Lake County (Styer, et al. (1995) and Utah Counties (Pope, et al. 1992; Lyon, et al. 1995), did not take advantage of nonparametric regression techniques such as smoothing splines, which may control better for mid- to

long-term trends in mortality due to weather or epidemics than more traditional linear or polynomial regression terms.

It is important to consider separately those groups most likely to be affected by particulate air pollution. The elderly and those with cardiopulmonary diseases are likely to be susceptible to the effects of air pollution. Location at time of death may also be important, as those dying in a hospital may have less exposure to particulate air pollution due to more sophisticated ventilation systems, compared to those dying at home or in a nursing home (Lyon, et al. 1995; Lillquist, et al. 1996).

Objectives of this study are (1) to compare estimates of excess mortality in Salt Lake County with estimates in Utah County using models which incorporate nonparametric smoothing, and with a longer series of data than in earlier Salt Lake County studies; (2) to evaluate PM_{10} exposure estimates from several sites in Salt Lake County to obtain the best possible overall exposure estimate; and (3) to compare risk estimates for various causes of death, locations at time of death and seasons in both counties, to see if a pattern of excess risk emerges which is consistent with a causal relationship.

Method

Information on daily mortality was obtained from death certificates filed with the State of Utah from June, 1985 - December, 1993 in Salt Lake County and from April 1985 - December 1992 in Utah County. Only those who were both residents of Salt Lake or Utah County and died in their county of residence of non-accidental causes (ICD-9 code less than 800) were included in the analysis. The daily mortality in Salt Lake County among this restricted group ranged from 0 to 24, with a mean of 9.46 deaths per day and a variance of 10.40. In Utah County the daily mortality ranged from 0 to 10 with a mean of 2.52 deaths per day and a variance of 2.68.

Hourly weather data was obtained from the National Climatic Data Center for Salt Lake County. Initial models used low temperature and low humidity as weather variables. Each of these was obtained by taking the minimum of the daily readings at 1 am, 7 am, 1 pm, and 7 pm. For some models low temperature was incorporated in six temperature categories: < 20 degrees, 20 - 30 degrees, 30 - 40 degrees, 40 - 50 degrees, 50 - 60 degrees, and above 60 degrees.

Age groups were categorized as <1, 1-59, 60-74, and 75+. Location at the time of death was categorized as "hospital" if it occurred at one of the hospitals in the county of residence, as "nursing home" if it occurred at a nursing home in the county of residence, and as "home or other" if it occurred at another location within the county of residence. From the death certificate information, 98.1% of deaths which did not occur in a hospital or nursing home occurred at home. Cause of death was categorized as "respiratory", "cardiovascular", or "other" by ICD-9 code as suggested in Schwarz and Dockery (1992b). The number of qualifying deaths in each county during the study period stratified by age, cause of death, and place of death is summarized in Table 1.

Synoptic weather categories as explained in Kalkstein (1993) were also considered in Salt Lake County models. Potential categorizations with from 12 to 50 weather categories were considered, and Poisson regression models were fit with 17 synoptic categories. The hourly weather data used in the synoptic categories was not available for Utah County.

Selection of PM_{10} Exposure

Care was taken to select appropriate PM_{10} measurement site(s) which would best represent population exposure in Salt Lake County. Data on PM_{10} in Salt Lake and Utah Counties was obtained from the

Table 1. Number of non-accidental deaths in salt lake county (4/85 - 12/93) and utah county (6/85 - 12/92) by age, cause of death, and place of death.

Category		Salt Lake County	Utah County
Age	< 1	922 (3.1%)	217 (3.0%)
	1 - 59	4624 (15.6%)	921 (12.9%)
	60 - 74	8573 (28.9%)	1951 (27.4%)
	75+	15545 (52.4%)	4036 (56.6%)
Total		29644 (100%)	7135 (100%)
Cause of Death	Respiratory	2492 (8.4%)	709 (10.0%)
	Cardiovascular	12617 (42.6%)	3211 (45.1%)
	Other	14555 (49.1%)	3205 (45.0%)
Total		29644 (100%)	7125 (100%)
Place of Death	Hospital	13022 (43.9%)	3445 (48.4%)
	Nursing Home	7419 (25.0%)	1612 (22.6%)
	Home + Other	9223 (31.1%)	2068 (29.0%)
Total		29644 (100.0%)	7125 (100.0%)

Environmental Protection Agency Aerometric Information Retrieval System. The largest block of PM_{10} data in Salt Lake County is from a site in Magna, Utah. From June 15, 1985, when the first PM_{10} measurement at Magna is available, until December 31, 1993, there were 2550 days with reported values out of a possible 3136. We also considered four other PM_{10} monitoring sites located in Salt Lake City and a suburb to the south of Salt Lake City. Table 2 contains the locations of all five sites as well as descriptive statistics.

Comparison of five day moving average PM_{10} at Magna with the other four PM_{10} sites considered in Salt Lake County showed that while the measurements have a high degree of correlation when the entire range of values was considered, the Magna (ma) measurements did not correlate well with the others when the Magna PM_{10} concentration was greater than 60 $\mu g/m^3$ (Table 3). The correlation of mg with North Salt Lake (n2) and downtown the Salt Lake monitor at 610 S 200 E (sl) are 0.80 and 0.86 respectively for the whole set of available data, but each is near 0.50 when restricted to mg > 60 $\mu g/m^3$.

Exploratory smoothing spline regression showed the relationship between PM_{10} and log daily mortality to be more nearly linear at sites other than Magna, although there was still a dip (Figure 1). (The Cottonwood site is not depicted as it is an every six day monitor.) The nonlinearity in the estimates from the Magna monitor was significant ($p = 0.03$), while the p-values for replacing the smoothing spline by a linear term were all > 0.25 at the other three sites.

An average of the PM_{10} measurements at the two downtown Salt Lake sites (am and sl) and Cottonwood site (cw) was used as our measure of PM_{10} exposure for Salt Lake County models. These

Table 2. Descriptive statistic on PM_{10} measurements.

County	Code	Location	Number of Days With PM_{10} Measurements	PM_{10} $\mu g/m^3$ Mean (SD)
Salt Lake County	ma	2945 S 8560 W, Magna, UT	2550	31.2 (22.6)
	n2	1795 N 1000 W, Salt Lake City, UT	2374	53.1 (30.0)
	am	500 S 261 W, Salt Lake City, UT	1605	43.9 (29.9)
	sl	610 S 200 E, Salt Lake City, UT	1164	39.6 (28.4)
	cw	5715 S 1400 E, Salt Lake City, UT	507	41.4 (29.9)
Utah County	ln	Lindon, UT	2717	44.4 (36.8)

Table 3. Pearson correlation between daily pm_{10} measurements at Salt Lake County sites.

a. All days.

Location[1]	Location[1]				
	ma	n2	am	sl	cw
ma	1	0.80	0.90	0.86	0.89
n2		1	0.84	0.82	0.80
am			1	0.97	0.94
sl				1	0.95
cw					1

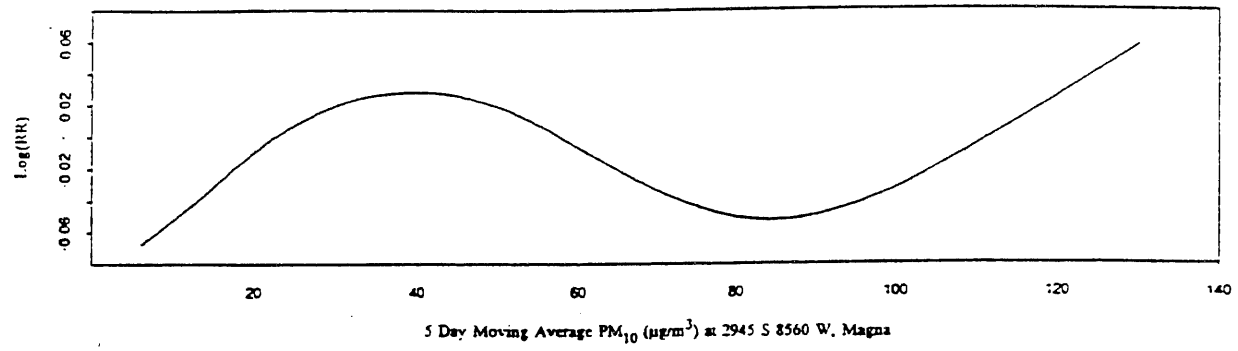
b. Days when concentration at Magna monitor exceeded 60 $\mu g/m^3$.

Location[1]	Location[1]				
	mg	n2	am	sl	cw
mg	1	0.49	0.67	0.51	0.75
n2		1	0.83	0.81	0.75
am			1	0.93	0.83
sl				1	0.94
cw					1

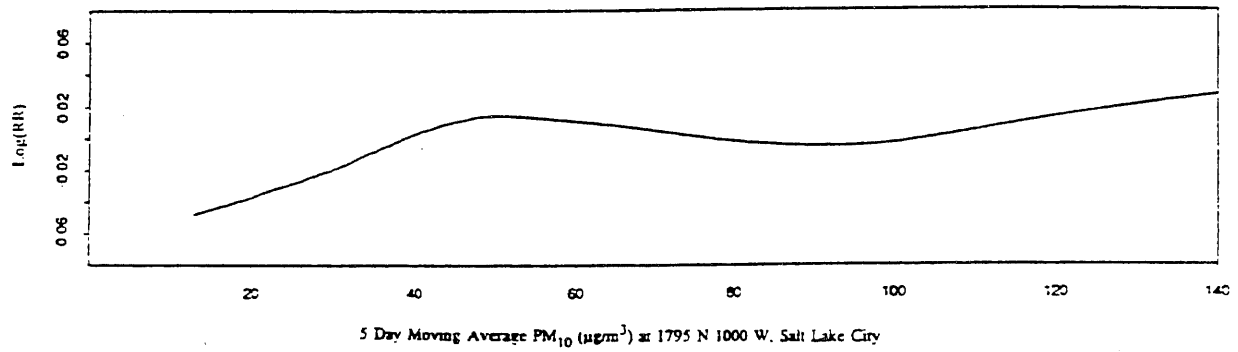
[1]See Table 2 for addresses of the locations.

Figure 1 Smoothing Spline Regression of Daily Mortality On 5 Day Moving Average PM_{10} at Several Salt Lake Valley Locations. Adjusted For Minimum Temperature, Relative Humidity, and Time Trend.

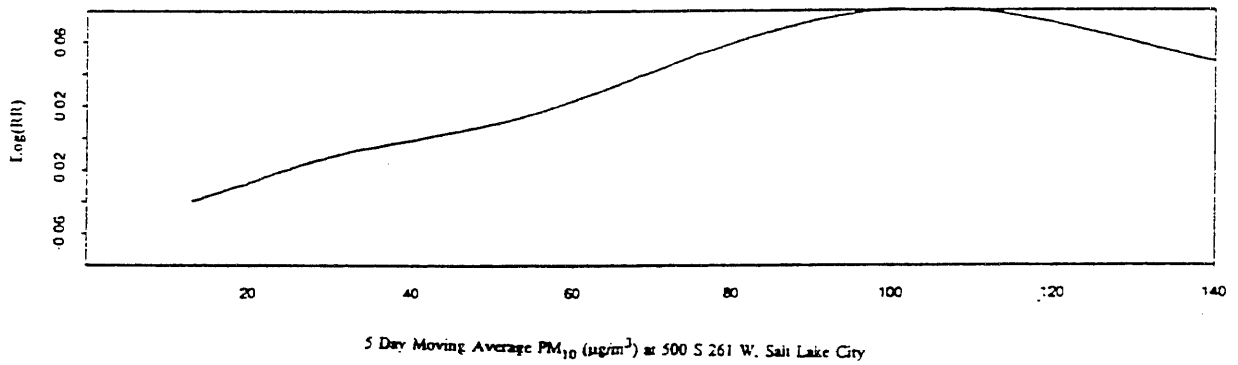
a)



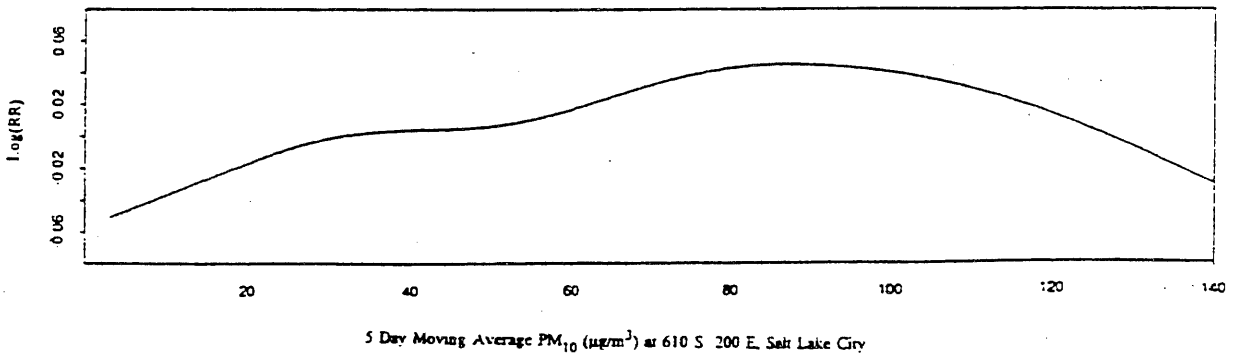
b)



c)



d)



three measurements have correlations > 0.94, have approximately equal mean values, and the monitor locations are more suitable for measuring population exposure. The mg and n2 monitors were each located to pick up pollution from an industrial site, a large copper smelter in the case of mg and several oil refineries in the case of n2. Unfortunately, the monitors am, sl, and cw each have limited data. The cw monitor, for example, was an every six day monitor.

Only those days in which at least one of these monitors had a measurement value were used in Salt Lake County models. If one or more of the monitors had missing values on these days, each value was filled in using simple linear regression before averaging. In each case the regression R^2 was greater than 0.90. We had 2063 days with a value for the new variable formed by averaging the am, sl, and cw measurements. The mean for the average exposure variable was 39.6 $\mu g/m^3$ with a standard deviation of 27.7 $\mu g/m^3$.

PM_{10} measurements for Utah County came from a monitor in Linden, Utah. There were 2717 days out of a possible 3197 with PM_{10} measurements during the study period. Although the Linden monitor was sited to capture pollution from a large steel mill, it is more centrally located, compared to the Magna monitor, as well as having the largest series of PM_{10} data within Utah County.

Regression Models

The autocorrelation in daily mortality at lags from 1 to 30 days ranged from -0.1 to 0.1 in Salt Lake County and from -0.05 to 0.05 in Utah County. Poisson regression models assuming independent mortality counts were believed to be more appropriate than a generalized estimating equation (GEE) approach. (See Moolgavkar, et al. 1995, and Mori, et al. 1996 for a discussion of GEE in this context). Due to the low autocorrelation, no prior filtering of the mortality time series was performed. All Poisson regression models were fitted using the statistical package SPLUS (MathSoft, Inc., Seattle, Washington).

A baseline Poisson regression model for total non-accidental deaths was chosen in each county before the incorporation of PM_{10}. The baseline models for each county were fit independently of one another. The Akaike information criterion (AIC) was used to select the optimal model using a stepwise procedure. Weather terms in the model were allowed to have one of three forms: 1) temperature categorized in 10 degree Fahrenheit increments, 2) a smoothing spline of temperature with from one to ten degrees of freedom (df), or 3) 17 synoptic weather categories (in Salt Lake County only). Time trends were incorporated in the models in the form of smoothing splines in time with from 1 to 100 df. A term with categories for day of the week was allowed. All models were adjusted for age.

In Salt Lake County the selected baseline model included a 40 df smoothing spline for date, a one df smoothing spline of temperature, four age categories, and categories for day of the week. In Utah County, the baseline model included six age categories, a two df smoothing spline for temperature, and two df smoothing spline for date. Categories for day of the week were not selected.

PM_{10} was then incorporated in each county to get a final model for total non-accidental deaths. Models with current day PM_{10} or an average of current day plus up to nine previous days PM_{10} were considered. Again the AIC was used to determine an appropriate model. The final model contained five-day moving average PM_{10} in Utah County and four-day moving average PM_{10} in Salt Lake County. Sensitivity to the particular model chosen was investigated by independently varying the number of degrees of freedom for the smoothing splines and the number of days of moving average. The smoothing spline for temperature was replaced by synoptic weather categories and by every 10 degree Fahrenheit temperature categories. Single day PM_{10} with lags of from 1 to 10 days were also considered. The smoothing splines were replaced by a local regression smoother loess provided by SPLUS with approximately the same degree of smoothing. Finally, a robust poisson smoother provided by SPLUS was used. The robust

smoother dampens the contribution to the deviance of data points which are far from their estimated means.

The same baseline model was then used for stratified analyses by place of death and season, by cause of death and season, by cause of death and age, and by year.

Results

There were 29664 deaths in Salt Lake County and 7125 deaths in Utah County that met the study criteria. The coefficients and standard error (SE) for PM_{10} in Salt Lake and Utah Counties were 0.000183 (SE = 0.000324) and 0.001017 (SE = 0.000423) respectively. The overall estimate of relative risk of non-accidental death from 50 $\mu g/m^3$ PM_{10} was 0.99 in Salt Lake County and 1.05 in Utah County. These risk estimates are adjusted for time trend, age, temperature, and in Salt Lake County only, for day of the week. No adjustment was made for overdispersion when calculating 95% confidence intervals (here or elsewhere). The estimated overdispersion for the total mortality models was less than one percent.

