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INTRODUCTION

A variety of potential effects on human health resulting from climate change have been identified in several assessments. According to an international panel[1] they include direct effects of extreme temperatures on cardiovascular deaths, secondary effects due to vector-borne diseases or crop yields, and tertiary effects such as those that might arise from conflicts over freshwater supplies. To this list we add the secondary effects of increased air pollution, which may result either directly from climate change or indirectly from increased air conditioning loads and the corresponding pollutant emissions from electric utilities. Higher ozone concentrations have been linked to increased ambient temperatures by both theory and observations of monitoring data. A similar association with particulate matter has been limited to observations, thus far. The

pollution-heat linkage has been recognized before[2] but health effects have not been evaluated in terms of predictions of the joint effects of both agents.

This paper has been prepared in two sections. First, we discuss the ozone situation with special reference to the Northeast Corridor and New York. In the second section, we present estimates of the health effects of climate change on New York and discuss some mitigation options.

AIR QUALITY

New York City, like most large cities, has an air pollution problem. We tend to notice this problem most in the hazy, hot, and humid days of July and August. Haze can reduce atmospheric visibility from a value greater than 50 km on a crisp cold day to 5 km or less. Although this haze has some natural components, it consists mainly of pollution-derived aerosol particles. This visible reminder of air pollution on hot days is one of the causes for the concern that air pollution will become much worse if temperatures rise as has been predicted in global warming scenarios.

Aerosol particles are not the only pollutant that occurs in hazy air. Dirty air tends to have high levels of many different pollutants and haze is often a visible surrogate for otherwise invisible pollutants which we can sense by smell, taste, or their effect on our eyes or breathing. A major concern is with ozone (O_3) and other ingredients of photochemical smog which may likewise increase in concentration on a warmer planet. Our concerns are magnified by the robust statistical

link between O_3 and temperature as revealed in monitoring data. A portion of this relation does appear to be cause and effect, whereby higher temperatures accelerate the rates of the chemical reactions generating photochemical smog. This, however, is not the whole story as both temperature and pollutant levels respond in common to changes in meteorological conditions such as increased sunshine. Distinguishing between the direct effects of temperature and the coincidental effects is critical to predicting the future.

This section will discuss the possible effects of a warmer climate on the air pollution problem in New York City focusing on ozone. We choose this focus for several reasons: Of the air pollutants regulated by the Environmental Protection Agency (EPA) (O_3, sulfur dioxide, nitrogen dioxide, carbon monoxide, lead, and particulate matter), the control of O_3 is the most problematic as judged from the limited success achieved from three decades of effort to lower O_3 levels. Significant adverse health effects have been attributed to O_3 at concentration levels common in urban atmospheres. Ozone is a component of photochemical smog, a combination of air pollutants that are formed in the atmosphere from chemical reactions involving nitrogen oxides (NO_x) and volatile organic compounds (VOCs, the principal components being various hydrocarbons). Most attention is directed towards ozone because of its impacts on respiratory function. However, many other ingredients of smog are eye irritants or noxious in other ways. Our understanding of the chemical and meteorological factors affecting the formation of O_3 and our past experience suggest that meeting the current Federal standards will be very difficult, possibly requiring extreme emission control measures. Global warming is expected to exacerbate this situation.

In the following sections we describe the current O_3 pollution problem, the chemical and meteorological factors that are controlling O_3 levels, and ways in which these controlling factors might change due to global warming. We use a combination of monitoring data and model calculations to estimate the direct effect of a greenhouse-caused temperature perturbation on O_3 levels. Other climate variables will also change and have their impacts on O_3 levels, but we regard these non-temperature effects as being beyond our forecast capabilities. The sensitivity of O_3 to temperature, so derived, is used in a following section to estimate increased mortality due to increased O_3 from greenhouse warming.

Aerosol particles are also of concern because they are major components of acid rain, they are responsible for visibility degradation, and they have adverse health impacts. As with O_3, there is a statistical link between particle concentration and temperature. Although it is plausible that higher temperatures accelerate the rates of atmospheric reactions leading to aerosol products, this effect has not been quantified. Current evidence suggests that day to day variations in particle concentration are driven mainly by ventilation factors such as wind speed and wind direction relative to the location of emission sources. How these factors will change in a warmer climate cannot be predicted confidently. Accordingly we will not attempt to estimate the perturbation to aerosols occurring in a warmer climate. Instead, for comparison purposes, an estimate for the change in aerosol health effects will be derived based on the current relation between aerosols and temperature, realizing that this relation is likely a surrogate for the effects of other meteorological variables on particle concentration.

The Ozone Problem

The Federal Clean Air Act amendments of 1970, 1977, and 1990 classify O_3 as a criteria pollutant and specify the control of O_3 based on its acute respiratory effects to a susceptible population that includes the elderly, the very young, and asthmatics. The current federal standard requires that the fourth highest hourly average O_3 concentration recorded in a three-year period shall not exceed 120 ppb. Ozone also has deleterious effects at lower concentration including reductions in crop yields and damage to forests. Additional standards to protect against long term exposure to moderate O_3 levels have been discussed but not implemented.