Models with the optimized explanatory variables were used to estimate risk ratio estimates for exposure to 50 $\mu g/m^3$ PM_{10} by place of death and season. The results are shown in Table 4. The comparisons between the two counties appear to be somewhat inconsistent. By location alone, in Utah County there appears to be an equal significant risk (RR = 1.05) regardless of the location at time of death, while in Salt Lake County there appears to be a small increase in risk only for those dying at home (RR = 1.01).

By season alone, the risk from particulates in Utah County appears to be highest in winter and spring (RR = 1.09 for winter and RR = 1.09 for spring), while in Salt Lake County winter and spring shows no increase in risk (RR = 0.98 and RR = 0.88 respectively), with the maximum risk ratio in Salt Lake County occurring in summer (RR = 1.02). Utah County appears to have no increased risk in summer (RR = 0.95).

Stratification by location and season gives no statistically significant relative risk estimates, with considerable variation between the two counties, although some of the individual estimates are high. The highest risk in Utah County was for those dying in a nursing home in winter (RR = 1.19). The highest estimated risk in Salt Lake County was for those dying in a hospital in summer (RR = 1.14).

Models with the optimized explanatory variables were used to estimate relative risk stratifying by cause of death and age, and by cause of death and season, with the results given in Tables 5 and 6. Stratifying by cause alone, the highest risk ratio in both counties was for dying of cardiovascular disease (RR = 1.01 in Salt Lake County and RR = 1.08 in Utah County), and next highest was for dying of respiratory disease (RR = 0.99 in Salt Lake County, RR = 1.05 in Utah County), with the smallest increase in risk for dying of other causes (RR = 0.97 in Salt Lake County, and RR = 1.02 in Utah County).

Stratified by age, the largest risk ratio was in the 1-59 age group for both Salt Lake and Utah Counties (RR = 1.04 and 1.12 respectively), with the age group 60 - 74 at no increased risk from PM_{10} (RR = 0.98 in Salt Lake County and RR = 0.96 in Utah County).

By age and cause of death, the largest and smallest relative risk estimates in both counties occurred for those under age one dying of cardiovascular and respiratory causes, respectively. The confidence intervals for these estimates are very large, however, and both contain one. Only one significant doubly stratified estimated risk ratio occurred for dying of cardiovascular disease among the 75+ age group (RR = 1.05).

Table 4. Mortality rate ratios and 95% confidence intervals for exposure to 50 $\mu g/m^3$ pm_{10}, stratified by place of death and season.

Season	Hospital		Nursing Home		Home + Other		Total	
	Salt Lake County[5]	Utah County[6]	Salt Lake County[5]	Utah County[6]	Salt Lake County[5]	Utah County[6]	Salt Lake County[5]	Utah County[6]
Winter[1]	0.94 (0.88, 1.00)	1.02 (0.85, 1.22)	0.99 (0.91, 1.07)	1.19 (0.93, 1.53)	1.03 (0.95, 1.11)	1.12 (0.89, 1.42)	0.98 (0.94, 1.02)	1.09 (0.96, 1.23)
Spring[2]	0.90 (0.75, 1.08)	1.12 (0.99, 1.25)	0.89 (0.70, 1.13)	1.05 (0.88, 1.26)	0.87 (0.71, 1.08)	1.11 (0.95, 1.30)	0.88 (0.78, 1.00)	1.09 (1.01, 1.19)
Summer[3]	1.14 (0.85, 1.38)	1.01 (0.90, 1.14)	0.82 (0.55, 1.01)	0.87 (0.73, 1.03)	1.01 (0.72, 1.44)	0.95 (0.80, 1.11)	1.02 (0.83, 1.24)	0.95 (0.88, 1.04)
Autumn[4]	0.96 (0.81, 1.13)	1.05 (0.95, 1.16)	0.88 (0.71, 1.10)	1.13 (0.99, 1.30)	0.99 (0.81, 1.20)	1.07 (0.94, 1.22)	0.95 (0.85, 1.06)	1.08 (1.04, 1.15)
Total	0.99 (0.94, 1.04)	1.05 (0.99, 1.12)	0.96 (0.91, 1.03)	1.05 (0.96, 1.14)	1.01 (0.96, 1.07)	1.05 (0.98, 1.14)	0.99 (0.96, 1.02)	1.05 (1.01, 1.10)

All models adjusted for age. Smoothing splines were used to adjust for temperature and time trend. Salt Lake County models adjusted for day of the week. Salt Lake models use four day moving average PM_{10} and Utah County models use five day moving average PM_{10}.

1 Winter months include December, January, and February.

2 Spring months include March, April, and May.

3 Summer months include June, July, and August.

4 Autumn months include September, October, and November.

5 Salt Lake County models included a 40 df smoothing spline for time trend, a 1 df smoothing spline for temperature, categories for day of the week, and are age adjusted (using age categories). Five day moving average PM_{10} is used in Salt Lake County models.

6 Utah County models included a 2 df smoothing spline for time trend, a 2 df smoothing spline for temperature, and are age adjusted (using age categories). Five day moving average PM_{10} was used in Utah County models.

Table 5. Mortality rate ratios and 95% confidence intervals for exposure to 50 $\mu g/m^3$ pm_{10} stratified by cause of death and age.

Age	Respiratory Deaths		Cardiovascular Deaths		Other Deaths		Total	
	Salt Lake County[1]	Utah County[2]	Salt Lake County[1]	Utah County[2]	Salt Lake County[1]	Utah County[2]	Salt Lake County[1]	Utah County[2]
< 1	0.24 (0.04, 1.26)	0.50 (0.08, 3.12)	1.43 (0.004, 475)	2.88 (0.67, 12.3)	0.98 (0.82, 1.18)	1.07 (0.85, 1.34)	0.96 (0.80, 1.15)	1.07 (0.85, 1.33)
1 - 59	1.12 (0.79, 1.58)	1.00 (0.59, 1.69)	1.08 (0.92, 1.27)	1.11 (0.89, 1.37)	1.02 (0.93, 1.12)	1.13 (0.98, 1.29)	1.04 (0.96, 1.13)	1.12 (1.00, 1.25)
60 - 74	1.13 (0.95, 1.36)	0.96 (0.74, 1.35)	1.00 (0.91, 1.10)	0.96 (0.84, 1.09)	0.94 (0.87, 1.03)	0.97 (0.85, 1.10)	0.98 (0.93, 1.04)	0.96 (0.89, 1.05)
75 +	0.91 (0.79, 1.04)	1.10 (0.94, 1.28)	1.01 (0.95, 1.07)	1.12 (1.05, 1.21)	0.96 (0.89, 1.03)	0.99 (0.92, 1.09)	0.98 (0.94, 1.02)	1.08 (1.02, 1.14)
Total	0.99 (0.89, 1.10)	1.05 (0.92, 1.19)	1.01 (0.96, 1.06)	1.08 (1.02, 1.15)	0.97 (0.93, 1.01)	1.02 (0.95, 1.09)	0.99 (0.96, 1.02)	1.05 (1.01, 1.10)

1 Salt Lake County models include a 40 df smoothing spline for time trend, a 1 df smoothing spline for temperature, indicator variable for day of the week, and are age adjusted (using age categories) if appropriate. Four day moving average PM_{10} is used in these models.

2 Utah County models included a 2 df smoothing spline for time trend, a two df smoothing spline for temperature, and are age adjusted (using age categories) if appropriate. Utah County models use a five day moving average for PM_{10}.

Table 6. Mortality rate ratios and 95% confidence intervals for exposure to 50 $\mu g/m^3$ pm_{10} stratified by cause of death and season.

Season	Respiratory		Cardiovascular		Other		Total	
	Salt Lake County[5]	Utah County[6]	Salt Lake County[5]	Utah County[6]	Salt Lake County[5]	Utah County[6]	Salt Lake County[5]	Utah County[6]
Winter[1]	0.97 (0.85, 1.10)	1.02 (0.69, 1.50)	1.02 (0.96, 1.09)	1.27 (1.06, 1.51)	0.95 (0.90, 1.01)	0.92 (0.76, 1.12)	0.98 (0.94, 1.02)	1.09 (1.01, 1.23)
Spring[2]	1.12 (0.76, 1.64)	0.99 (0.76, 1.30)	0.94 (0.78, 1.13)	1.12 (0.99, 1.26)	0.80 (0.67, 0.96)	1.11 (0.98, 1.26)	0.88 (0.78, 1.00)	1.09 (1.01, 1.19)
Summer[3]	1.57 (0.79, 3.14)	0.93 (0.70, 1.23)	0.98 (0.72, 1.35)	0.98 (0.87, 1.11)	0.97 (0.75, 1.27)	0.93 (0.82, 1.06)	1.02 (0.83, 1.24)	0.95 (0.88, 1.04)
Autumn[4]	0.91 (0.60, 1.35)	1.13 (0.94, 1.36)	0.92 (0.78, 1.10)	1.07 (0.97, 1.19)	0.96 (0.82, 1.12)	1.06 (0.95, 1.18)	0.95 (0.85, 1.06)	1.08 (1.04, 1.15)
Total	0.99 (0.89, 1.10)	1.05 (0.92, 1.19)	1.01 (0.96, 1.06)	1.08 (1.02, 1.15)	0.97 (0.93, 1.01)	1.02 (0.95, 1.09)	0.99 (0.96, 1.02)	1.05 (1.01, 1.10)

[1] Winter months include December, January, and February
[2] Spring months include March, April, and May
[3] Summer months include June, July, and August
[4] Autumn months include September, October, and November
[5] Salt Lake County models included a 40 df smoothing spline for time trend, a 1 df smoothing spline for temperature, categories for day of the week, and are age adjusted (using age categories). Five day moving average PM^{10} is used in Salt Lake County models.
[6] Utah County models included a 2 df smoothing spline for time trend, a 2 df smoothing spline for temperature, and are age adjusted (using age categories). Five day moving average PM_{10} was used in Utah County models.

There was considerable seasonal variation in the rate ratio estimates for cause of death, with the greatest rate ratio occurring in Salt Lake County for dying of respiratory causes in summer (RR = 1.57). The confidence interval for this estimate contains one, however, and the corresponding rate ratio for Utah County is less than one. The largest rate ratio for Utah County was for dying of cardiovascular disease in winter (RR = 1.27). The corresponding Salt Lake County risk estimate was only slightly above one (RR = 1.02).

RR's were estimated separately by year for each county (Table 7). There is considerable variability, with the RR's for Salt Lake County varying from 0.81 to 1.16 and those from Utah County varying from 0.96 to 1.18, but in six of the seven years with estimates from both counties, the Utah County estimate is higher. There is only one year with a statistically significant RR in Salt Lake County (RR = 0.81 in 1987) and two with statistically significant RR's in Utah County (RR = 1.10 in 1986 and RR = 1.13 in 1989).

Discussion

Monitor Location

The dip in estimated mortality risk ratios when PM_{10} ranges from 60-100 $\mu g/m^3$ as measured by the Magna monitor can most easily be explained as due to exposure measurement error. Evidence for this explanation is provided by the lack of correlation of Magna PM_{10} concentrations with those measured at other locations in the Salt Lake Valley. The monitor in Magna (mg) was placed to measure pollution from a large copper smelter. The one in North Salt Lake (n2) was placed to measure pollution from several oil refineries. It would not be surprising if neither of these monitors gave good estimates of population exposure. A real decline in the risk as exposure to PM_{10} increases seems highly implausible.

Risk Estimates

The overall estimated statistically significant risk ratio of 1.05 for exposure to 50 $\mu g/m^3$ in Utah County contrasts with an estimate of 0.99 for Salt Lake County. In addition to the estimates for total mortality, there is a pattern of higher risk ratio estimates in Utah County. All the estimates for place of death and cause of death are higher in Utah County. Three of four estimates for season, and the same for age are higher in Utah County.

The pattern of risk ratios for cause of death is consistent with a causal relationship, as the estimates for respiratory and cardiovascular death are higher than for other causes. The seasonal estimates are consistent with a causal relationship in Utah County (where the RRs are highest in winter and spring when PM_{10} is high), but not in Salt Lake County, where the highest RR is in summer). The overall risk estimates for age are highest for the group one might expect to be at lowest risk, age 1 - 59.

No clear pattern of risk emerges upon double stratification. Although there were some very high risk ratios (RR = 1.57 for respiratory causes in Salt Lake County in summer, 2.88 for cardiovascular deaths in Utah County in winter), these were balanced either by risk ratios of less than one in the alternate county, or by extremely large confidence intervals. Both of these are indicative of small sample size. Comparing the two counties, three-fourths of the risk estimates were higher for Utah County than the comparable estimate for Salt Lake County.

The estimated risk in Utah County is consistent with the RR = 1.05 - 1.07 reported in the meta analysis of Schwarz (1994) for 100 $\mu g/m^3$ total suspended particulate (TSP), since PM_{10} concentrations are roughly 50% of TSP (Pope, et al. 1992). The estimate from Salt Lake County is not consistent with

Table 7. Total mortality rate ratios from exposure to 50 $\mu g/m^3$ pm_{10}, by year.

Year	Rate Ratio For Salt Lake County[1]	Rate Ratio For Utah County[2]
1985	—	0.98 (0.82, 1.16)
1986	1.16 (0.88, 1.54)	1.10 (1.01, 1.21)
1987	0.81 (0.68, 0.98)	0.96 (0.74, 1.24)
1988	0.90 (0.80, 1.00)	1.07 (0.96, 1.19)
1989	1.01 (0.94, 1.09)	1.13 (1.01, 1.26)
1990	1.11 (0.97, 1.26)	1.18 (0.97, 1.42)
1991	0.97 (0.92, 1.03)	0.99 (0.89, 1.10)
1992	0.94 (0.87, 1.02)	1.05 (0.89, 1.19)
1993	1.00 (0.91, 1.10)	—

[1] All rate ratio estimates for Salt Lake County are from Poisson regression models incorporating a 5 df smoothing spline for time trend, a 1 df smoothing spline for temperature, categories for day of the week, and categories for age.

[2] All rate ratio estimates for Utah County are from Poisson regression models incorporating a 2 df smoothing spline for time trend, a two df smoothing spline for temperature and categories for age.

Schwarz and might be suspect for this reason. On the other hand, the estimate for Salt Lake County is obtained from more data (i.e., more deaths) than the estimate for Utah County and may be more reliable.

Sensitivity Analysis

A limited sensitivity analysis is presented for the total mortality model (Table 8). The relative risk estimates do not appear to be very sensitive to the specific model. For every model considered, the rate ratio for Salt Lake County is less than the rate ratio for Utah County. The rate ratio estimate vary from 0.98 - 1.01 in Salt Lake County and from 1.02 - 1.06 in Utah County, depending on the model. In no case is the estimate for Salt Lake County statistically significant. In all but three of the models the rate ratio for Utah County is statistically significant, and the statistically insignificant models all relate exposure to death at most one day later. All models which related exposure to death two or more days later in Utah County had statistically significant results.

An indication of harvesting can be found in Utah County by putting an eight to ten day lag PM_{10} in the model. The rate ratio for 50 $\mu g/m^3$ PM_{10} at an eight day lag is 0.96 with 95% CI = (0.93, 1.00).

Particulate Composition

Toxicity of particulates may depend on composition as well as total mass. A possible explanation for the observed variation in risk estimates in Salt Lake and Utah Counties is a difference in the particulate mix, particularly if the mix is more hazardous in Utah County when the particulate concentrations are high.

An examination of the contribution of various elements to total PM_{10} filter weight in Salt Lake and Utah Counties was carried out by Walton, et al. (1995). The filters were analyzed by the State of Utah Department of Air Quality on days when PM_{10} concentrations exceeded 120 $\mu g/m^3$. The relative importance of the components of PM_{10} as a percentage of total filter weight is approximately the same in both counties, although the exact percentages differ. There is more organic and elemental carbon, sodium and chlorine in the particulate mix in Salt Lake County. Utah County appears to have more calcium, potassium ion, iron, and zinc. The greatest difference between the two counties (measured as a ratio of percentages) was in iron, with Utah County having more than twice as much iron in the particulate mix as a percent of total filter weight. The contribution of iron to the total filter weight was only 1.06% in Salt Lake County and 2.39% in Utah County. We note that several researchers have reported changes in epithelial cells after in vitro exposure to residual oil fly ash (Ghio, 1996; Devlin, 1996). These changes were reported to be transition metal dependent.

Copollutants

Daily measurements of ozone, SO_2, NO_2, and carbon monoxide from the Utah Division of Air Quality were available for Salt Lake County. These were taken at 135 South State Street (CO only) and 610 S 200 E (the others) in downtown Salt Lake City. We incorporated these into Poisson regression models, with PM_{10}, individually and as a group (Table 9). The rate ratio estimates for PM_{10} were slightly though not significantly reduced. The rate ratio estimates for the copollutants were generally larger than one. The only statistically significant rate ratio estimate was for carbon monoxide (RR = 1.08). As levels of ozone are generally high in Salt Lake County in summer, an adjustment for ozone was performed separately for the summer season. The rate ratio for 50 $\mu g/m^3$ PM_{10} was decreased from 1.02 to 0.92 when adjusted for ozone.

Copollutants were not incorporated into the daily time series analysis for Utah county. The levels of SO_2 found in Utah County are generally believed to be too low to be the cause of increased risk attributed

Table 8. Results of sensitivity analysis for total mortality models.

Change in Model[1]	Salt Lake County RR (95% CI)	Utah County RR (95% CI)
Same day PM_{10}	0.99 (0.96, 1.01)	1.02 (0.98, 1.06)
One day lag PM_{10}	1.01 (0.99, 1.04)	1.02 (0.98, 1.06)
Two day lag PM_{10}	1.00 (0.98, 1.03)	1.04 (1.00, 1.07)
Two day moving average PM_{10}	0.99 (0.96, 1.02)	1.03 (0.99, 1.07)
Three day moving average PM_{10}	0.99 (0.96, 1.02)	1.04 (1.00, 1.08)
Four day moving average PM_{10}	0.99 (0.96, 1.02)	1.05 (1.01, 1.09)
Five day moving average PM_{10}	1.00 (0.97, 1.03)	1.05 (1.01, 1.10)
1 df time trend smoothing spline	1.01 (0.98, 1.05)	1.05 (1.01, 1.10)
2 df time trend smoothing spline	1.01 (0.98, 1.05)	1.05 (1.01, 1.10)
10 df time trend smoothing spline	1.01 (0.98, 1.04)	1.06 (1.02, 1.11)
40 df time trend smoothing spline	0.99 (0.96, 1.02)	1.07 (1.02, 1.11)
100 df time trend smoothing spline	0.99 (0.96, 1.03)	1.07 (1.02, 1.11)
1 df temperature smoothing spline	0.99 (0.96, 1.02)	1.04 (0.99, 1.08)
2 df temperature smoothing spline	0.99 (0.96, 1.02)	1.05 (1.01, 1.10)
5 df temperature smoothing spline	0.98 (0.95, 1.02)	1.06 (1.02, 1.10)
Every 10 degree F temperature categories	0.99 (0.96, 1.03)	1.06 (1.02, 1.11)
Synoptic weather categories	1.01 (0.97, 1.05)	----
Robust Poisson Regression[2]	1.00 (0.97, 1.03)	1.04 (1.00, 1.09)
Loess smoother[3]	1.00 (0.97, 1.03)	1.06 (1.02, 1.11)

[1] The optimized model for Salt Lake County contained a 40 df smoothing spline for time trend, a 1 df smoothing spline for temperature, categories for age, categeories for day of the week, and 4 day moving average PM_{10}. The optimized model for Utah County contained a 2 df smoothing spine for time trend, a 2 df smoothing spline for temperature, categories for age, and 5 day moving average PM_{10}. Reported rate ratios are for changing one term in the model.

[2] The robust Poisson regression models contain the same terms as the optimized model.

[3] The loess smoothers for time trend and temperature have approximately the same degrees of freedom as the smooothing splines.

Table 9. Adjustments to mortality rate ratios from incorporating copollutants in the Salt Lake County Poisson regression models.

Copollutant	Mean of Copollutant	RR for 50 $\mu g/m^3$ PM_{10}	Rate Ratio For Mean Level of Copollutant	Correlation Between Estimated Coefficients
None	—	0.99 (0.96, 1.02)	—	—
None (Summer Only)	—	1.02 (0.83, 1.24)	—	—
Ozone (ppm)	0.047	0.99 (0.95, 1.02)	0.94 (0.88, 1.01)	-0.004
Ozone (Summer Only)	0.067	0.92 (0.74, 1.14)	1.18 (0.98, 1.43)	-0.31
SO_2 ($\mu g/m^3$)	37.4	0.97 (0.93, 1.02)	1.01 (0.99, 1.04)	-0.66
NO_2 (ppm)	0.054	0.92 (0.86, 0.99)	1.07 (0.96, 1.19)	-0.76
CO (ppm)	2.74	0.96 (0.92, 0.99)	1.08 (1.02, 1.14)	-0.65
All (Ozone, SO_2, NO_2, and CO)	—	0.94 (0.85, 1.05)	—	—

All models contain an 40 df smoothing spline for time, a 1 df smoothing spline for temperature, categories for day of the week and age, and four day moving averages for all pollutants in the model.

to particulates (Schwarz, 1994), and daily measurements of SO_2 are not available from the EPA. It has also been asserted that ozone could not confound the relationship between PM_{10} and acute mortality in Utah County, because ozone and PM_{10} tend to be high at different times of the year (Schwarz, 1994).

Potential Bias

There are several additional sources of potential bias which should be considered. Daily mortality, cause of death, location at time of death, and place of residence were all taken from death certificates filed with the State of Utah. No attempt was made to verify this data. It is reasonable to assume that any errors in the coding are random, and do not bias the study. Only those who died in Salt Lake County and were residents of Salt Lake County were included in the study. It is possible that severely ill patients were transported from another location to Salt Lake City and died there, but these patients were excluded from the analysis, and do not bias the risk estimates for Salt Lake County. It is possible that the Utah County estimates might be biased by severely ill patients being transferred from Utah County to neighboring Salt Lake County and dying there. There is a single tertiary care facility in Utah County which takes in over 70% of hospital admissions, and which may transfer patients to a single hospital in Salt Lake County owned by the same corporation. Between April 1985 and March 1991 less than 1% of the admissions to this facility were transferred in this manner (Lyon, et al. 1995).

Conclusion

It is difficult to determine at the present time whether the difference in risk estimates from PM_{10} in Utah and Salt Lake Counties is due to random variability, inadequate measure of exposure, confounding by a copollutant, or a difference in the particulate mix itself. Certainly better (more local) measures of exposure, more careful analysis of the interaction of PM_{10} with copollutants, and daily speciation of PM_{10} are all needed in order to sort out this complex relationship. Even more important, however, is determination of a mechanism or mechanisms by which humans die in a matter of days from exposure to levels of PM_{10} commonly found outdoors in Utah and elsewhere. Until such a mechanism is found it will remain extremely difficult to put to rest questions about the causal nature of the association between particulates and acute mortality.

References

1. Lyon, J. L.; Mori, M;, and Gao, R.: Is there a causal association between excess mortality and exposure to PM-10 air pollution? Additional analyses by location, year, season, and cause of death. Inhal. Toxicol. 7:603-614 (1995).

2. Schwarz, J. and Marcus, A.: Mortality and air pollution in London: A time series analysis. Am. J. Epidemiol. 131:185-195 (1990).

3. Schwarz, J.: Air pollution and daily mortality in Detroit. Environ. Res. 56:204-213 (1991).

4. Schwarz, J.: Air pollution and daily mortality in Birmingham, Al. Am. J. Epidemiol. 137:1136-1147 (1993).

5. Schwarz, J. and Dockery, D.W.: Particulate air pollution and daily mortality in Steubenville, Ohio. Am. J. Epidemiol. 135: 12-19 (1992a).

6. Schwarz, J. and Dockery, D.W.: Increased mortality in Philadelphia associated with daily air pollution concentrations. Am. Rev. Respir. Dis. 145:600-604 (1992b).

7. Pope, C.A.; Dockery, D.W.; Spengler, J.D.; and Raizenne, M.E.: Respiratory health and PM-10 pollution: A daily time series. Am. Rev. Respir. Dis. 144:668-674 (1991).

8. Pope, C.A.; Schwarz, J.; and Ransom, M.R.: Daily mortality and PM-10 pollution in Utah Valley. Arch. Environ. Health 47:211-217 (1992).

9. Pope, C.A.; Dockery, D.J.; and Schwarz, J.: Review of epidemioligical evidence of health effects of particulate air pollution. Inhal. Toxicol. 7:1-18 (1995).

10. Styer, P.; McMillan, N.: et al.: Effects of outdoor airborne particulate matter on daily death counts. Environ. Health Perspect. 103:490-497 (1995).

11. Moolgavkar, S.H.; Luebeck, E.G.; Hall, T.A.; and Anderson, E.L.: Particulate air polution, sulfer dioxide, and daily mortality: a reanalysis of the Steubenville data. Inhal. Toxicol. 7: 35-44 (1995).

12. Lillquist, D.R.; Lee, J.S.; Ramsay, J.R.; et al.: A comparison of indoor/outdoor PM_{10} concentrations measured at three hospitals and a centrally located monitor in Utah. Submitted to Appl Occup Environ Hyg (1996).

13. Kalkstein, L.S.: A new approach to evaluate the impact of climate on human mortality. Environ. Health Perspect. 96:145-150 (1991).

14. Mori, M.; Carter, R.A,; and Self, S.G.: Evaluation of statistical method for assessing association between daily mortality and air pollution. Submitted to Environmetrics (1966).

15. Schwarz, J.: Air pollution and daily mortality: a review and meta analysis. Environ. Res. 64:36-52 (1994).

16. Walton, Z.; Lillquist, D.; Lee, J.S.: A comparison of PM_{10} constituents in Utah and Salt Lake Counties. Unpublished Manuscript. Rocky Mountain Center for Occupational and Environmental Health, University of Utah (1995).

17. Ghio, A.J.; Carter; J.D., Samet, J.; et al.: Respiratory epithelial cell production of lactoferrin after in vitro exposure to an air pollution particle is metal dependent (Abstract). Proceedings of the 2nd Colloquium on Particulate Air Pollution and Health, Park City, Utah, 1966, University of Utah.

18. Devlin, R.B.; Carter, J.; Ghio, A.J.; Samet, J.; and Reed, W.:1996. In vitro exposure of human airway epithelial cell to an urban particulate pollutant induces IL-6, IL-8, and TNFα mRNA and protein expression (Abstract). Proceedings of the 2nd Colloquium on Particulate Air Pollution and Health, Park City, Utah, 1996, University of Utah.

Continuous assessment of indoor fine particles with a portable nephelometer

Michael Brauer, Andrea 't Mannetje, Barbara Lang
The University of British Columbia, Occupational Hygiene Program
2206 East Mall, Vancouver, BC V6T 1Z3 Canada.

Abstract

Locations where relationships between ambient particle concentrations and health outcomes have been observed share combustion processes as a common particulate source. Since individuals spend the majority of their time indoors, fine particles generated in indoor combustion processes (cooking, smoking, woodburning,) are also important for health effects assessment.

We evaluated a continuous monitor, a portable nephelometer, for the assessment of indoor particulate, produced in combustion processes. Simultaneous sampling with PM_{10} and $PM_{2.5}$ impactors was undertaken to determine the relationship between particle light scattering coefficient (σ_{sp}) and particle mass concentration in field and environmental chamber settings. Chamber studies evaluated nephelometer measurements of environmental tobacco smoke and particles produced from toasting bread and frying foods. The nephelometer technique was applied to the assessment of particulate levels in restaurants and bars with different smoking restrictions and in residential kitchens where fried foods were prepared.

σ_{sp} and particle mass were highly correlated (r^2 values of 0.47 - 0.99) over a wide range of concentrations (7 - 381 $\mu g/m^3$). Different σ_{sp} vs. particle mass slopes were observed for the different sampling environments, reflecting the influence of particle composition on light scattering. However, in similar indoor environments, the relationship between particle light scattering and mass concentration was consistent enough to use independent nephelometer measurements as estimates of mass concentrations. Indoor concentrations were significantly higher in establishments with no smoking restrictions (mean $PM_{2.5}$ concentration = 190 $\mu g/m^3$, range: 47 - 253) than in restaurants with partial smoking restrictions (mean $PM_{2.5}$ concentration = 57 $\mu g/m^3$, range: 11 - 163). Cooking was also found to be a significant source of indoor particulate as indicated by measurements in non-smoking restaurants (mean $PM_{2.5}$ concentration = 38 $\mu g/m^3$, range: 7 - 65) and in residential kitchens during cooking periods (mean $PM_{2.5}$ concentration = 75 $\mu g/m^3$, range: 14 - 201). In each environment, 5 minute average peak $PM_{2.5}$ concentrations above 400 $\mu g/m^3$ were observed. These data indicate the potential for high particulate exposures associated with cooking and environmental tobacco smoke indoors.

Introduction

In the diverse locations where associations between ambient particle concentrations and adverse health outcomes have been observed, one common feature of the particulate sources are combustion processes. Since individuals spend the majority of their time indoors, fine particles generated in indoor combustion processes (cooking, smoking, woodburning,) are also important for health effects assessment. The nature and magnitude of indoor particle exposures can change rapidly because of the multiple sources and differences in ventilation. When used with filter sampling, continuous monitoring of fine particles improves exposure assessment by characterizing the impact of time-varying indoor sources. An example of a real-time measurement device that is very easy to use, is the nephelometer, which measures the light scattering extinction coefficient (σ_{sp}) of the particles in the air every second. However, the applicability of the nephelometer for the assessment of indoor aerosol levels has yet to be studied and evaluated. One method of evaluation involves studying the correlation between particle mass and the mean light scattering coefficient. This correlation can be influenced by the size distribution of the indoor particulate. The

correlation is shown to be best for aerosols in which the mass mean diameter is 0.2-1.0 μm, which is the same size range associated with most of the mass of particles produced in combustion processes[1,2]. The correlation also depends on the composition of the aerosols.

Several studies have evaluated the relationship between the particle mass and light scattering for ambient aerosols. Waggoner and Weiss showed that for fine ambient aerosols (aerodynamic diameter < 3 μm), the nephelometer particle scattering coefficient (σ_{sp}) and mass concentrations were highly correlated (r^2 = 0.9 - 0.94) with slopes (m^2/g) of approximately 3.2 for five different sites (urban and rural) in which particulate composition was expected to differ[2]. Koenig and colleagues reported ambient woodsmoke particulate concentrations were highly correlated with one week average σ_{sp} (r^2 = 0.95) with m^2/g of 4.9[3]. Previously we have reported on the relationship between particle mass and light scattering for 22 homes in rural Mexico and in six homes in rural British Columbia where biomass combustion (primarily burning of wood and crop residues) resulted in high levels of fine particulates. Different relationships were found between the Mexico and British Columbia particulate, indicating the importance of particle composition and ambient conditions as factors affecting light scattering, even indoors[4].

Here we report on more measurements of the relationship between particle mass and light scattering for environmental tobacco smoke (ETS) and cooking particulate in chamber and field settings. In chamber experiments particles were produced from single sources: cigarettes, from burned toast and from frying foods. Measurements of PM_{10}, $PM_{2.5}$ and light scattering were then made in indoor environments where the major source of particles were from smoking and cooking: restaurants, bars and residential kitchens. As an application of this approach to particle exposure assessment, we evaluated the impact of different smoking restrictions on indoor particulate levels.

Methods

PM_{10} and $PM_{2.5}$ particulate measurements were made with inertial impactors[5] operated at a flow rate of 4 ± 0.2 L min^{-1}. During chamber experiments the impactors were connected to indoor sampling pumps (URG 3000-02Q) and to portable pumps (SKC Aircheck 224-PCXR7) during field measurements. Pump flow rates were measured before and after sampling with a calibrated rotameter (Matheson, 603). Particulates were collected on 41 mm (PTFE) Teflon membranes with a polyolefin ring (Gelman R2POJ41). Membrane filters were equilibrated 48 hours prior to pre- and post-sample weighing in a controlled atmosphere of 25 ± 0.5°C, 39 ± 7% relative humidity. All weighings were made on an analytical filter microbalance (Sartorius M3P) with a resolution of 1 μg and ± 2μg sensitivity. Two repeat weighings were made of each filter (samples and field blanks) before and after sampling.

All impactor samples were accompanied by light scattering measurements made with a portable integrating nephelometer (Radiance Research, M903). Use of this device in the assessment of indoor particle levels has been described in detail elsewhere[4]. The nephelometer measures the particle light scattering extinction coefficient (σ_{sp}) with a wavelength defining optical filter (530 nm) and a variable rate flashlamp. The instrument also has an internal datalogger and real time clock which stores σ_{sp} averages and internal instrument operating parameters. The nephelometer was operated (without heating the inlet air stream) at a flash rate of 2 hz with a signal time constant of 32 seconds. 5 minute averages were recorded for all measurements. Air was drawn into the chamber of the nephelometer with a small fan. The nephelometers was purged and calibrated after each sampling period with particle-free air, generated by passing ambient air through two 0.2 μm bacterial air vent filters (Gelman) which were connected in series. Zero-air offsets

were applied to all data prior to final processing. Span calibrations were performed, with Freon-12 (Dupont) and SF_6 as checks of span stability.

Chamber Study

A series of chamber experiments were performed in order to obtain the correlation between light scattering and particle mass for particles of a known source under controlled conditions. Three sources of particles were used: cigarettes, burned toast and emissions from frying foods. The chamber had a volume of 56.6 m^3 with unpainted gypsum walls, vinyl floor tiles and acoustic ceiling tiles. The room was also equipped with an adjustable ventilation system. The ventilation rate of the environmental room was measured using the SF_6 tracer decay method, and ranged from 0.6 - 2.4 hr_{-1} during experiments. For all experiments a small household fan was operated to facilitate mixing within the chamber. For cigarette tests, Winston 100 cigarettes were burned by connecting them to a portable pump (Aircheck Sampler Model 224-PCXR7) set at a flow of 20 ml/min, programmed to draw air through the cigarette for 60 seconds every two minutes. Mainstream smoke was collected in a series of filters, while sidestream smoke was allowed to disperse into the chamber. The burned toast particles were produced by burning bread in a regular toaster oven left on for one more minute after the toast was ready. Frozen french fries were fried at 150 °C for 15 minutes in 16 ml of Canola oil at 30 minute intervals during the sampling period. Cigarettes and toast were burned at regular intervals over the approximately 5 hour sampling duration, in order to establish a mean concentration in the room over that time. Different mean concentrations were established (on different days) by changing the frequency of lighting a new cigarette / burning toast / frying or by changing the ventilation in the chamber. Particle sizes were measured using an optical particle counter (Climet, CI-7300).