The EPA and corresponding agencies at the state level have been monitoring O_3 levels for more than three decades. In recent years (1982 -1989) an average of 84 metropolitan areas, containing approximately half the population of the United States, reported O_3 levels in excess of the federal standard. Hourly concentrations close to 500 ppb have been reported in Los Angeles (1980) and 300 ppb in the New York area (1987)[3].

With three decades of progressively more stringent emission controls there has been a modest improvement in the urban O_3 problem, but O_3 levels in most large cities still violate the 120 ppb standard. Part of the reason that O_3 has not been susceptible to control measures is that over the same three decades there have been significant increases in population, industrial activity, and, especially, automobile traffic. Trends are difficult to determine because of year to year variability in meteorological conditions and the associated variability in O_3. Between 1980 and

1989, the maximum 1-hour O_3 concentration in urban areas decreased by 1.4% per year[4]. Ozone levels in 1988 were very high because of a particularly hot and dry summer over much of the United States; a trend based on the decade previous actually indicates an increase in O_3[5]. Exceedence of the O_3 standard in New York State for the period 1973 to 1992 are indicated in Figure 1. Most of the standard violations occur in the New York metropolitan area. Both the number of violations and their severity have been decreasing.

In recent years concern about high O_3 levels has been extended to non-urban areas even though these areas usually do not violate the 120 ppb standard. Based on a combination of routine monitoring, intensive field experiments, and model calculations, our current picture of photochemical smog is that it is often a regional scale phenomena. Under optimum conditions for O_3 production, concentrations higher than 80 ppb can occur over much of the eastern United States. Embedded within a regional event are hot spots corresponding to urban areas containing large NO_x and hydrocarbon emission sources. The northeast corridor, encompassing the cities of Washington, DC, Baltimore, Philadelphia, New York, and Boston, can have O_3 levels approaching and above the Federal standard over its whole length. Ozone levels are usually at their highest 50 or more km downwind of major source regions. Thus, the highest O_3 recorded during 1994 in the Northeastern United States was in Stratford, Connecticut (180 ppb). Ozone has a multi-day lifetime in the atmosphere and can thereby be transported significant distances. The plume from the northeast corridor is responsible for O_3 concentrations greater than 175 ppb observed at Acadia National Park, Maine. Further to the east, the plume has been observed over Yarmouth, Nova Scotia causing an O_3 concentration in excess of 130 ppb[6]. With further

eastward transport, the North American pollutant plume becomes part of the Northern hemisphere background. Man-made pollutants are thought to be responsible for more than half of the O_3 in the Northern hemisphere, even in remote regions[7,8].

The regional O_3 background in the eastern United States is due to a combination of the melding together of plumes from multiple urban areas as well as O_3 generation in areas relatively remote from large emission sources. Because the air entering an urban area may contain an O_3 concentration that is a significant fraction of the federal standard, it becomes difficult (some would say impossible) for a city to control its own air pollution problem. The regulatory agencies are becoming more cognizant of this concern. Thus, the most recent amendment to the Clean Air Act mandates nation-wide controls for NO_x and VOCs from motor vehicles and NO_x from power plants. The 1990 amendment also establishes an interstate ozone transport region extending from Washington, DC to Maine, explicitly recognizing that standard violations can be due to upwind source regions.

Ozone Photochemistry

The essential ingredients to form O_3 in the atmosphere are NO_x, VOCs and sunlight. Note that we are restricting our attention to the troposphere; a different set of starting materials and reactions assumes primary importance in the stratosphere. NO_x and VOCs have both natural and man-made sources. In urban areas the man-made sources far outweigh the natural ones. This is often not the case in rural areas where emissions of isoprene and other very reactive hydrocarbons

from trees can outweigh the man-made sources. Principal sources of hydrocarbons are motor vehicle exhaust, gasoline evaporation, and solvent evaporation. NO_x is emitted from high temperature combustion in motor vehicles and power plants.

Photochemistry is initiated by the absorption of a near UV photon splitting a chemical bond and forming a reactive free radical. Ozone itself is often the most important initiator producing a hydroxyl radical, (OH):

$$O_3 + h\nu \text{ ---> } O_2 + O(^1D) \qquad \text{(R1)}$$

$$O(^1D) + H_2O \text{ ---> } 2\,OH \qquad \text{(R2)}$$

Next follows a series of chain propagation steps which can involve reaction of OH with any of the thousands of VOCs present in the ambient atmosphere. A simple example involves reaction of CO.

$$OH + CO \xrightarrow{O_2} HO_2 + CO_2 \qquad \text{(R3)}$$

$$HO_2 + NO \text{ ---> } NO_2 + OH \qquad \text{(R4)}$$

$$NO_2 + h\nu \text{ ---> } NO + O \qquad \text{(R5)}$$

$$O + O_2 \text{ ---> } O_3 \qquad \text{(R6)}$$

The net effect of this sequence is that CO is oxidized to CO_2 and that one molecule of O_3 is created. VOCs (using CO as an example) are in essence the fuel that runs the photochemistry. NO_x is not removed in the above cycle and can be regarded as a catalyst. Chain termination occurs by reactions of free radicals with each other, for example,

$$2\,HO_2 \text{ ---> } H_2O_2 + O_2 \qquad \text{(R7)}$$

and more importantly in urban ares by reactions of NO_x with free radicals such as

$$OH + NO_2 \text{ ---> } HNO_3. \qquad \text{(R8)}$$

The last reaction removes NO_x from the active photochemistry and thereby necessitates a continued supply of NO_x for the photochemistry to proceed. NO_x and free radicals can be temporarily removed from the active photochemistry via the formation of peroxyacetyl nitrate (PAN).