Measurements in restaurants, bars and residential kitchens

Five to six hour measurements were made in 20 restaurants and bars in Vancouver BC and vicinity. The restaurants were divided into 3 categories based on their smoking policy: 5 non-smoking restaurants, 11 restaurants with both a no-smoking (regulations require at least 40% of seating to be non-smoking) and a smoking section (restricted smoking), and 4 bars (with food service) where smoking was unrestricted. Indoor measurements were made for approximately 6 hours with two inertial impactors (PM_{10} and $PM_{2.5}$) and a nephelometer, all placed on a table. In the restaurants with both a smoking and no-smoking section, the measurement devices were situated in the no-smoking area, as close as possible to the smoking area, in order to obtain a worst-case situation for a non-smoking customer. For every restaurant/bar, the room-dimensions were measured and, every 5 minutes the number of customers and the number of burning cigarettes were counted. Similar procedures were used in residential kitchens, although sampling durations were shorter. In kitchens, samplers were operated for 15-30 minutes prior to cooking, during the cooking period and for at least 30 minutes after cooking ceased. Sampling durations were 1.5 - 3 hours and cooking periods were 24 - 54 minutes.

Results and Discussion

Correlation between particle mass and light scattering

PM_{10} and the $PM_{2.5}$ concentrations were compared with the average σ_{sp} measured by the nephelometer. This was done for three chamber experiments, as well as for the measurements in the restaurants, bars and kitchens. Table I shows the linear relationship between the average σ_{sp} measured with the nephelometer and mass concentrations for the chamber experiments.

Although there were clearly some differences in the magnitude of the slopes, for each microenvironmental / source category (with the exception of non-smoking restaurants) there was a high correlation between the two measurement techniques. Further, the slope for the chamber ETS experiments was identical to that observed in the unrestricted smoking restaurants, providing strong evidence that the ETS slope is consistent

between controlled and field settings and that ETS was the major source of particulate in the unrestricted smoking establishments. Agreement was not as great between residential kitchens where foods were fried and the chamber experiments with french fries. This could be due to the variety of foods cooked in the kitchens as well as the influence of other particulate sources such as ambient aerosols. Measurements of the particulate size distribution indicated that cigarette smoke had a count median diameter of between 0.5 - 0.6 μm (GSD = 1.4) and toast smoke had a count median diameter between 0.5 - 0.8 μm (GSD = 1.6), suggesting that particle size did not have a strong influence on the differences in nephelometer response.

Particulate levels in restaurants with different smoking policies

Table II presents the means and ranges for the particle mass measured in the restaurants/pubs with different smoking policies. While there was a clear difference between the unrestricted and restricted restaurants, there was little difference between the restricted and the non-smoking establishments. Some possible explanations for this finding were that sitting in the non-smoking section does offer some protection from particulate exposure, that the restricted smoking restaurants were busier and therefore had more cooking in a given volume, and that the $PM_{2.5}$ and PM_{10} measurements also included ambient particulate which would tend to obscure any differences in indoor levels. Non-smoking restaurants had 0.023 customers/m^3 whereas the restricted smoking restaurants had 0.13 customers/m3 (unrestricted smoking establishments had 0.036 customers/m^3), suggesting that the non-smoking establishment may indeed have had more cooking taking place, leading to higher particulate exposures. Comparing the mean light-scattering levels in the three establishments further implicated ETS, since greater differences were seen between the three types of establishments and since the light scattering measure would not be affected by larger particles (>0.1 μm) infiltrating from outdoors or from resuspended dust.

The mean cigarette count for every restaurant was calculated and found to be correlated with $PM_{2.5}$ (r=0.8) (slope = 28 μg/ m^3-cigarette, intercept=39 μg/m^3), the significant intercept of 39 μg/m^3 indicates, that even without cigarettes, fine particulate matter is present in restaurants and bars. For every restaurant we also compared the every five minute cigarette and customer counts with the continuous light scattering measurements. To determine if the number of cigarettes and/or the number of customers had any influence on the changing of the indoor level of particles over time, a linear regression for the correlation between cigarettes and σ_{sp} was performed for every restaurant. There was wide variation between establishments in the level of correlation, although in general σ_{sp} was more highly correlated with the number of customers than with the number of cigarettes. In establishments where smoking took place, cigarette and customer counts were also highly correlated. The correlation between the nephelometer σ_{sp} and the cigarette count were relatively low for the restricted smoking. Creating a lag by shifting the σ_{sp} several minutes to account for the delay of the effect of cigarettes on the air quality, did not improve the correlation. The correlations were stronger for the unrestricted establishments, probably because the number of cigarettes smoked was higher and there was a clear trend in the number of cigarettes over time, while in the other establishments the number of smoked cigarettes were lower and a more random pattern of cigarette smoking was observed. This is illustrated in Figures 1-3. In these examples the light scattering coefficient was converted to $PM_{2.5}$ levels, using the slopes indicated in Table I.

Figure 1 illustrates a restaurant where the correlation between light scattering and the number of cigarettes over time was low (r = 0.40), although there were clearly increases in σ_{sp} directly after cigarettes were smoked. In this case, customers sitting in the no-smoking section can experience an influence of cigarette smoke from the smoking area. Figure 2 shows an example of a non-smoking restaurant. Here the particulate level increases after the number of customers increases, indicating that particles originating from cooking can lead to relatively high indoor particulate levels. Figure 3 is an example of an unrestricted

smoking establishment with a mean of 42 customers people and a gradual increase in the number of cigarettes which is highly correlated (r = 0.80) with an increase in the number of customers. This contrasts with the situation observed in Figure 1, in which the number of cigarettes returns to zero periodically. To explore the influence of cooking in restaurants with smoking, $PM_{2.5}$ levels were normalized for the influence of cigarettes by dividing them with the mean number of cigarettes smoked. The mean normalized particulate level was higher in restricted smoking restaurants than in unrestricted restaurants. This suggests that other factors, such as cooking and ventilation, influence the particulate level in restaurants.

Particulate levels in residential kitchens

Measurements in residential kitchens also established the potential for high particulate exposures associated with cooking fried foods. Results are presented in Table III. Peak (5 minute) concentrations as high as 1 mg / m^3 were observed. Since all participants were instructed to cook as they normally would, only one kitchen used a range hood. Figures 4 illustrates an example of the particulate levels (converted from the nephelometer measurements) measured in one of the residential kitchens (4G). During this measurement period, ground beef, vegetables and bacon were cooked on a gas stove in a 25 m^3 kitchen for 34 minutes.

Conclusions

These measurements indicate that in similar indoor environments, the relationship between particle light scattering and mass concentration was consistent enough to use independent nephelometer measurements as estimates of mass concentrations. σ_{sp} and particle mass were highly correlated (r^2 values of 0.47 - 0.99) over a wide range of concentrations (7 - 381 $\mu g/m^3$). Differences in σ_{sp} vs. particle mass slopes were observed for the different sampling environments.

Our measurements indicated that indoor fine particle concentrations were significantly higher in establishments with no smoking restrictions (mean $PM_{2.5}$ concentration = 190 $\mu g/m^3$, range: 47 - 253) than in restaurants with partial smoking restrictions (mean $PM_{2.5}$ concentration = 57 $\mu g/m^3$, range: 11 - 163). These results are consistent with those from other studies which suggest that smoking restrictions reduce but do not eliminate ETS exposure and that even non-smoking restaurants still have significant indoor particulate sources. Ott and colleagues. (1994) measure RSP inside a sports tavern before and after the prohibition of smoking[6]. During the smoking period, the average RSP concentration was 56.9 $\mu g/m^3$ above the outdoor concentration, decreased by 77% to 13.1 $\mu g/m^3$ above outdoor, after prohibition of smoking. Lambert and colleagues compared respirable suspended particulate levels in no-smoking and smoking sections of restaurants and concluded that the simple separation of smokers and nonsmokers reduced, but did not eliminate, the exposure of nonsmokers to ETS[7].

We also found cooking to be a significant source of indoor particulate as indicated by measurements in non-smoking restaurants (mean $PM_{2.5}$ concentration = 38 $\mu g/m^3$, range: 7 - 65) and in residential kitchens during cooking periods (mean $PM_{2.5}$ concentration = 75 $\mu g/m^3$, range: 14 - 201). In all types of restaurants and in the kitchens, 5 minute average peak $PM_{2.5}$ concentrations above 400 $\mu g/m^3$ were observed. Very few studies have reported on the impact of cooking. Kamens and colleagues reported that cooking was the most important fine particle-generating activity in non-smoking homes[8], while Ozkaynak and colleagues indicated that in homes where cooking took place, cooking accounted for 25% of the indoor $PM_{2.5}$ mass[9]. These data indicate the potential for high particulate exposures associated with cooking and environmental tobacco smoke indoors.

References

1. Thomas A and Gebhart J. Correlations between gravimetry and light scattering photometry for atmospheric aerosols." Atmos Environ. 28:935-938 (1994)

2. Wagonner AP, Weiss RE. Comparison of fine particle mass concentration and light scattering extinction in ambient aerosol. Atmos Environ. 5:623-626 (1980).

3. Koenig JQ, Larson TV, Hanley QS, Rebolledo V, Dumler K, Checkoway H, Wang SZ, Lin D, Pierson WA.. Pulmonary function changes in children associated with fine particulate matter. Environ Res. 63:26-38 (1993).

4. Brauer M. Assessment of indoor aerosols with an integrating nephelometer. J Expos Anal Environ Epidemiol. 5(1):45-56 (1995).

5. Marple VA, Rubow M, Turner W, Spengler JD. Low flow rate sharp cut impactors for indoor air sampling: design and calibration. J Air Poll Contol Assoc. 37:1303-1307 (1987).

6. Ott W, Wilson NK, Klepeis, N, Switzer P. Real-time monitoring of polycyclic aromatic hydrocarbons and respirable suspended particles from environmental tobacco smoke n a home. Presented at the International Symposium on the Measurement of Toxic and Related Air Pollutants" Durham, NC May 3-6, 1994.

7. Lambert WE, Samet JM, Spengler JD. Environmental tobacco smoke concentrations in no-smoking and smoking sections of restaurants. Am J Pub Health 83:1339-1341 (1993).

8. Ozkaynak h, Xue J, Spengler J, Wallace L, Pellizazari E, Jenkins P. Personal exposure to airborne particles and metals: Results from the particle TEAM study in Riverside, California. J Expos Anal Environ Epidemiol 6(1): 57-78 (1996).

9. Kamens R, Lee C, Wiener R, Lieth D. A study to characterize indoor aprticles in three non-smoking homes. Atmos Environ 25A(5/6):939-948 (1991).

Table I. Correlations between particle mass ($PM_{2.5}$) and light scattering for different microenvironments / particle sources. *Chamber measurements. # Reported previously[4].

Particle source / microenvironment	N	slope (m2/g)	r^2	Range $PM_{2.5}$ ($\mu g/m^3$)
ETS*	5	4.7	.97	57 - 381
Toast*	6	3.7	.93	26 - 270
French fries*	5	5.3	.87	43 - 250
Unrestricted smoking	5	4.7	.99	47 - 253
Restricted smoking	11	3.6	.92	11 - 163
Non-smoking	5	0.9	.47	7 - 65
Residential kitchens	5	3.3	.89	24 - 201
Woodburning residences#	6	6.6	.84	9 - 47

Table II. Summary statistics for measurements in restaurants, bars and residential kitchens. Concentrations are approximately 6 hour averages for restaurants and 1.5 - 3 hour averages for residential kitchens.

Location	PM_{10} ($\mu g/m^3$)			$PM_{2.5}$ ($\mu g/m^3$)		
	N	Mean	Range	N	Mean	Range
Non-smoking restaurants	5	44	8 - 74	5	38	7 - 65
Restricted smoking.	10	54	24 - 89	11	57	11 - 163
Unrestricted smoking	4	190	51 - 268	4	190	47 - 253
All restaurants	19	80	8 - 268	20	79	7 - 253
All residential kitchens	6	153	18 - 281	6	75	14 - 201

Table III. Residential Kitchens. *Range hood/fan operated. G = gas stove, E= electric stove.

Home	Cooking duration (min)	Sample duration (min)	Kitchen volume (m^3)	PM_{10} ($\mu g/m^3$) Mean	PM_{10} Peak	$PM_{2.5}$ ($\mu g/m^3$) Mean	$PM_{2.5}$ Peak
1G	54	185	26	281	440	201	330
2E*	24	132	38	18	64	14	45
3E	42	89	13	70	96	70	73
4G	34	185	25	38	150	24	113
4G	NA	137	25	265	NA	139	NA
5E	24	177	16	247	1006	189	763

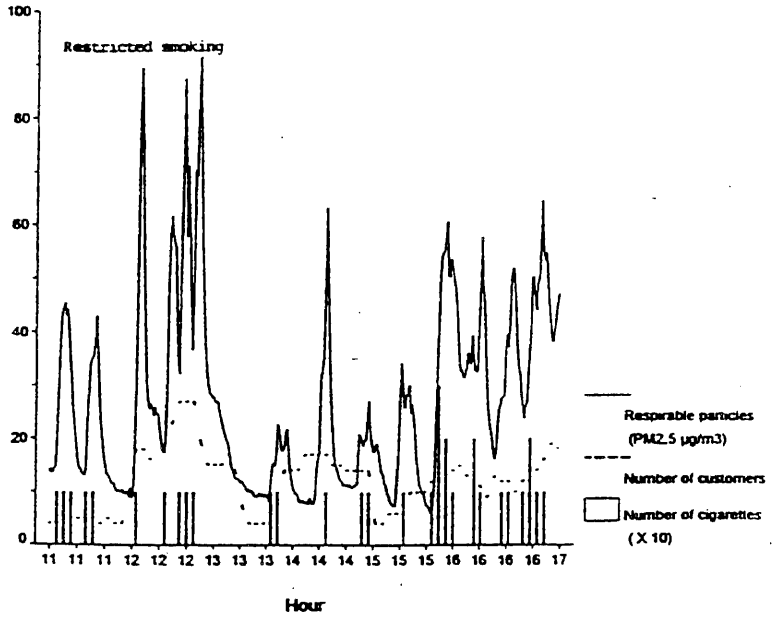


Figure 1. Estimated $PM_{2.5}$ concentrations (from light scattering measurements), customer and cigarette counts inside restaurant with restricted smoking.

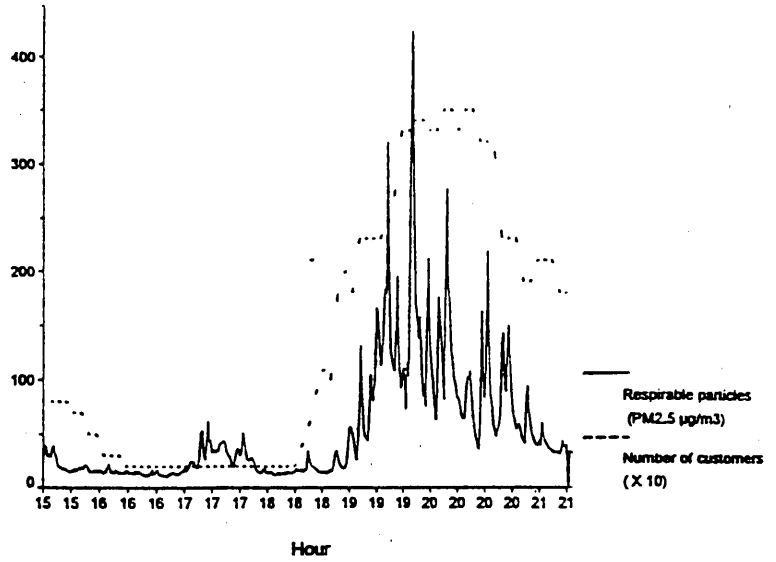


Figure 2. Estimated $PM_{2.5}$ concentrations (from light scattering measurements) and customer counts inside non-smoking restaurant.

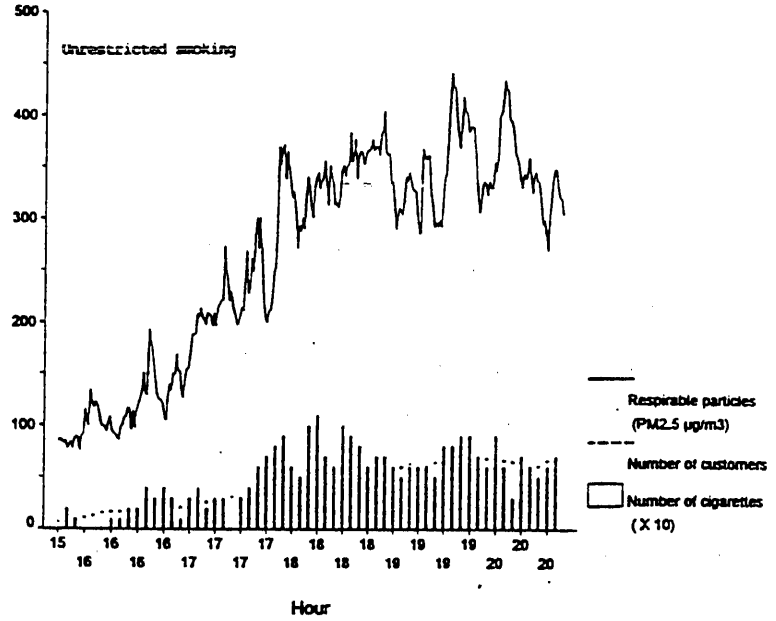


Figure 3. Estimated $PM_{2.5}$ concentrations (from light scattering measurements), customer and cigarette counts inside unrestricted smoking restaurant / bar.