$$CH_3CO_3 + NO_2 \text{ ---> } PAN \qquad \text{(R9)}$$

The above reactions are a schematic outline of the 100 or so reactions typically treated in computer models of the polluted lower (altitude) atmosphere. Such models also treat the

transport of pollutants, necessitating a description of wind fields and a geographically accurate map of emission sources. Depending on application, the spatial modeling domain can cover an urban area, the entire planet (albeit with less detail in the chemistry) or something in between. Much of what we know about the generation of photochemical smog is the result of computer calculations coupled with observations of the ambient atmosphere or coupled with controlled smog chamber experiments.

Control strategies are based in large part on model calculations. Typically a model is run for a high O_3 episode for which there are observations to compare against. Having demonstrated skill in predicting the observed O_3 levels (and less often concentrations of other photochemically active substances), the model is then rerun with different levels of NO_x and VOC emissions corresponding to plausible (and in some cases, implausible) control strategies. Results are often displayed as isopleth maps showing the maximum O_3 produced in an air parcel during the course of a day as a function of NO_x and VOC emission rates (or sometimes as a function of early morning concentrations of NO_x and VOCs). A schematic example is given in Figure 2. Arrows on the diagram indicate three possible control strategies that would result in meeting a 120 ppb standard starting from a point with an O_3 concentration of 200 ppb. In this case the standard could be met with control of NO_x, control of VOC, or combined control of NO_x and VOC.

Ozone isopleth graphs, while specific to individual cities, all have common generic features. Ozone production occurs most efficiently when the ratio of VOC to NO_x is about 8 to 10. At a very high ratio, the atmosphere is NO_x limited. Ozone production is then insensitive to

the amount of VOC; only NO_x controls are effective. At a very low ratio, the atmosphere is VOC limited. Ozone production can then be controlled by reducing VOC, but reductions in NO_x can in some extreme cases lead to more O_3. NO_x limited and VOC limited regions are shown in Figure 2.

The relative amounts of NO_x and VOCs emitted and in the air is thereby an important piece of information in deciding on an O_3 control strategy. Regulatory efforts have focused largely on control of VOC emissions. There has been vigorous debate as to whether this is an optimum policy[9]. Recent evidence shows that most of the United States is in a NO_x-limited condition most of the time. VOC-limited conditions are still expected in major urban areas, especially at times leading to peak O_3 levels. Sillman[7] notes the difficulty that this situation poses for the regulatory agencies. The optimum strategy for an individual city is often to lower O_3 production in the city by reducing VOC emissions while persuading the neighboring upwind regions to reduce NO_x emissions in order to lower O_3 in the "background" air entering the city.

Meteorological Factors

Figure 2 presents information on the response of O_3 to changes in its chemical precursors, NO_x and VOCs, but leaves unsaid the fact that O_3 concentrations depend on a host of other variables. Primary factors are ventilation, sunlight, temperature, humidity, rain fall, and wind direction vis-a-vis the location of upwind emission sources. In areas with poor ventilation (low wind speed and low mixing height) there is a continued input of pollutants to an air mass,

resulting in high concentrations of emitted substances as well as high concentrations of their reaction products. Stagnant conditions in the summer usually co-occur with high humidity, temperature, and solar intensity and tend to produce high concentrations of O_3 and other reaction products. In many instances wind direction is a controlling variable. With winds from one direction, an ocean could be upwind, while from another direction, an urban area could be upwind.

Effect of Global Climate Change on O_3

Rising CO_2 levels are expected to perturb the climate of the earth. For the purpose of this planning exercise we rely on the scenarios generated for the "Metropolitan New York in the Greenhouse" workshop (A. J. Broccoli, personal communication, based on the IPCC report[10]) which specify a best estimate global annual average temperature rise of 1.1°C in the year 2030 and 2.4°C in the year 2070. The wintertime and summertime temperature rise in New York are predicted to be 1.5 to 3°C and 1 to 2°C, respectively. By 2070, the temperatures are predicted to rise by 3 to 6°C in the winter and 2 to 4°C in the summer.

There are multiple ways in which a temperature change of this magnitude coupled with changes in other climate variables could have an impact on atmospheric chemistry in general and formation of O_3 in particular. Mechanisms suggested include:

1. An intrinsic dependence of rates of atmospheric chemical reactions on temperature.

Oxidation reactions of OH with hydrocarbons (schematically indicated by R3) proceed faster at high temperature. PAN formation (R9), which inhibits the production of O_3 by removing NO_x and radicals, becomes less significant at high temperature.

2. Changes in emission rates in response to increases in temperature. Anthropogenic emissions of NO_x and hydrocarbons would increase (all other things being equal) due to increases in solvent evaporation, motor vehicle use, and electric generation. Natural emissions of hydrocarbons from trees are a steeply increasing function of temperature. The large contribution of natural hydrocarbons to O_3 generation over much of the eastern United States is one of the major reasons that O_3 control policies, based on restricting emissions of anthropogenic VOCs, have been less successful than anticipated.