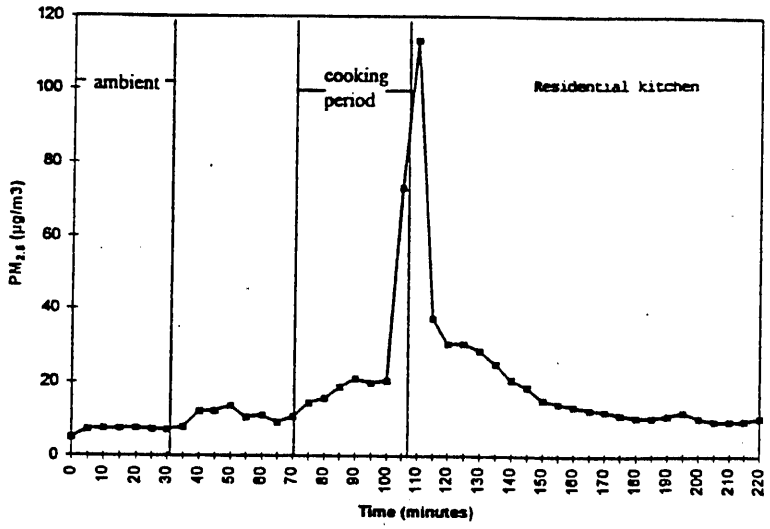


Figure 4. Estimated $PM_{2.5}$ concentration inside a residential kitchen with a 34 minute cooking period.

1

HOST RESPONSIVENESS IN HEALTH AND DISEASE:

a brief overview of animal models

Flemming R. Cassee and Leendert van Bree
National Institute of Public Health and Environment, Laboratory of Health Effects Research
Bilthoven, The Netherlands

I. Critical aspects of health effects by PM

The data base of health effects associated with exposure to particulate air pollution is rapidly growing by data from epidemiological studies (1). These data indicate that increase in daily mortality and hospital usage, exacerbation of asthma, increase of respiratory symptoms and lung function decline are associated with levels of ambient particulate matter measured or estimated as PM_{10}. These levels of particulate matter are significantly lower than those previously thought to affect and protect human health. In addition, recent epidemiological data also indicate significant associations between long-term exposure to PM_{10} or $PM_{2.5}$ levels and excess mortality, lung function decline, and respiratory symptoms, in particular at urban areas with relatively high traffic density. These data are being discussed at the level of national and international authorities in terms of whether or not the ambient air quality standards of particulate matter should be revised, both in terms of the particle size, the averaging time, as well as the level of the ambient PM standard.

In regard of possible mechanisms of respiratory toxicity of particulate matter it should be noticed that, different from effects seen with ozone-linked summer-type smog, decrements of lung function are relatively small compared to morbidity and mortality changes. This may point to a specific mode of action for PM.

The data base on health effects of ambient particulate matter points to a specific host responsiveness. Increased mortality, morbidity, and respiratory symptoms and complaints are particularly found in specific subgroups of the human population. These risk groups include 1) elderly, presumably with a weak physical condition, 2) infants, and 3) people with pre-existing cardiorespiratory diseases like congestive heart disease, pulmonary hypertension, asthma, chronic bronchitis, heart arrhythmics, emphysema, as well as airway infections. Exposure to PM will frequently result in an exacerbation of the disease and related effects (mortality and morbidity). Although there are no quantitative data on the precise extent of increased responsiveness of these subpopulations in comparison with healthy people, their increased risk is considered as rather substantial.

A particular remark have to be made on the aspect of mortality, because from the data from the epidemiological findings it can be calculated that PM exposure is associated with accelerated mortality. This has been interpreted as such that following acute PM exposure people at risk die a number of days or weeks earlier. Following chronic PM exposure mortality could be accelerated up to a number of months or years.

The mechanisms underlying the increased host responsiveness for exposure to ambient particulates are largely unknown. In general, this increased risk may depend on factors related to biology and dosimetry.

Subjects may be biologically more susceptible to effects of PM due to an exacerbation of their existing pathological condition. Examples of these susceptibility responses may be 1) increased inflammatory responses in airways during an asthmatic and infectious condition, 2) increased edema formation at pulmonary hypertension conditions, or 3) weakened cardiopulmonary circulation during cardiac malfunctioning.

Subjects may be also more susceptible for dosimetric reasons. It has been estimated that in compromised airways deposition of particles is considerable enhanced compared with healthy airways. The increased deposition is supposed to depend also on the particle size, favoring the more smaller particles to deposit at higher fractions of the exposure concentrations, however these modeling outcomes have not yet been validated in controlled experiments. The finding that enhanced dosimetry in pre-existing airway disease is an important risk factor in PM induced health effects, seems also to be supported by data on a larger impairment of lung clearance of the particles. Dosimetry models can be very useful in predicting deposition of particles in the airways. However, key and critical data gap that both empirical and theoretical models agree upon is the lack of anatomical information, especially for compromised animal models as described below.

In human clinical studies with PM it is therefore also crucial to study subjects with compromised airways, both with respect to enhanced biological sensitivity and enhanced dosimetry and deposition. Obviously, these studies have their limitations from an ethical point of view, possibly excluding prolonged (multiday) exposures and the use of the most sensitive subgroups. Animal toxicity studies have therefore to answer probably many critical questions related to the understanding of health effects of particulate matter. A crucial feature of these studies will be the selection of relevant animal models, mimicking as closely as possible many aspects of the various human diseases.

II. Animal models to study health effects of PM

There are at present no human and animal toxicology data on effects of ambient particulates that could explain these type of exposure-response relationships and provide plausible mechanisms that could relate these effects with the mass, size, number, (physico-)chemical composition, or origin of the particles. Also from a biological point of view, the effects of PM are still poorly understood. Increased airway reactivity and inflammation, enhanced coagulation of the blood and decreased the clearance of particles by alveolar macrophages are some of the biological effects which have been suggested.

The absence of toxicity data on ambient particulates is partly caused by the lack of suitable PM model components to run the controlled exposures and to the lack of relevant laboratory animal models, characteristic of compromised human subjects. A number of these models does exist but their further development is highly warranted. Laboratory animal models have been very recently introduced into PM research. These models for human pulmonary and cardiovascular diseases include (allergic) asthma, pulmonary emphysema and aging, chronic bronchitis, pulmonary hypertension, and other cardiopulmonary disorders.

The following models can be considered:

- IgE-mediated respiratory allergy (asthma)

- chronic bronchitis
- pulmonary emphysema
- aging
- pulmonary hypertension
- congestive heart failure
- respiratory infection (bacterial, viral)

Allergic asthma models

Many people suffer from asthma, a chronic illness characterized by persistent bronchial hyperreactivity to a variety of physical, chemical and pharmacological stimuli including viral infection and allergen inhalation. Histopathologic studies have shown characteristic changes like epithelial damage, deposition of collagen beneath the basement membrane, eosinophilic and lymphocytic infiltration, and hypertrophy and hyperplasia of goblet cells, submucosal glands, and airway smooth muscle. In patients with allergic asthma inhalation of allergen results in an immediate asthmatic response (a Th2 immune phenotype), which is thought to be evoked by immunglobulin E (IgE)-mediated release of mast cell and eosinophil mediators, which lead to the constriction of airway smooth muscle (spasm, of the smooth muscles of the airways) and increased mucosal exudation and inflammatory infiltrates.

A frequently used model to study effect of air pollution on the respiratory tract is a model which mimics IgE-mediated asthma. There are several protocols to induce asthma in a laboratory animal. These protocols frequently use ovalbumine to sensitize and challenge rats, mice or guinea pigs (2-5) and during a 14-21 day period. Typically, these protocols use 1-2 i.p. injections with ovalbumine (with or without alum adjuvants) and a single 5-30 min aerosol challenge at day 14. Although all of these models induce the typical symptoms of, a single challenge does not result in increased eosinophil influx and thus no signs of inflammation are observed. The protocol of Brusselle *et al* (2) (a 7 day challenge period) and an even longer protocol developed by Hessel *et al.* (4) (7 i.p. injections on alternate days and after four weeks 8 ovalbumin aerosol challenges) increased hyperresponsiveness to metacholine. In addition, increased IgE, IL2 and IFNγ levels and eosinophils and neutrophils counts in the bronchoalveolar lavage fluid were found. Although this latter model is more time consuming it better reflects human asthma and it also does not require the use of an (irritating) adjuvant.

Other models for asthma use house dust mite that seem to be more representative for human asthma. Using a similar approach as used in the ovalbumine model a stable hyperactive response during at least 3 days has been shown in rats. In contrast to the ovalbumine model, multiple antigen challenges resulted in less inflammation compared to a single challenge (6).

In general, the abovementioned asthma model allows one to study the possible effects of PM before sensitization, during sensitization, before challenge, as well as following a challenge. Although it has not been elucidated what the underlying mechanism is by which an asthmatic person responds to PM, the protocol by which PM exposure occurs just before or during the allergen challenge seems highly relevant with respect to the human exposure situations. If one assumes that PM could cause an asthmatic response by sensitizing originally healthy people following long-term exposure, the use of asthma models by which PM exposure takes place before sensitization should be considered.

The biological endpoints which should be measured in such an animal model for human asthma are airway reactivity and IgE levels, as well as airway inflammation which should be measured both

histopathologically and biochemically by lysosomal and cytoplasmic enzymes, edema, interleukines, cell differentials.

Chronic bronchitis model

Several epidemiological studies showed that air pollution may contribute to the increased prevalence of chronic bronchitis. Chronic bronchitis is characterized by chronic deep airway inflammation, increased mucus production, mucociliary dysfunction, and hypersecretion. Data presented at the previous PM colloquium (Irvine) showed a clear correlation between $PM_{2.5}$ levels and chronic bronchitis in California residents.

Increased sputum production and chronic bronchitis are associated with occupational exposure to endotoxin-contaminated organic dust. This fact was used to develop an animal model for chronic bronchitis by exposing rats to 0.3 $\mu g/m^3$ endotoxin aerosols for 3 hr/day for 3 days (7). This resulted in a rapid induction of goblet cell metaplasia, influx of neutrophils and mucus hyperproduction and no serous cells in the intrapulmonary airways that persisted during a month recovery period. Similar results were found in hamsters by (8). Thus, this model can be induced by concentrations that can be found in some urban or occupational atmospheres. A clear advantage of this model is that it can be induced in days rather than in a few weeks or months. There is no need to aerosolize the endotoxin since endotoxin can be intranasally instilled as well. This method provides a good model for studying the effects of PM in combination with chronic bronchitis, although the induced mucus cell metaplasia can also be found in airways of patients with cystic fibrosis and chronic asthma.

Another chronic bronchitis was developed at Harvard School of Public Health (Boston). Adult female rats were exposed to 230 ppm SO_2 for 5 h/day for periods of 1 day to 5 wk (9) and already used in studies with concentrated particles from ambient air by Godleski and co-workers (see elsewhere in these proceedings). Results indicate that this animal model of chronic bronchitis mimics the mucus hypersecretion, airway obstruction, and increased airway responsiveness observed in human bronchitis.

Parameters to be studied in models for chronic bronchitis are histopathological changes (Goblet cell metaplasia) and mucus production in bronchoalveolar lavage fluid.

Pulmonary emphysema and aging model

Elderly people with COPD or with pre-existing cardiorespiratory disease were found to be groups at increased risk when exposed to ambient PM. Both increased morbidity and mortality were associated with the PM levels in the environment. The use of animal models for pulmonary emphysema with or without an aging factor are needed to study and to understand the toxicological mechanism of the effect of PM on the elderly. There are at least three models that can be used: endotoxin induced pulmonary emphysema in hamsters, elastase induced pulmonary emphysema in the rat and mice (8,10) as well as a genetic mouse model (11).

The endotoxin induced model is based on intratracheal instillation of endotoxin (500μg in 200 μl, once a week for 3-5 weeks) and results in pulmonary emphysema, that in turn could be reduced for 50% by simultaneous instillation of an elastase inhibitor. Instillation of endotoxin seems to cause recruitment of neutrophils into the lung and neutrophil-derived products, including elastase, can induce pulmonary

emphysema. This protocol might serve as a model for both pulmonary emphysema and chronic bronchitis.

The second way to induce pulmonary emphysema is to instill mice with porcine pancreatic elastase (10). Three weeks after instillation the emphysema is well developed.

The Tight Skin mouse model (Tsk mouse) is a genetic model of emphysema with additional development of pulmonary hypertension in the aged mouse. This model might be relevant in relating effect of ambient PM levels on elderly populations. The emphysema that develops in the Tsk mouse has been shown to be based on elastolytic processes at an early age (11). The advantage of the Tsk mouse is that it is commercially available.

Specific parameters that should be measured in the pulmonary emphysema and aging models are lung lavage parameters (enzymes, cell differentials, cytokines) and lung histopathology. Moreover, the use of lung function tests might be useful to study the development of pulmonary emphysema in time.

Pulmonary hypertension model

An increasing number of people suffer from chronic obstructive pulmonary diseases (COPD) that can also include pulmonary hypertension. Approximately 50% of people over the age of 50 show clinical signs of COPD. It is known form epidemiological studies that this group of people shows increased death rates upon exposure to ambient PM.

Pulmonary hypertension can be induced chemically by monocrotaline in rats by giving them a single i.p. injection at the age of 5-12 weeks (12). After three weeks pathological changes of the lung, increased right heart size, and pulmonary hypertension are observed. Medial hypertrophy of the pulmonary arteries associated with enhanced production/synthesis of insoluble elastin and an increased number of elastin fragments are some of the characteristics for this model. Local production of endothelin in the lung contributes to the progression of pulmonary hypertension. This model has very recently been used in relation to PM exposures.

A second method to induce pulmonary hypertension is to expose rats to normobaric oxygen (10% O_2, 2 wk) (13). Chronic hyperoxia-induced pulmonary hypertension has also been suggested to be a result of increased synthesis of endothelin-A that acts on the ETA receptor. The normobaric exposure resulted also in right ventricular hypertrophy.

Routine markers of injury and inflammation (LDH, total protein, cell counts and differentials in the lung lavage fluid) as well as histopathological examination can be used to study effects. Moreover, arterial oxygen transport, cardiac and lung function can be used for evaluation purposes.

Congestive heart failure model

Congestive heart failure is the leading cause of death in many industrialized countries. A number of studies has shown that exposure to ambient PM can be related with an increase death rate in these individuals. Congestive heart failure in man is predominantly caused by myocardial infarction and chronic hypertension. A number of models for cardiomyopathy, increased overload and myocardial ischemia/infarction have been developed to study the pathophysiology of heart failure (for review, see 14-16).

Special hamsters strains (e.g. strain BIO 14.6, BIO T0-2) are frequently used to study congestive heart failure (17). Since at higher age (> 6 months) these animals develop cardiomyopathy that is similar to changes found in myocytes of patients with terminal heart failure, the hamster might serve as a model for heart failure in man. Cardiac failure can be ascertained by hemodynamic measurements such as mean arterial pressure, diastolic pressure and blood flows. A major advantage of this model is that the congestive heart disease that develops in these animals resembles the weakened cardiac state of individuals who have recovered from myocardial infarction (18).

Other widely used approaches include surgically induced pressure (19) and volume overload (20) and toxic myocardial depression using barbiturates, adryamycin, streptozotocin, cobalt or catacholamines (for review, see 14, 16). Rats can be used to study chronic heart failure but this model requires coronary artery ligation. Such rats can commercially be obtained. The advantage of the rat model is that is available within a month whereas the hamster model requires an early start of an experiment. The (cost of the) surgical operation might be a drawback. However, at presents, hereditary cardiomyopathy has been described in substrains of the spontaneously hypertensive rats (16) and Syrian hamsters (18). Histopathological examination of the heart (necrotic injury) and heart function measurements (electrocardiogram) are useful parameters to study in this model.

Respiratory infection model

In general, respiratory infections will lead to an modified health status and exposure to PM may decrease the host defense to bacterial and viral infections. Many animal species have been used to study effects of noxious gases or aerosols in combination with a challenge to infectious agents. The infectious agents that have been used include bacteria such as Streptococcus pyogenes, Klebsiella pneumoniae and Pseudomonas aeruginosa (21-23) but also viruses such as Influenza A and Respiratory syncytial virus (24).

Respiratory syncytial virus (RSV) is worldwide in distribution and infects almost all children during the first 2 years of life. The mouse model of RSV lung disease has been very successful in reproducing many aspects of the human disease. Infection with this virus results in the induction of T helper 2 cells and the subsequent appearance of lung eosinophilia. However, since this virus seems to effect only very young children and there are no epidemiological data about effects of ambient PM on this section of the population, the applicability of this model is questionable.