3. Changes in atmospheric humidity. Water vapor is a reactant in the photochemical smog system leading to the production of reactive free radicals (R2).

4. Changes in cloud cover, which affect solar intensity, which in turn affects the rates of photolysis reactions (i.e., R1 and R5).

5. Changes in precipitation patterns. The air is almost always cleaner during and following rain events. This is due to the removal of some pollutants, most notably aerosol particles which are responsible for visibility degradation. Precipitation is also associated with good ventilation and reduced sunlight.

6. Changes in wind patterns. Of particular importance are the frequency and severity of stagnation conditions that allow for the multi-day build up of air pollutants. Changes in the patterns of prevailing winds could bring air that is more or less polluted to a particular receptor site.

7. Changes in the height of the mixed layer (the layer of the atmosphere above the surface of the earth, typically extending to an altitude of about 1 km in which there is relatively vigorous mixing) making available a greater or lesser volume for dilution.

The global climate models that are used for predicting the temperature affects of increased greenhouse gases also yield predictions for the changes in other climatic variables such as precipitation, storm tracks, and cloud cover. However, we regard these as too uncertain to base a planning exercise on.

We can quantify the effects of a warmer climate on O_3 levels in two ways; by examining the historical relation between O_3 and temperature and by performing model calculations in which temperature or other climatic factors are varied. The historical relation, based on coincident observations of temperature and O_3, is the more direct approach but does not tell us whether temperature itself is the controlling factor or whether both temperature and O_3 are varying in response to another factor such as sunlight intensity. This is a critical question since we are hypothesizing a future in which temperature increases but, for example, solar intensity may remain constant or even decrease. Models have their own uncertainties but have the advantage of being

able to address the effects of several factors independently.

Observations

Monitoring data indicate that O_3 concentration tends to increase with temperature, the effect being most pronounced at temperatures above 27°C. Figures 3 and 4 illustrate the relation between daily maximum values of O_3 and temperature for New York City and for a set of four nonurban locations. There is considerable scatter to the relation between O_3 and temperature, reflecting the influence of other chemical and meteorological factors such as emission rates, sunlight, and wind speed. As noted in the EPA Criteria Document,[11] there is a well defined upper bound on O_3 concentration that reflects the maximum concentration achieved under optimum conditions. This maximum is seen to increase with temperature. Statistics for several locations are summarized in Table 1. Jones et al.[12] present another view of the relation between temperature and O_3 by comparing the number of exceedences of the Federal O_3 standard with the number of days in which the temperature was above 90°F, for several cities including New York City. As noted in the National Research Council report[9], there is a strong correlation between these two variables. There are also strong correlations between temperature and other air pollutants as indicated in Figure 5 for aerosol particles in Philadelphia.

A straight-forward explanation of the data in Figures 3 and 4 is that maximum O_3 concentrations do indeed depend in a causal way on temperature. However, it is likely that other factors contribute to this relation, in particular for regional high-O_3 episodes over the eastern

United States. Jacob et al.[13] note that regional episodes appear to be driven primarily by stagnation conditions. Low wind speeds allow for the continued addition of emitted NO_x and hydrocarbons to an air mass and cause O_3 to build up over a multi-day period. Jacob et al. further note, that the correlation between O_3 and temperature may reflect the dependence of temperature on air mass origin or solar radiation. Part of the association between temperature and O_3 in New York City may be due to the circumstance that warm air generally arrives from the south or southwest, from regions that have high emission rates of O_3 precursors and high O_3 concentrations. This association is also thought to explain much of the dependence of aerosol concentration on temperature.

Models

Morris et al.[14] applied a regional photochemical model to the study of O_3 episodes in central California and the Midwest/Southeastern United States. Results of increasing temperature by 4°C with an attendant increase in water vapor varied from day to day and place to place. In California the daily maximum O_3 concentration increased by 2 to 20%, while in the Midwest/Southeast the change varied from -2% to 8%. In both studies there were large increases in the population exposed to O_3 levels above the 120 ppb standard. The sensitivity of O_3 to temperature in New York City was tested by Gery et al.[15] using an "EKMA" model similar to that used to generate the isopleth map shown in Figure 2. A 2°C temperature rise was found to cause an increase in O_3 from 125 to 130 ppb. Similar results were found for Philadelphia and Washington, DC. A modeling study by Sillman and Samson[16] predicts that approximately half of

the observed increase of O_3 with temperature in Detroit and a rural site in Michigan can be accounted for by a combination of temperature-dependent emissions of natural hydrocarbons and temperature-dependent rate constants. A mechanistic analysis indicates that the temperature dependence results primarily from the formation of PAN tying up free radicals and NO_x as the temperature decreases, thereby inhibiting O_3 production under NO_x limited (rural) and VOC limited (urban) conditions.

Although the three simulations cited above are for different regions and incorporate somewhat different assumptions on what else changes when temperature is increased, they agree to the extent that they predict urban O_3 increases of about 3 ppb per degree C temperature increase. This is about 1/3 of the observed O_3 trend in NYC; ancillary changes in meteorological conditions accounting for the other 2/3. Our sensitivity estimate is based on starting with about 120 ppb of O_3. Additional calculations are needed to determine if a larger increase in O_3 occurs if the base case concentration is greater. We expect a temperature rise to change the rates of hydrocarbon reactions and the availability of NO_x, effectively mimicking emission increases of these substances. This will shift our position on the O_3 isopleth map, changing the optimum emission control strategy. The magnitude and even direction of such a shift cannot be determined without detailed calculations specific to the New York metropolitan area.

A postulated 2°C increase in temperature, according to the above analysis, translates into a 6 ppb increase in O_3. Given the year to year variations in O_3, such as displayed in Figure 1, it is not likely that a change of this magnitude would be evident from monitoring data. Other changes

in climate variables could make this figure lower or higher.

THE HEALTH EFFECTS OF CLIMATE CHANGE

It has long been known that extreme weather can have extreme effects on health, and through the centuries man has developed effective means of protection. Further, populations have been shown to adapt to their climates in various ways, including design of shelters and clothing and through personal habits. Thus, it is to be expected and analysis has shown that perturbations about the normal weather patterns carry larger risks than do steady gradual changes in climate. For example, a heat wave occurred in Athens in July 1987, with daily mean temperatures around 35°C (95°F) for six days with a peak daily maximum value of 41°C (106°F)[17]. This was an increase of only about 7-9°C over normal levels, but the daily death rate increased by an order of magnitude, from an average of about 23 to over 200 deaths per day. Yet, such temperature levels are experienced routinely in many parts of the world, including the United States. In New York in 1993, for example, a peak temperature of 102°F was reached with a mean of 91°F, over a 3-day period. If daily deaths increased during this period in New York, the increase went largely unnoticed.

The first section of this paper indicated that changes in climate may also bring changes in ambient air quality, both directly and indirectly. Direct changes may result from changes in atmospheric reaction rates, for example. Indirect effects may result from changes in atmospheric transport and dispersion factors, and from increased pollutant emissions resulting from man's

attempts to escape the heat.

Health Effects of Air Pollution and Weather

The early analyses of the most severe health effects of air pollution,i.e., premature mortality and respiratory hospitalizations, tended to focus on winter episodes[18]. In winter, air quality problems tend to be caused by primary emissions from space heating and by reduced atmospheric dispersion during stagnation episodes. Such winter events may coincide with the periods of peak mortality and the presence of infectious respiratory diseases, such as influenza and pneumonia. However, since the Clean Air Act of 1970 and its progeny, space heating fuel quality has improved markedly, especially in New York City, to the point where summer may be the period of peak air pollution. In summer, secondary pollutants including ozone and fine particles are the focus of concern. This not only changes the chemical focus, but also the locations of concern, since the ambient concentrations of secondary pollutants usually peak well downwind of their sources, say ca. 50-100 km or more, and the pollution becomes dispersed over a wide area. Also, in hot weather, many people spend relatively more time outdoors, thus increasing their exposure to air pollution. This may be especially true of those lacking access to air conditioning.

The numbers of deaths officially classified as "heat related" may substantially underestimate the true effects, in part because of the uncertainties of cause-of-death coding. For example, in Philadelphia, not only is a hot environment required but the decedent must also be elderly or infirm to be listed under this cause of death[19]. During 1979-1988, only 4523 deaths or

about 0.023% of all deaths in the United States were officially attributed to "excessive heat exposure," and 1700 of those were in one year. Statistical analyses of the timing of death (described below) tend to find much larger effects of increased ambient temperatures. Interestingly, air conditioning is recommended as a protective measure, but not fans.[19] Increased ventilation of air above skin temperature would exacerbate heat stress, and bringing outside air into the home could add to air pollutant exposure. 118 deaths were attributed to heat stress over a 9-day period in Philadelphia (about 26% excess);[20] this heat wave was also experienced in New York City, where the peak temperature reached 102°F and ozone levels also peaked.

The health effects of these excursions in environmental conditions are typically analyzed using the methods of time-series analysis[21-30]. The basic technique involves various types of multiple regression analysis, in which daily counts of deaths, hospital admissions, or other indices of health are the dependent variable. Independent variables include daily measures of air quality and other temporal factors which might be correlated with health and air quality. These include weather variables, of which temperature is usually the most important, and other perturbations such as days of the week, holidays, and influenza epidemics. It is also important to account for long-term trends that might confound the analysis. These include both secular trends and seasonal cycles. In the former, it is possible that coincident trends in improved health, resulting from better medical care or healthier lifestyles, for example, might be (incorrectly) associated with a concurrent trend in improved air quality resulting from imposition of emissions controls. Seasonal cycles may confound, for example, if the expected winter peak in mortality or respiratory illness (resulting from infectious disease cycles) is inappropriately associated with the seasonal cycle in

air quality that results from increased space heating emissions and reduced atmospheric dispersion in winter.