In addition, most asthma exacerbations seem to be provoked by viral respiratory infection. Preliminary studies suggest that viral infection and allergen inhalation may involve the activation of different pathways, with viral infection activating production of cytokines by airway epithelial cells. Until today there is no experimental model that mimics viral induced asthma.

III. Conclusions

When using animal models that mimic some kind of human disease one should always consider the strength an limitations of the model. An ideal model can be described by a set of criteria: a) completely mimic one or more of the different clinical symptoms, b) have sufficient precision and reproducibility to function as a bioassay, c) allow several parameters and responses to be measured and d) be undemanding in terms of costs, time and expertise (15). There are no ideal models available at present. However, it is clear that in toxicological studies the use of compromised animals is highly recommended.

In addition to studies on the relative importance of specific PM features like mass, particle size and number, chemical composition, as well as (reactive) surface area, human and animal toxicity studies on ambient PM should also focus on the causality of exposure-dose-effect relationships and on the relevance of the (ultra)fine particles in both healthy and disease-compromised lungs.

A research strategy should be developed targeted to identify:

- who is experiencing what type of effect following what kind of PM exposure,
- what are the pathological features and cellular and molecular mechanisms underlying these (adverse) effects,
- which animal models and parameters should be chosen in toxicity experiments to cover induction or progression of the various health effects by PM, and
- what are the critical factors in terms of biological susceptibility and exposure and airway dosimetry, responsible for increased responsiveness of human subpopulations

These data should provide the biological, dosimetrical, as well as mechanistical plausibility of the epidemiological findings on increased host responsiveness of exposure to ambient particulate air pollution.

References

1. Lippman, M.: The challenge of the epidemiologic evidence for excess mortality and morbidity associated with atmospheric aerosols. In: Aerosol inhalation recent research frontiers, pp 1-28. J.C.M. Marijnissen and L. Gradon, Eds. Kluwer Academic Publishers, Dordrecht, NL (1996).
2. Brusselle, G.; Kips, J.; Joos, G.; *et al.*: Allergen-induced airway inflammation and bronchial responsiveness in wild-type and interleukin-4-deficient mice. Am. J. Respir. Cell. Mol. Biol. 12: 254-259 (1995).
3. Dunn, C.J.; Elliott, G.A.; Oostvee, J.A.; *et al.*: Development of a prolonged eosinophil-rich inflammatory leukocyte infiltration in the guinea pig's asthmatic response to ovalbumin inhalation. Am. Rev. Respir. Dis. 137: 541-547 (1988).
4. Hessel, E.M.; Van Oosterhout A.J.M.; Van Ark, I.; *et al.*: Temporal association between airway hyperresponsiveness and the rpesence of inflammatory cells in bronchoalveolar lavage but not in airway tissue. Submitted for publication (1996).
5. Kung, T.T.; Jones, H.; Adams, G.K.; *et al.*: Characterization of a murine model of allergic pulmonary inflammation. Int. Arch. Allergy Immunol. 105: 83-90 (1994).
6. Gilmour, M.I.: Interaction of air pollutants and pulmonary allergic responses in experimental animals. Toxicology 105: 335-342 (1995).
7. Gordon, T.; Harkema J.R.: Effects of inhaled endotoxin on intraepithelial mucosubstances in F344 rat nasal and tracheobronchial airways. Am. J. Respir. Cell. Mol. Biol. 10: 177-183 (1994).
8. Stolk, J.; Rudolphus, A.; Davies, P.; *et al.*: Induction of emphysema and bronchial mucus cell hyperplasia by intratracheal instillation od lipopolysaccharide in the hamster. J. Pathol. 167: 349-489 (1992).
9. Shore, S.; Kobzik, L.; Long, N.C.; *et al.*: Increased airway responsiveness to inhaled metacholine in a rat model for chronic bronchitis. Am. J. Respir.Crit. Care Med. 151: 1931-1938 (1995).

Reduction in Vehicle Emissions Attributable to Alternative Transportation Fuels and its Prospective Impact on Air Quality and Public Health

T. C. Coburn, K. J. Kelly, and B. K. Bailey

The National Renewable Energy Laboratory

Gasoline- and diesel-powered automotive emissions are suspected to contribute to, if not cause, the proliferation of a number of human health disorders. However, the evidence from animal studies to date indicates that emissions from vehicles operating on alternative fuels may pose less of a risk. As part of its efforts to reduce dependence on foreign oil and improve overall air quality, the U. S. Department of Energy has undertaken a major study of alternative fuel vehicles (AFVs) which includes a comprehensive emissions testing program. The data collected thus far indicates that, for light-duty vehicles, the levels of regulated emissions from alternative fuel models are lower than those from their conventional fuel counterparts. Emissions of aromatic toxics, as well as ozone forming potential, are also lower for the light-duty alternative fuel vehicles. The reductions in emissions from heavy-duty alternative fuel vehicles are greatest for particulate matter. The combination of these findings with results from recent medical studies suggests that communities having higher concentrations of alternative fuel vehicles may experience lower emissions-related risks of disease and human health disorders, and may ultimately realize reduced costs of medical care, lower individual insurance premiums, and a generally more favorable business climate.

KEY WORDS: Automotive Emissions, Alternative Fuels, Public Health Impacts, Toxicity, Particulate Matter, Air Quality

INTRODUCTION

Emissions from automobiles and other vehicles are known to detrimentally affect urban air quality. These same emissions are suspected to contribute to, if not directly cause, a number of human health disorders. Although findings based on human studies are not yet conclusive, results from animal investigations are more compelling. Marnett,[1] Pennington,[2] Bromberg,[3] Wright,[4] Clarkson,[5] Wood,[6] and Speizer and Ferris[7,8] all discuss the prospective human health impacts of vehicular emissions.

As part of the global effort to improve air quality, engineering and scientific attention has recently focused on investigations of transportation fuels which represent alternatives to conventional gasoline and

diesel, and which have the potential to contribute lower overall levels of pollutants to the atmosphere. Such fuels include methanol, ethanol, natural gas, propane, electricity, and some forms of biodiesel.

The ongoing investigation of alternative transportation fuels also includes a number of studies aimed at evaluating their human health impacts. Much of the early work, particularly with regard to the biological and human health effects of methanol, is summarized by Timourian and Milanovich.[9.] In addition, Böhm, Massad, Saldiva, et. al.[10] compare the toxicity of exhaust fumes from alcohol- and gasoline-fueled automobiles, and Gouveia, Pasqualucci, Saldiva, et. al.[11] address the pathological aspects of ethanol car exhaust emissions. Egebäck and Bertilsson[12] have reported on the chemical and biological characterization of vehicle exhaust emissions associated with several different fuels; and a very recent report of the International Energy Agency[13] also compares the relative health effects of various fuels.

The most recent toxicological research includes the work of Maejima, Suzuki, Numata, et. al.,[14,15,16] who discuss the impacts of methanol-fueled engine exhaust on rats, and that of Massad, Saldiva, Cardoso, et. al.,[17] who relate similar findings concerning the impacts of ethanol-fueled engine exhaust. Acute toxicity of exhaust from vehicles operating on these two alcohol fuels is deemed to be significantly less than acute toxicity of the exhaust from otherwise identical gasoline-fueled vehicles.

For its part, the U. S. Department of Energy (DOE) is heavily promoting the development and deployment of alternative transportation fuels and alternative fuel vehicles (AFVs). This policy is geared to both a reduction in domestic dependence on imported foreign oil and an overall improvement in air quality. Given the results pertaining to alternative fuels reported in the medical literature, a shift in both personal and commercial transportation toward the use of AFVs should result in the additional benefit of fewer emissions-related health disorders. What remains to be demonstrated is that the levels of emissions from AFVs are, in fact, lower than those from conventional vehicles, and that the reductions are large enough

to sustain improvement in public health.

As part of its efforts to promote the use of alternative fuels and AFVs, DOE has initiated an extensive and ongoing evaluation of such vehicles in use in the Federal fleet, including a comprehensive emissions testing program. A primary objective of the study is to definitively determine the levels to which emissions constituents would be reduced as a result of switching to use of alternative transportation fuels.

This program, which targets three alternative fuels--methanol, ethanol, and compressed natural gas--encompasses a number of vehicle makes, models, and model years operating at various sites throughout the United States. It is principally designed to determine the outputs of regulated emissions, air toxics, particulate matter, and ozone forming potential from AFVs relative to those from standard vehicles (controls) operating on conventional transportation fuels. The National Renewable Energy Laboratory manages the program for DOE.

This paper presents some of the early results from DOE's extensive emissions testing program, which are shown to be highly positive in favor of alternative fuels. These test results are linked with recent evidence from the medical literature to formulate hypotheses concerning the prospective impact of alternative fuel usage on public health. Experimental aspects of the testing program are described, and findings to date are summarized. Finally, some questions to be addressed in further research are posed.

EMISSIONS TESTING PROGRAM

DOE's emissions testing program encompasses both light-duty and heavy-duty vehicles. Light-duty vehicles are selected from those in use within the Federal fleet and include sedans and delivery vans. Heavy-duty vehicles are selected from those in commercial use around the country and include transit buses, snow plows, garbage packers, and line haul trucks. These categories encompass the range of vehicles that might be found in a typical U. S. community.

The targeted alternative fuels are M85 (a blend of 85% gasoline and 15% methanol), E85 (a blend of 85% gasoline and 15% ethanol), and CNG (compressed natural gas). Vehicles designed to run on M85 and E85 are flexible-fuel types,

in the sense that they may operate on either the alternative or conventional fuel. Vehicles designed to run on CNG (not aftermarket conversions) are dedicated to that fuel. For light-duty vehicles, the conventional fuel is gasoline; for heavy-duty vehicles it is diesel.

The light-duty emissions testing program is a statistically-designed, multi-year program of experimentation comparing various makes, models, and model years of AFVs to comparable vehicles operating on gasoline (controls). Commercial labs are selected to perform the emissions tests, which conform to the Federal Test Procedures (FTP) prescribed by the U. S. Environmental Protection Agency (EPA).[18] All vehicles are tested in the laboratories under controlled conditions using their targeted fuels. Control vehicles are tested on California Phase 2 reformulated gasoline (RFG).

Regulated emissions data (carbon monoxide, oxides of nitrogen, hydrocarbons) are collected on all vehicles using the FTP. Particulate matter is not collected because the levels in all late-model light-duty vehicles are lower than EPA's original standards, which it no longer enforces. Air toxics (benzene, 1,3-butadiene, formaldehyde, and acetaldehyde) are additionally determined for a subset of the vehicles via gas chromatographic speciation of exhaust gases (cost prohibits speciation of exhaust gases from all vehicles). Ozone forming potential is also calculated from the speciation results using the Carter[19] method. All data are in-use emissions values as opposed to certification values.

Table I shows the make/model combinations of light-duty vehicles tested to date, and the allocation between conventional and alternative fuel types. As of this writing, a total of 148 AFVs have been tested, along with 146 gasoline controls. The selection and arrangement of vehicles in the program facilitates statistical comparisons among fuels, vehicle make/model/year combinations, geographic locations, and testing labs (although not all factors are completely crossed). Details of the testing program, as well as the procedures for vehicle selection and allocation, fuel composition, and various test fleet characteristics, are described more fully in Kelly, Bailey, Coburn, et. al.[20,21,22]

Table I. Types and Numbers of Vehicles Included in the Emissions Testing Program

MAKE/MODEL	CONVENTIONAL FUEL	ALTERNATIVE FUEL
DODGE SPIRIT	70	71
CHEVROLET LUMINA	22	22
FORD ECONOLINE VAN	18	16
DODGE B250 VAN	38	37
TRANSIT BUSES (DDC ENGINES)	17	20
TRANSIT BUSES (CUMMINS ENGINES)	14	21
LINE HAUL TRUCKS	1	4
SNOW PLOWS	1	2
GARBAGE PACKERS	3	6
TOTAL	**184**	**199**

To date, the heavy-duty emissions testing program has involved fewer vehicles, partly due to higher testing costs and vehicle unavailability. The smaller number of heavy-duty vehicles also reflects the smaller population of these vehicles on the road relative to that of light-duty vehicles. Table I lists the types and counts of vehicles tested as of this writing, and shows the allocation between alternative fuel and conventional diesel types for the various vehicle categories. The heavy-duty testing program is also a multi-year endeavor.

For heavy duty vehicles, data is collected on various exhaust constituents, including carbon monoxide, oxides of nitrogen, hydrocarbons, and particulate matter, using procedures developed by West Virginia

gasoline and M85 flexible-fuel Ford Econoline vans). For CNG and gasoline vehicles, hydrocarbons are reported as a non-methane quantity, while for E85 and M85 vehicles they are stated as an organic matter non-methane quantity. Individual test results reported by different labs and for different geographic locations have been pooled. Table II lists the EPA emissions standards to which light-duty vehicles are required to conform in model year 1994.

In Figure 1, three of four alternative fuel models--the CNG vans, E85 sedans, and M85 vans--are all shown to emit lower levels of CO, on average, than their gasoline counterparts. For the CNG vans, the average value of CO is 2.54 g/mi versus 4.81 g/mi for the gasoline controls; for the E85 sedans, the average value of CO is 2.29 g/mi versus 3.39 g/mi for the gasoline controls; and for the M85 vans, the average value of CO is 1.49 g/mi versus 3.22 g/mi for the gasoline controls. The reductions in the average values of CO for these alternative fuel models compared to the gasoline controls are on the order of 30% or more. The average value of CO for the M85 sedans--the fourth alternative fuel model--is slightly higher than the corresponding average for the gasoline controls (1.68 g/mi versus 1.48 g/mi, for an increase of 12.0%). All average values of CO for both gasoline and alternative fuel models are within the respective EPA standards.

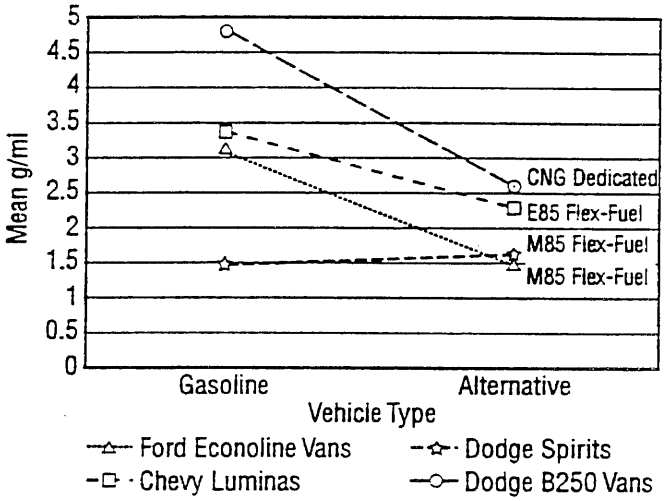


Figure 1. Comparison of CO emissions from four light-duty AFVs and their gasoline-fueled counterparts.

In Figure 2, all four alternative fuel models are shown to emit lower levels of NO_X, on average, than their gasoline counterparts. For the CNG vans, the average value of NO_X is 0.53 g/mi versus 0.74 g/mi for the gasoline controls; for the E85 sedans, the average value of NO_X is 0.17 g/m versus 0.58 for the gasoline controls; for the M85 sedans, the average value of NO_X is 0.19 g/mi versus 0.32 g/mi for the gasoline controls; and for the M85 vans, the average value of NO_X is 0.84

g/mi versus 0.87 g/mi for the gasoline controls. Except for the M85 vans, the reductions in average values of NO_X for the alternative fuel models compared to the corresponding gasoline controls are on the order of 30% or more. Except for the E85 gasoline controls, the average values of NO_X for all models are within the respective EPA standards.

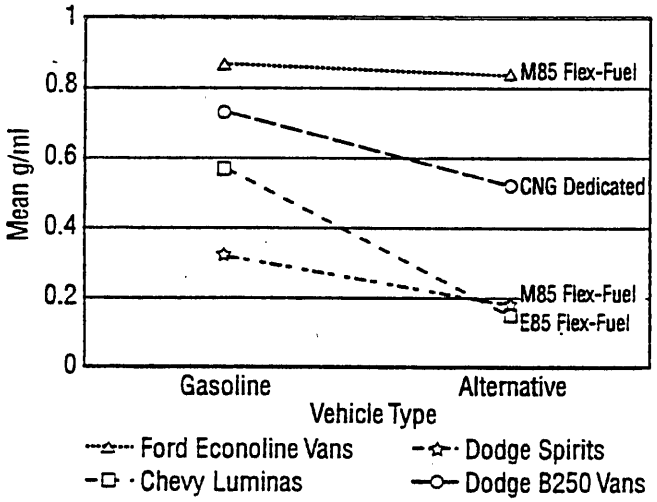


Figure 2. Comparison of NO_X emissions from four light-duty AFVs and their gasoline-fueled counterparts.

The results for hydrocarbons are similar to those for CO. Figure 3 indicates that CNG vans, E85 sedans, and M85 vans all emit lower levels of hydrocarbons, on average, than their gasoline counterparts. For the CNG vans, the average value of HC is 0.05 g/mi versus 0.28 g/mi for the gasoline controls; for the E85 sedans, the average value of HC is 0.11 g/mi versus 0.18 for the gasoline controls; and for the M85 vans, the average value of HC is 0.14 g/mi versus 0.27 g/mi. The reductions in the average values of HC for these alternative fuel models compared to the gasoline controls are on the order of 40% or more.