Epidemiologists have developed two major techniques for distinguishing the short-term effects of air pollution from those of these potential confounders. Schwartz and his colleagues,[25] for example, use a number of auxiliary independent variables as controls. These include trend variables and dummy variables for extreme events like heat waves; daily temperature is used primarily as a control for seasonal cycles. In general, this method does not yield a regression coefficient for hot weather effects as a function of maximum temperature. Other investigators[22,26,27] have used the classical time-series methodology called "filtering," in which new "perturbation" variables are created by subtracting the appropriate running mean from each original variable. Daily temperature is used as a predictor of daily health status in this technique, rather than as a seasonal control. Filtering absolutely removes the long-term trends, but the regression coefficients may be sensitive to the details of the filtering process[24]. Comparison of these two basic regression techniques suggests that filtering tends to assign less of the observed health variance to air pollution, which has been interpreted as a failure of the first method to control sufficiently for long-term trends and seasonal cycles[26].

It is important with all analyses of daily variations in health status to account for lags between exposure and response; plots of response versus lag may help us interpret the regression results. For example, the response must lag the exposure if the association is to be regarded as causal. In addition, the degree of prematurity of response may be assessed by considering

whether initial positive responses may be effectively compensated by subsequent negative responses, such that the sum over a longer time period approaches zero. In mortality analysis, this compensation process has been referred to as "harvesting" or mortality "displacement"[2]. Figures 6-8 compare some of these lag plots. In Figure 6, hospital admissions in Southern Ontario in July-August[23] increase with ozone and sulfate aerosol beginning on the day after exposure and show no indication of a subsequent decrease (perhaps longer lags should have been examined). In Figures 7 and 8, daily mortality on hot days in Philadelphia also increases with ozone or particulates on the day after exposure and then decreases on subsequent days. In contrast, the effects of temperature are felt immediately (Figure 9), and show subsequent decreases, more so for those under age 65.

These air pollution effects have been shown to exist at air quality levels well within current State and Federal standards; thus exceedances of those standards is not an appropriate criterion for evaluating the severity of health effects. The Federal government is currently considering lowering these standards, but it seems unlikely that this type of health risk can ever be completely eliminated. The question that naturally arises is the extent to which air conditioning might help: air conditioning can not only protect against atmospheric heating, but, given appropriate air filters, it can also protect against outside air pollution. Rogot et al.[31] estimated that mortality during "hot" weather (average temperature > 70°F) was 42% lower nationwide for those having central air conditioning (a/c), compared to those with no a/c, based on a nationwide sample of 2275 deaths. Their analysis used mortality during "non-hot" weather as controls to preclude socioeconomic confounding. Rogot et al.[31] found a/c to be of greatest benefit in Florida. Without

Florida, the national mortality benefit dropped to 29%; their analysis showed no a/c benefit in New York, but there were only 11 New York deaths during hot weather in this sample. However, Kalkstein[2] cited an estimate of 21% savings in heat-related deaths in New York due to a/c, using a different analysis technique.

In the analysis of Rogot et al.[31], the benefit attributed to a/c could include relief from both heat and from outside air pollution and we regard 21% as an upper limit estimate for New York. The Philadelphia statistical analysis[26] probably provides a better basis for estimating effects in New York, due to the proximity of the two cities, although Philadelphia summers tend to be slightly hotter than New York.

Estimated Effects of Global Warming in New York City

In the first section of this paper, an increase of 6 ppb ozone over a baseline of 120 ppb was predicted, based on a temperature increase of 2°C. We will use these estimates to predict changes in daily mortality in New York City, assuming no changes in pollution emissions. Although air pollution and especially ozone can have many different types of health effects, we use premature mortality as an index, in part because of its irreversible nature and in part because of the availability of the required coefficients for the effects of heat and of air pollution.

Mendell et al.[32] found an average summer mortality/temperature coefficient in several different locations of 0.45% per degree F (0.81% /°C); however, these studies may have

neglected the harvesting effect mentioned above. In addition, graphical analysis shows that the effects of temperature in many cities tend to increase dramatically after some threshold is crossed, typically around 33°C (91°F). Thus, hot days should be examined separately to gain a better understanding of the health responses. Wyzga and Lipfert[26] looked at days with maximum temperatures over 85°F in Philadelphia, as a subset of all days from 1973-80. Their analysis combined lags up to 4 days in order to allow for mortality harvesting. On a year round basis, for example, using the Philadelphia analysis for all days, a 2°C temperature increase would increase daily mortality by 0.45%, or by about 340 deaths per year in New York City. Computed using the regression results for "hot" days alone (assumed to be 18% of the total), the effect of 2°C would be about 3.4%, or about 470 deaths per year. This exercise suggests that mortality changes due to temperature may be neglected in other seasons.

Interactions between summer ambient temperatures and air pollution have been shown in several cities, including New York (see Figures 3-5). After filtering to remove trends, Thurston et al.[28] showed positive correlations in Toronto from 0.38 to 0.68 between maximum daily temperature and ozone, suspended particulate matter, sulfate aerosol, NO_2, SO_2, and aerosol acidity. Lipfert and Hammerstrom[23] showed similar correlations there based on the unfiltered data. Schwartz[29] showed temperature correlations of 0.67 and 0.32 for ozone and PM10, respectively, in Detroit. In their study of summer hospital admissions in New York State, Thurston et al.[30], found positive bivariate correlations between daily maximum temperature and hospital admissions, as well as with ozone, sulfate aerosol, and aerosol acidity.

In order to estimate the effects of the air pollution increases that would accompany increases in ambient temperature, regressions are required that estimate the temperature and pollution effects jointly (along with any other potentially confounding variables). The ozone-mortality coefficients from the analysis of the filtered Philadelphia data[26] were 0.008 for those under 65 and 0.002 for those 65 and over on a year-round basis, expressed as percent change in mortality for a 1% change in ozone. An increment of 6 ppb over a baseline of 120 ppb is an increment of 5%, which then translates into a mortality change of about 0.02% or about 16 deaths per year. The mortality effects of ozone were not stronger on hot days, perhaps because of the competing direct effect of temperature.

However, of the hospitalization studies, only Burnett et al.[27] provided a regression coefficient for temperature; their results indicated that the 2°C temperature increase from global warming would increase respiratory hospital admissions by about 0.5% and the accompanying ozone (6 ppb) would create an additional 0.6% increase in admissions, for the summer period.

The health effects of increased airborne particles should also be taken into account in these estimates, which requires separation of the effects of daily perturbations from seasonal and long-term effects. The association between mortality and particulates was shown by Wyzga and Lipfert[26] to increase dramatically on hot days; the relationship between TSP and temperature is also much stronger on hot days in Philadelphia. On a year-round basis, an increase of 2°C would correspond to a TSP increase of 0.5 ug/m^3 or about 0.7%, based on a regression model which also accounted for wind speed, precipitation, relative humidity and change in barometric pressure.

This would result in an increase in mortality of 0.01% for those aged 65 and over. For days with maximum temperatures of 85°F or more, which amounted to about 18% of the total, an increase of 2°C would correspond to an increase of 2.8 ug/m^3 or about 3% on those days. This would result in an increase in mortality of 0.05% for those under 65 and 0.32% for those 65 and over. These figures result in a weighted year-round increase in total mortality of about 0.04% The fact that hot days account for a disproportionate share of the total effect implies a nonlinear relationship, which was confirmed by investigating other ranges of daily maximum temperature. TSP was not significantly associated with temperature in any other (lower) temperature range, and the slope became negative below about 40°F, apparently because lower temperatures in winter imply higher emissions from space heating. The results of these calculations are summarized in Table 2. (Note that we have disregarded the lack of statistical significance of the effects of air pollution on daily mortality seen in Figures 7 and 8, for the purpose of making these estimates.) Based on the 1987 death count for New York City of about 76,250, the estimated total effect of global warming would result in an annual increase of about 511 deaths, 90% of which are due to the effects of temperature. The combined estimate for respiratory hospital admissions[30] would be about 200, based on summer days only. Premature mortality thus appears to be the more important health impact, in terms of both severity and frequency.

Mitigation Options

Air Conditioning

In theory, it should be possible to mitigate a large portion of the effects of global warming through air conditioning. We assume that most of the excess deaths occur to those who are already sick, and therefore we use residential air conditioning as the paradigm. These estimates are necessarily crude and are presented here only for the sake of illustration and discussion. In order to provide a worst-case illustration, we assume that the additional energy is to be supplied by new fossil-fueled power plants located in the metropolitan area.

Some handbooks estimate that about 0.003 ton of air conditioning or 36 Btuh is required for each additional square foot of conditioned space. We assume that efficiencies have increased over time and take 2 tons of refrigeration per household or 1 ton per person as a rough estimate for New York City. This requires an electrical capacity of about 2 kw per person, and electrical system planning data indicate that about 2000 kwh per household-year is required in New York for central air conditioning (personal communication, P. Coffey, 1994). Taking the required additional market penetration of residential air conditioning at 50%, we estimate that about 7500 MW of new electrical generating capacity would thus be required, or an approximate doubling of the present Con Edison system. At $0.15 per kwh, the additional seasonal operating cost would be around $300 per household or about $560 million, to which the costs of capital for air conditioning and generation should be added (increases in electric rates as a result of new plant construction are not included in this estimate).

The additional air pollution from this electrical generation must also be considered. We estimated emission increases of about 4,130 tons of SO_2 and NO_x (each) and about 1,030 tons of

particulate matter over a 100-day period of air conditioning. If emitted from tall stacks, these emissions would amount to about 1.7 and 0.43 ug/m^3 at ground level, respectively, or increases of about 5% or less over present background concentrations. The increased NO_x emissions (about 5% on a summer day) would probably also contribute to the formation of additional O_3 downwind of the city. These increases are of the same order as the corresponding global warming increases (which of course would still be there).

The use of air conditioning can protect against all heat waves and most of the outside air pollution, not just the increases from global warming. This implies that many additional lives would be prolonged, beyond those associated with the global warming increase. Since economists often use several million dollars as the value of a "statistical life," a policy of increased residential air conditioning seems cost effective. This is likely to be the case even when other types of environmental costs are considered.