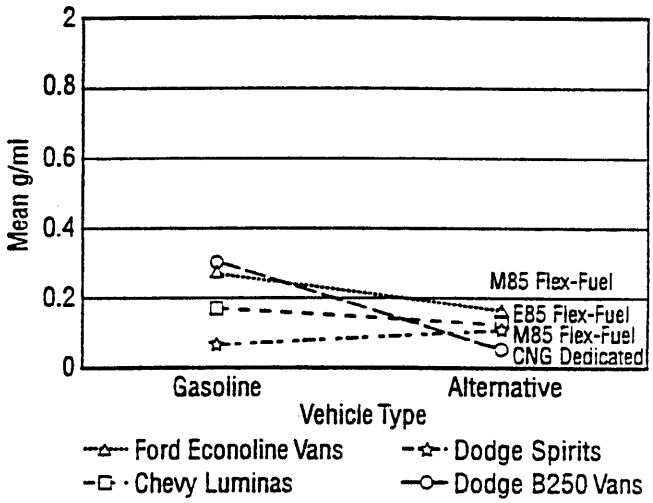


Figure 3. Comparison of HC emissions from four light-duty AFVs and their gasoline-fueled counterparts.

The average value of HC for the M85 sedans is slightly higher than the corresponding average for the gasoline controls (0.10 g/mi versus 0.08 g/mi). All average values of HC for both gasoline and alternative fuel models are within the respective EPA standards.

Air Toxics from Light-Duty Vehicles

Figures 4-7 show comparative average emissions of four light-duty vehicle exhaust constituents determined by EPA to be toxic emissions: benzene (C_6H_6), 1,3-butadiene (C_4H_6), formaldehyde (HCHO), and acetaldehyde (CH_3CHO). The average values reported in these figures summarize data obtained on a subset of the vehicles for which regulated exhaust emissions are reported in Figures 1-3. Table III lists the numbers and types of vehicles for which toxic emissions are reported. There are no EPA vehicular standards for these quantities.

Table III. The Numbers and Types of Light-Duty Vehicles for Which Air Toxics and Ozone Forming Potential are Reported.

MAKE/MODEL	CONVENTIONAL FUEL	ALTERNATIVE FUEL
DODGE SPIRIT	6	6
CHEVROLET LUMINA	1	2
FORD ECONOLINE VAN	2	2
DODGE B250 VAN	3	2
TOTAL	**12**	**12**

As expected, the average values of the aromatics--benzene and 1,3-butadiene--are lower for all the alternative fuel models in comparison to their respective gasoline counterparts. The size of the reduction typically depends on the fuel in question, and to a different degree, the vehicle technology.

In the case of benzene (Figure 4), the greatest reductions are observed for the CNG and M85 vans--the CNG vans exhibit an average value of 0.70 mg/mi versus 10.30 mg/mi for the gasoline controls, and the M85 vans exhibit an average value of 1.70 mg/mi versus 7.80 mg/mi for the gasoline controls. These decreases are on the order of 75% or more. For the sedans,

the E85 models exhibit an average value of benzene of 2.48 mg/mi versus 3.74 mg/mi for the gasoline controls, and the M85 models exhibit an average value of 1.39 mg/mi versus and average value of 2.15 mg/mi for the gasoline controls (decreases on the order of 35% or more).

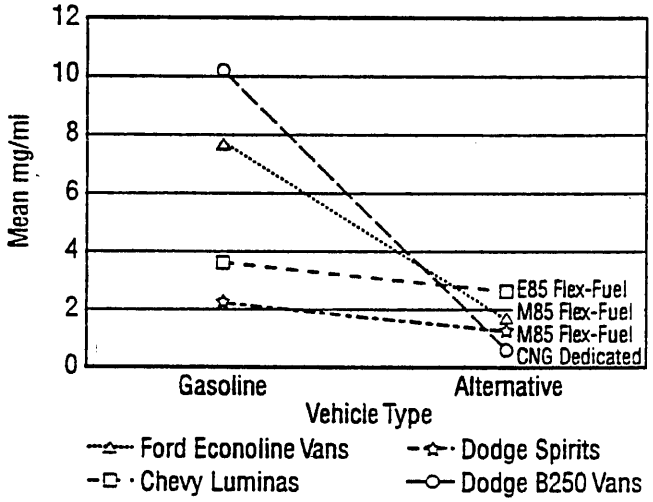


Figure 4. Comparison of C_6H_6 emissions from four light-duty AFVs and their gasoline-fueled counterparts.

For 1,3-butadiene (Figure 5), the greatest reductions are observed for the CNG vans and the E85 sedans--the CNG vans exhibit an average value of 0.10 mg/mi versus an average of 1.93 mg/mi for the gasoline controls, and the E85 sedans exhibit an average value of 0.17 mg/mi versus an average of 1.20 mg/mi for the gasoline controls. The decreases in average values of butadiene attributable to the alternative fuel models are on the order of 85% or more. The two M85 models also exhibit lower average values of 1,3-butadiene than their gasoline counterparts, although the reduction is not as large. The average value for the M85 vans is 0.10 mg/mi versus 0.40 mg/mi for the gasoline controls; and the average value for the M85 sedans is 0.10 mg/mi versus 0.30 mg/mi for the gasoline controls--reductions which are still more than 65%.

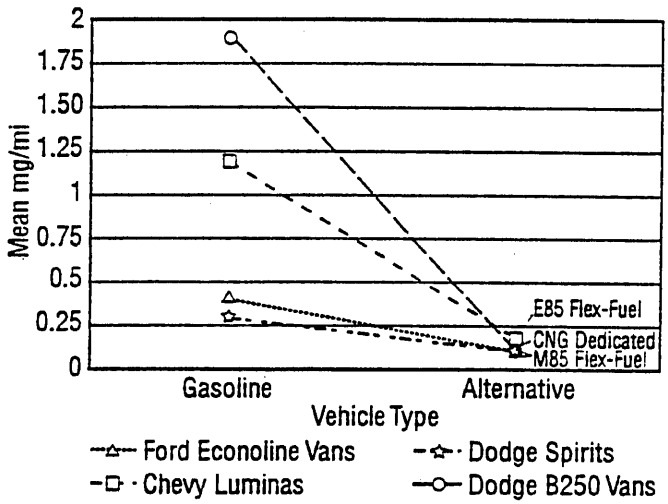


Figure 5. Comparsion of C_4H_6 emissions from four light-duty AFVs and their gasoline-fueled counterparts.

Also as expected, the aldehyde levels exhibited in exhaust emissions from alternative fuels tend to be higher, depending on the fuel and

vehicle technology in question. The elevated levels of aldehydes attributable to alternative fuels is the subject of continued research in both the automotive and medical communities.

Figure 6 indicates that all alternative fuel models in this study exhibit higher average levels of formaldehyde than their respective gasoline counterparts, the largest increases (more than 75%) being associated with the M85 vehicles.

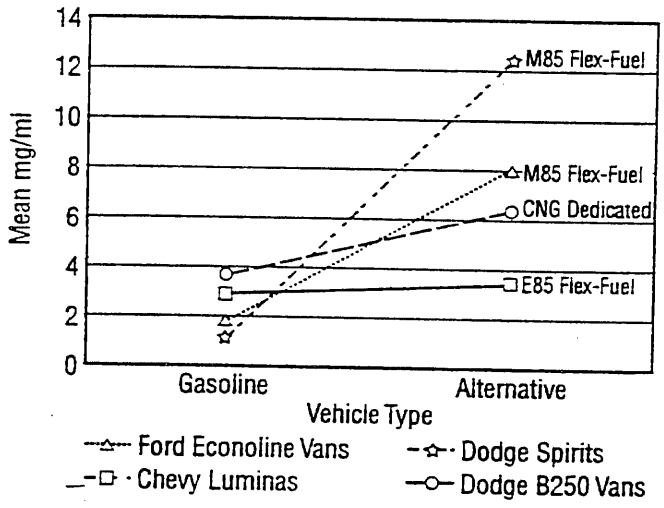


Figure 6. Comparison of HCHO emissions from four light-duty AFVs and their gasoline-fueled counterparts.

In the case of acetaldehyde (Figure 7), the results are considerably different. The CNG vans, the M85 vans, and the M85 sedans all exhibit decreases from the already minimal levels exhibited by their gasoline counterparts (0.39 mg/mi versus 1.02 mg/mi, and 0.15 mg/mi versus 0.63 mg/mi, and 0.25 mg/mi versus 0.30 mg/mi, respectively). For the E85 sedans, an opposite result is observed. The average value of acetaldehyde for this model is 17.21 mg/mi, whereas the corresponding value for the gasoline controls is only 0.93 mg/mi--a difference of approximately 95%.

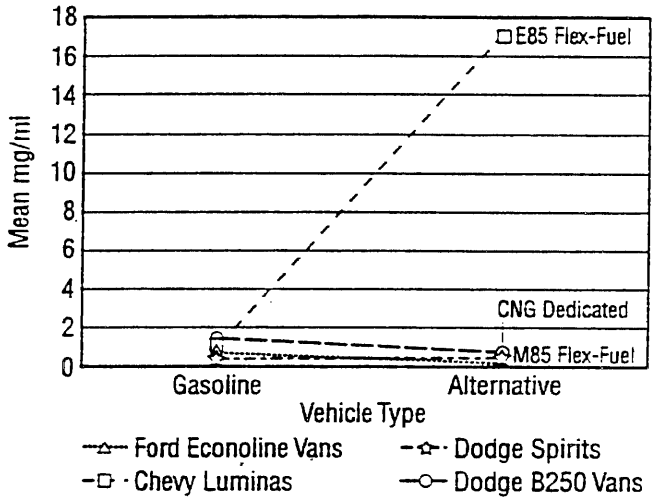


Figure 7. Comparison of CH_3CHO emissions from four light-duty AFVs and their gasoline-fueled counterparts.

These results for air toxics parallel those reported by Black and Gabele.[25]

Ozone Forming Potential for Light-Duty Vehicles

Ozone forming potential (OFP) is reported for the same subset of light-duty vehicles for which air toxics have been determined through speciation of exhaust gases (Figures 4-7). As previously noted, the procedure for calculating OFP is provided by Carter[19].

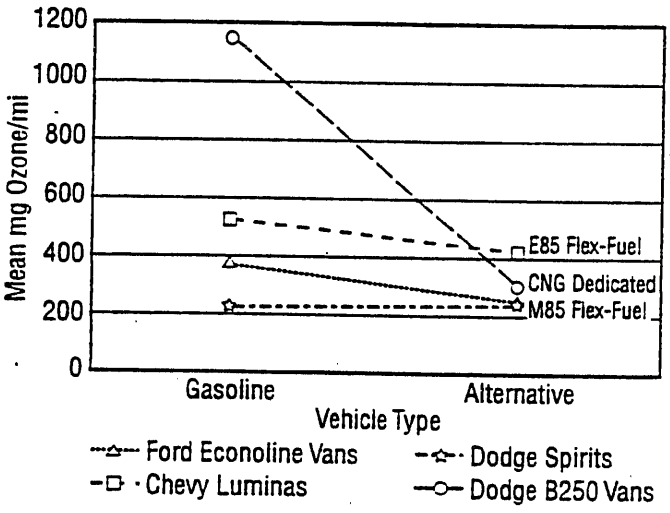


Figure 8. Comparison of OFP from four light-duty AFVs and their gasoline-fueled counterparts.

This quantity, which involves the relationship between volatile organic compounds and non-methane organic gases found in vehicular exhaust, is stated in terms of milligrams of ozone per mile (mg ozone/mi). While there is no EPA standard for OFP, it specifically gauges the potential of vehicles to contribute ozone, a major factor in the formation of urban smog, to the atmosphere.The respective average values of OFP for the four models of AFVs and their corresponding gasoline counterparts are compared in Figure 8. Average OFP for three of the four alternative fuel models (CNG vans, E85 sedans, and M85 vans) is lower than for the corresponding controls. For CNG vans, average OFP is 294.05 mg ozone/mi, while it is 1149.41 mg ozone/mi for the controls; for E85 sedans, average OFP is 415.53 mg ozone/mi, while it is 484.41 mg ozone/mi for the controls; and for M85 vans, average OFP is 265.70 mg ozone/mi versus 388.00 mg ozone/mi for the controls. The average for the fourth alternative fuel model, the M85 sedans, is slightly higher (260.00 mg ozone/mi) than the average for the corresponding controls (211.00 mg ozone/mi), although though it is still low enough to be essentially equal to the average for the M85 vans.

Emissions from Heavy-Duty Vehicles

Heavy-duty engine technology to accommodate the use of alternative fuels is not nearly so advanced as light-duty vehicle technology. Nonetheless, engine certification data obtained and reported by manufacturers indicates a

strong potential for reductions in emissions from heavy-duty alternative fuel vehicles. Certification standards promulgated by EPA are evolving, and are focusing on reducing particulate matter (PM) without affecting NO_X (typically, a reduction in PM will be accompanied by an increase in NO_X). Capabilities to test and measure in-use emissions are also still developing, although in-use emissions standards are not yet available. Early results from DOE's testing program, as depicted in Figures 9-13 below, indicate that substantial reductions in PM can be achieved, but that the levels of some other exhaust constituents are not yet as low as desired (individual test results representing different geographic locations have been pooled). The ability to simultaneously achieve lower levels of all these emissions constituents is an active area of heavy-duty engine research.

Transit Buses

Figures 9-12 compare the average emissions of CO, NO_X, HC, and PM, respectively, for three models of alternative fuel buses (CNG, E85, and M85) and their corresponding diesel controls. In Figure 9, the average value of CO for the CNG buses is lower than for their diesel counterparts (1.29 g/mi versus 16.87 g/mi). The average value of CO for the M85 buses (16.31 g/mi) is about the same (though slightly higher) as it is for their diesel counterparts (16.02 g/mi). For the E85 buses, the average value of CO (39.60 g/mi) is considerably higher than the average for their diesel counterparts (7.87 g/mi).

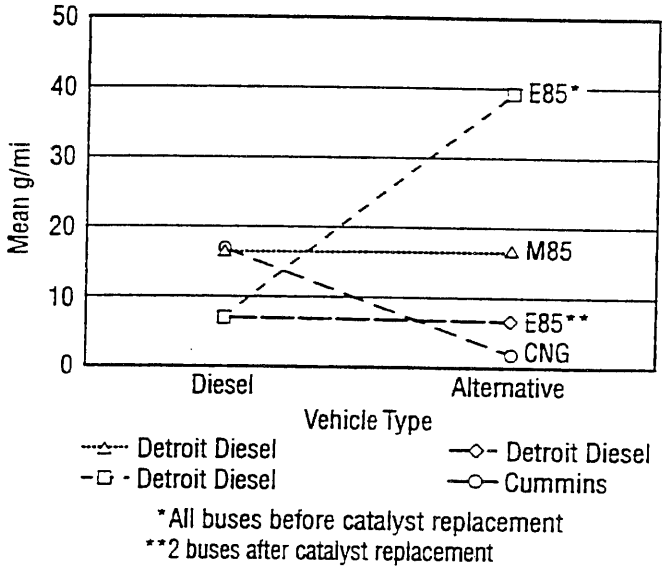


Figure 9. Comparison of CO emissions from three alternative fuel transit buses and their diesel-fueled counterparts.

Upon removing the values associated with E85 buses found to have faulty catalytic converters, the average value decreased to 7.25 g/mi, a level essentially equal (though slightly lower) to the average for the diesel controls. This finding underscores the

necessity of careful maintenance in the quest to control heavy-duty vehicle emissions.

Figure 10 shows the average values of NO_X for transit buses. All three models of alternative fuel transit buses have lower average values of NO_X than their diesel counterparts. For the CNG buses, the average value of NO_X is 11.75 g/mi, whereas for their diesel counterparts it is 24.21 g/mi; for the E85 buses, the average value of NO_X is 17.70 g/mi, whereas for their diesel counterparts it is 25.74 g/mi; and for the M85 buses, the average value of NO_X is 9.11 g/mi, whereas it is 26.67 g/mi for their diesel counterparts.

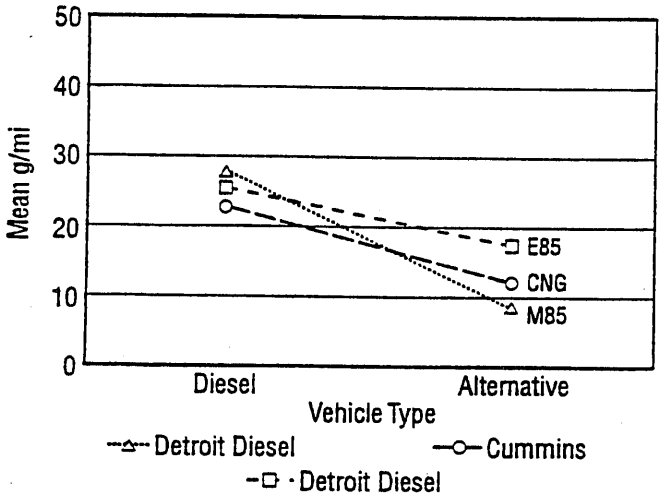


Figure 10. Comparison of NO_X emissions from three alternative fuel transit buses and their diesel-fueled counterparts.

Average values of HC for transit buses are compared in Figure 11. The average values for all three alternative fuel models are higher than those of their diesel counterparts. While this result is somewhat discouraging, it is believed that careful tracking of vehicle performance and maintenance will result in substantial improvement.

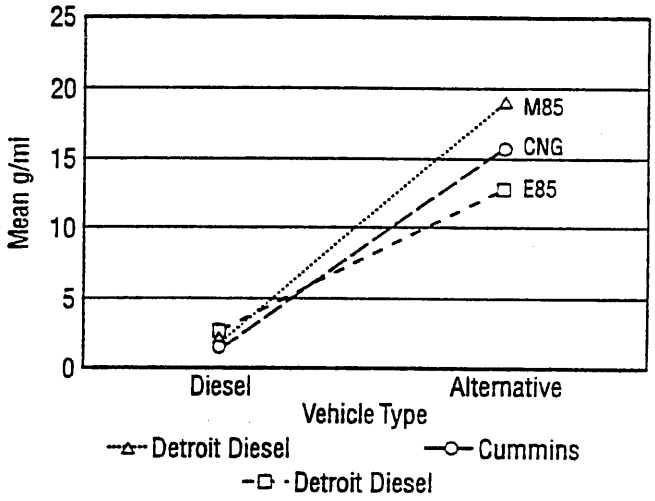


Figure 11. Comparison of HC emissions from three alternative fuel transit buses and their diesel-fueled counterparts.

Figure 12 compares the average values of PM (all values are for PM10) for the three models of alternative fuel transit buses and the corresponding diesel controls. In all three cases, considerably lower average values of PM were obtained for the alternative fuel models. For

the CNG buses, the average value of PM is 0.03 g/mi, whereas it is 1.76 g/mi for their diesel counterparts; for the E85 buses, the average value of PM is 0.56 g/mi, whereas it is 0.92 g/mi for their diesel counterparts; and for the M85 buses, the average value of PM is 0.26 g/mi, whereas it is 2.53 g/mi for their diesel counterparts. The average values for the CNG and M85 models represent reductions of 90% or more.

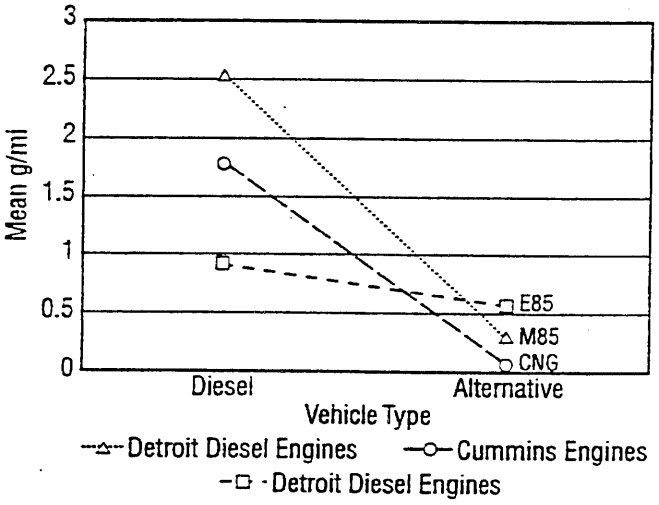


Figure 12. Comparison of PM emissions from three alternative fuel transit buses and their diesel-fueled counterparts.

Other Heavy-Duty Vehicles

For other heavy-duty vehicles in DOE's emissions testing program, the most promising improvement to date in emissions output also pertains to PM. Figure 13 compares the average values of PM (all values reported as PM10) for three heavy-duty alternative fuel vehicles and their diesel counterparts: CNG garbage packers, E85 snow plows, E85 line-haul trucks. As noted above (Table I), the total numbers of vehicles of each type is relatively small.

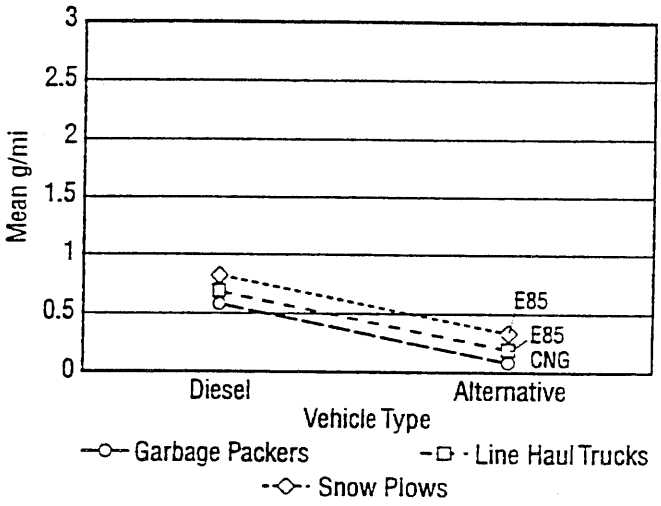


Figure 13. Comparison of PM emissions from three heavy-duty AFVs and their diesel-fueled counterparts.

For all three types of vehicles, the alternative fuel models exhibit lower average values of PM than the corresponding diesel controls. For the CNG garbage packers, the average value of PM is 0.02 g/mi, whereas for their diesel counterparts it is 0.66 g/mi; for the E85 snow plows, the average value of PM is 0.37

g/mi, whereas it is 0.81 g/mi for their diesel counterparts; and for the E85 line-haul trucks, the average value of PM is 0.28 g/mi, whereas it is 0.76 g/mi for their diesel counterparts. The reductions in average PM for alternative fuel vehicles are all least 50%.

SUMMARY OF FINDINGS

For light-duty vehicles, which include sedans and delivery vans, most alternative fuel models tested to date have exhaust emissions levels which are consistently lower for all EPA-regulated constituents (CO, NO_X, HC), on average, than those exhibited by otherwise identical gasoline models. The average levels of CO, NO_X, and HC for most models of both gasoline vehicles and AFVs are below EPA's Tier 1 standards, although the average values exhibited by AFVs indicate they could meet even lower criteria. PM is not reported for light-duty vehicles because the emitted levels in all new units are so low that EPA no longer requires manufacturers to meet specific requirements (this does not preclude the possibility that older model vehicles still in use are high emitters of PM).

Regarding air toxics, the performance of light-duty AFVs relative to gasoline controls depends on the alternative fuel in question. The aromatics (benzene and 1,3-butadiene) are uniformly lower, on average, for all the AFV models. On the other hand, the average levels of aldehydes (formaldehyde and acetaldehyde) are higher in some cases. Such increases are not unexpected, particularly for the alcohol-based fuels (E85 and M85). Ozone forming potential is lower, on average, for almost all models of AFVs tested to date when compared to otherwise identical gasoline models.

For heavy-duty vehicles, the most promising results pertain to particulate matter, with alternative fuel transit buses, as well as alternative fuel models of other types of vehicles, exhibiting consistently lower average values than their diesel counterparts. The average levels of other pollutants emitted by alternative fuel heavy-duty vehicles are not yet as low as desired, and in some cases, they are higher than those of their diesel counterparts. So far, alternative fuels are showing the greatest potential for overall emissions improvement in transit buses (for example, the average values of NO_X reported here for alternative fuel models are all lower than those reported for the corresponding diesel controls). Continued

engine research development, and further emissions testing, should result in additional notable improvements.

RESEARCH CONCLUSIONS AND PUBLIC HEALTH IMPLICATIONS

The DOE emissions testing program, the most comprehensive undertaking of its kind regarding alternative fuels, has established a statistical baseline from which projections can be made concerning further reductions in atmospheric pollutants attributable to the transportation sector. As government policies pertaining to urban air quality evolve, and as additional medical evidence concerning the public health impacts of vehicular emissions accumulates, this collection of information will provide an important reference for future work. Some important conclusions can already be drawn.

The testing conducted to date on light-duty vehicles yields considerable evidence that alternative fuel models do indeed have improved overall emissions profiles (regulated exhaust emissions, air toxics, and ozone forming potential) compared to those of otherwise identical gasoline vehicles. While this finding corroborates the results from a number of other studies, it is particularly persuasive due to the extent of the data set.

Heavy-duty vehicles are also showing promise that alternative fuel models can achieve lower average emissions than otherwise identical diesel models. The mass of emissions from heavy-duty vehicles is much greater than from light-duty vehicles, which presents great opportunity to achieve large percentage reductions.

Nonetheless, the sheer numbers of light-duty vehicles in service compared to the numbers of heavy-duty vehicles accentuates the importance of the improvements already achieved in the light-duty arena. In the near term, the widespread proliferation of alternative fuel sedans, delivery vans, and pickups could have the largest positive impact on air quality and public health.

The results for light-duty AFVs reported here strongly complement the findings of the most recent medical studies concerning alternative fuels. While not totally eliminated, the evidence solidly suggests that the risk of emissions-related disease and health disorders in humans is lower as a result of using alternative transportation fuels than it is with using conventional gasoline.

The public health consequences of particulate matter are potentially quite serious, and the mechanism(s) by which it pervades the human body is the subject of considerable medical research. Unfortunately, for the reasons mentioned above, no particulate matter data (neither PM10 nor PM2.5) is reported here for light-duty vehicles. It is assumed that the PM10 levels are within acceptable limits for all vehicle types, although the human tolerance levels to both PM10 and PM2.5 are still be open to debate.

For heavy-duty vehicles, however, PM10 measurements have been collected as part of the present study and are reported here, yielding important insight into the overall performance of vehicles operating on alternative fuels relative to conventional fuel controls. For transit buses, as well as other heavy-duty vehicle types, the average values of particulate matter are lower for all alternative fuel models than for their diesel counterparts, and in some instances the reductions are substantial. One obvious benefit of this finding is that the visible range of urban air pollution could be substantially mitigated. Additionally, however, the evidence again strongly suggests that widespread use of alternative transportation fuels can have a positive impact in reducing the risk of emissions-related health problems.

It is apparent from this study that DOE's policies of promoting the development of alternative fuels and of stimulating markets for alternative fuel vehicles can significantly contribute to the national goal of improved air quality, particularly if use of such fuels and vehicles is fairly widespread. It can be further suggested that communities having high concentrations of alternative fuel vehicles would record fewer emissions-related health problems; and that this situation, in turn, would translate into reduced costs of medical care, lower health insurance premiums for individuals, and a generally more favorable business climate.

The extent to which these issues can be positively linked and quantified is an important subject for continued research. Where possible, epidemiological and engineering studies should be undertaken to establish the direct relationship between AFV usage and the incidence of various public health outcomes. In addition, further emissions testing, along with continued development of engine technology, should be pursued.

ACKNOWLEDGMENTS

This work was sponsored by the Office of Transportation Technologies of the U. S. Department of Energy in Washington, DC. John Garbak and Frank Mallgrave were the technical monitors.

REFERENCES

1. Marnett, Lawrence J.: Health Effects of Aldehydes and Alcohols in Mobile Source Emissions. In: Air Pollution, The Automobile, and Public Health, pp. 579-603. Ann Y. Watson, Richard R. Bates, and Donald Kennedy, Eds. National Academy Press, Washington, DC (1988)

2. Pennington, James E: Effects of Automotive Emissions on Susceptibility to Respiratory Infections. In: Air Pollution, The Automobile, and Public Health, pp. 499-518. Ann Y. Watson, Richard R. Bates, and Donald Kennedy, Eds. National Academy Press, Washington, DC (1988).

3. Bromberg, Philip A.: Asthma and Automotive Emissions. In: Air Pollution, The Automobile, and Public Health, pp. 465-497. Ann Y. Watson, Richard R. Bates, and Donald Kennedy, Eds. National Academy Press, Washington, DC (1988).

4. Wright, Joanne L.: Relation of Pulmonary Emphysema and Small Airways Disease to Vehicular Emissions. In: Air Pollution, The Automobile, and Public Health, pp. 441-463. Ann Y. Watson, Richard R. Bates, and Donald Kennedy, Eds. National Academy Press, Washington, DC (1988).

5. Clarkson, Thomas B.: Evaluation of Automotive Emissions as Risk Factors for the Development of Atherosclerosis and Coronary Heart Disease. In: Air Pollution, The Automobile, and Public Health, pp. 605-629. Ann Y. Watson, Richard R. Bates, and Donald Kennedy, Eds. National Academy Press, Washington, DC (1988).

6. Wood, Ronald W.: Identifying Neurobehavioral Effects of Automotive Emissions and Fuel Components. In: Air Pollution, The Automobile, and Public Health, pp. 631-657. Ann Y. Watson, Richard R. Bates, and Donald Kennedy, Eds. National Academy Press, Washington, DC (1988).

7. Speizer, F. E.; Ferris, Jr., B. G.: Exposure to Automobile Exhaust. I. Prevalence of Respiratory Symptoms and Disease. Arch. Environ.

Health 26:pp. 313-318 (1973).

8. Speizer, F. E.; Ferris, Jr., B. G.: Exposure to Automobile Exhaust. II. Pulmonary Function Measurement. Arch. Environ. Health 26: pp. 319-324 (1973).

9. Timourian, H.; Milanovich, F.: Methanol as a Transportation Fuel: Assessment of Environmental and Health Research. Report UCRL-52697, Lawrence Livermore National Laboratory, pp. 35-60. Livermore, CA (1979).

10. Böhm, G. M.; Massad, E.; Saldiva, P. H.; et al.: Comparative Toxicity of Alcohol and Gasoline Fueled Automobile Exhaust Fumes. In: Development in Science and Practice of Toxicology, pp. 479-482. A. N. Nayes, R. C. Schnell, and T. S. Miya, Eds. Elsevier, Amsterdam (1983).

11. Gouveia, M. A.; Pasqualucci, C. A. G.; Saldiva, P. H.; et al.: Pathological Aspects of Intoxication by Ethanol Car Exhaust Emission. J. Pathol. 142(A9) (1984).

12. Egebäck, K. E.; Bertilsson, B. M.: Chemical and Biological Characterization of Exhaust Emissions from Vehicles Fueled with Gasoline, Alcohol, LPG and Diesel. Statens naturvårdsverk (SNV PM 1635). Solna (1983).

13. International Energy Agency: Comparison of Relative Environmental Impacts of Alternative and Conventional Motor Fuels--Final report, Annex VII. IEA Alternative Motor Fuels Agreement, pp. 99-108. Paris (1995).

14. Maejima, K.; Suzuki, T; Numata, H.; et al.: Toxicity to Rats of methanol-Fueled Engine Exhaust Inhaled Continuously for 28 Days. J. Toxicol. Environ. Health 37:pp. 293-312 (1992).

15. Maejima, K.; Suzuki, T.; Numata, H.; et al.: Recovery from Changes in the Blood and Nasal Cavity and/or Lungs of Rats Caused by Exposure to Methanol-Fueled Engine Exhaust. J. Toxicol. Environ. Health 39:pp. 323-340 (1993).

16. Maejima, K; Suzuki, T; Numata, H; et al.: Subchronic Inhalation Toxicity Study of Methanol-Fueled Engine Exhaust in Rats. J. Toxicol. Environ. Health 41(3):pp. 315-327 (1994).

17. Massad, Eduardo; Saldiva, Carmen Diva; Cardoso, Luiza

Maria Nunes; et al.: Acute Toxicity of Gasoline and Ethanol Automobile Engine Exhaust Gases. Toxicol. Lett. 26(2-3):pp. 187-192 (1985).

18. Federal Register: Standards for Emissions from Natural Gas-Fueled, and Liquefied Petroleum Gas-Fueled Motor Vehicles and Motor Vehicle Engines, and Certification Procedures for After-Market Conversions. 40 CFR Parts 80, 85,86,88, and 600. Final Rule:59(182). Washington, DC (September 21, 1994).

19. Carter, William P. L.: Development of Ozone Reactivity Scales for Volatile Organic Compounds. Journal of the Air Waste Management Association 44:pp. 770-777 (1985).

20. Kelly, Kenneth J.; Bailey, Brent K.; Coburn, Timothy C.; et al.: FTP Emissions Test Results from Flexible-Fuel Methanol Dodge Spirits and Ford Econoline Vans. SAE Technical Paper Series, 961090. Society of Automotive Engineers, Warrendale, PA (1996).

21. Kelly, Kenneth J.; Bailey, Brent K.; Coburn, Timothy C.; et al.: Round 1 Emissions Test Results from Compressed Natural Gas Vans and Gasoline Controls Operating in the U. S. Federal Fleet. SAE Technical Paper Series, 961091. Society of Automotive Engineers, Warrendale, PA (1996).

22. Kelly, Kenneth J.; Bailey, Brent K.; Coburn, Timothy C.; et al.: Federal Test Procedure Emissions Test Results from Ethanol Variable-Fuel Vehicle Chevrolet Luminas. SAE Technical Paper Series, 961092. Society of Automotive Engineers, Warrendale, PA (1996).

23. Bata, R.; Clark, N.; Lyons, D.; et al.: A Transportable Heavy Duty Vehicle Emissions Testing Laboratory: A New Dimension in Vehicle Testing. ASME Technical Paper 92-ICE-21. The American Society of Mechanical Engineers, New York, NY (1992).

24. U. S. Department of Energy: Federal Alternative Motor Fuels Programs, Fifth Annual Report to Congress. USDOE, Office of Energy Efficiency, Office of Transportation Technologies, Washington, DC (1996).

25. Black, F.; Gabele, P.: Environmental Protection Agency: Impact of Methanol and CNG Fuels on Motor Vehicle Toxic Emissions. EPA Report No. EPA/600/D-91/240. EPA. Research Triangle Park, NC (1991).