Trading Reduced Air Pollution Against Increased Heat

Additional air conditioning will result in increased CO_2 emissions (about 3.5 million tons in the example above) if the energy is supplied by fossil fuels. This will provide a positive feedback effect which will tend to worsen global warming along an accelerating path, although the increase in CO_2 will be small because the additional energy is only needed during hot weather, not year round. It has been suggested that a better policy might be to reduce air pollution by imposing additional controls on emissions or fuel use, such that the increased health effects of

heat are offset by better air quality. Such a scheme could have a negative feedback effect if overall fuel use is reduced as a result of the policy.

The magnitudes of the air pollution reductions required may be estimated from the figures given in Table 2. The leverage of air pollution effects with respect to temperature is less for mortality, which we use as the basis for this example. If we attempt to use ozone reductions to compensate for the effects of heat waves, the reduction would have to be about 30 times the increment used in Table 2 or about 150 ppb. Since the baseline level was 120 ppb, this scheme is clearly impractical. However, the compensating reduction that would be required from particulate matter is more modest, about 15 times the increment shown in Table 2 or 7.5 ug/m^3 on a year-round basis or 42 ug/m^3 on the hottest days (a reduction of about 30%). While such a tradeoff scheme may seem feasible in theory, one must realize that this is an attempt to trade "statistical lives" and that compensation may not occur on an individual basis. Heat and pollution events will not always coincide, and individuals may respond differently to different environmental insults. We have no guarantee that easing the overall particulate burden would in fact protect someone who happens to be especially sensitive to heat.

Concluding Discussion

This analysis has shown that the air quality and health effects to be expected from global warming in New York City appear to be relatively modest, based on ozone and particulates as indices of air quality and mortality as an index of health effects. However, there are other

considerations. The analysis is based on a uniform temperature increase, while the actual mortality effects of heat seem to result from daily perturbations. Unfortunately, we have no predictions of changes in the variability of conditions under global warming. Also, the analysis does not account for downwind effects in suburban or rural areas, and there are other types of health effects to consider, such as respiratory symptoms, some of which are more responsive to ozone. Effects of a less severe nature tend to apply to a larger segment of the population and thus could be more important than the mortality and hospitalization paradigms used here.

With regard to mitigation options, it seems clear that increased use of air conditioning can prolong lives and that reductions in fuel use may not only prolong lives but also mitigate the increase in global warming (if implemented on a sufficiently large scale). The challenge for the future is to find ways to accomplish both of these goals.

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Table 1. Observed dependence of peak O_3 on diurnal maximum temperature (ppb°C^{-1}) between April 1 and September 30, 1988 (Sillman and Samson, 1994).

	$\Delta O_3/\Delta T$	
Location	T < 27 °C	T > 27 °C
urban sites:		
NY-NJ-CT	1.5	8.8
Detroit	1.4	4.4
Atlanta	3.2	7.1
Phoenix	—[1]	1.4
Southern Calif.	11.3	—[1]
nonurban sites:		
Williamsport, PA	1.2	4.0
Saline, MI	0.8	3.1
Mammoth Cave, KY	—[2]	4.4
Kentucky, cleanest site	—[2]	3.4
Williston, ND	—[2]	0.8
Billings, MT	—[2]	0.7
Medford, OR	0.5	3.3

[1] Too few cases to be included

[2] Trend not statistically significant

Table 2 Health Effects of a Temperature Increase of 2°C.

	mortality	resp. hospital admissions
from temperature alone:	0.61%[26]	0.45%[27]
from ozone:	0.02%[26]	0.58%[27]
from airborne particles:	0.04%[26]	0.32%[29]
total effect:	0.67%	1.3%

Figure captions

1. Ozone trends in New York State. Source, NY/DEC, 1993.

2. An example of an O_3 isopleth map, showing predicted O_3 concentration as a function of NO_x and VOC. Adapted from Dodge[33].

3. Daily maximum O_3 concentration as a function of daily maximum temperature in New York City during the months from May to October, 1988 to 1990. Source, United States EPA[11].

4. Daily maximum O_3 concentration as a function of daily maximum temperature at four nonurban sites during the months from May to October, 1988 to 1990. Source, United States EPA[11].

5. Relationship between particle concentration and daily maximum temperature in Philadelphia, 1973-80. Adapted from Wyzga and Lipfert, 1994.

6. Correlations between hospital admissions for respiratory causes and air quality in Southern Ontario. (a) ozone. (b) sulfate aerosol. Adapted from Lipfert and Hammerstrom[23].

7. Regression coefficients for ozone on the deviations of daily death counts from a 15-day moving average, with various covariates included, on days with maximum temperature of 85°F or more. Adapted from Wyzga and Lipfert[26].

8. Regression coefficients for TSP on the deviations of daily death counts from a 15-day moving average, with various O_3 and temperature, on days with maximum temperature of 85°F or more. Adapted from Wyzga and Lipfert[26].

9. Regression coefficients for daily maximum temperature on the deviations of daily death counts from a 15-day moving average, with TSP and O_3 included , on days with maximum temperature of 85°F or more. Adapted from Wyzga and Lipfert[26].

Figure 1

Ozone (ppb)
0
50
100
150
200
250
300
1973
1974
1975
1976
1977
1978
1979
1980
1981
1982
1983
1984
1985
1986
1987
1988
1989
1990
1991
1992
Year
max
2nd max
0
10
20
30
40
50
Hours > 120 ppb

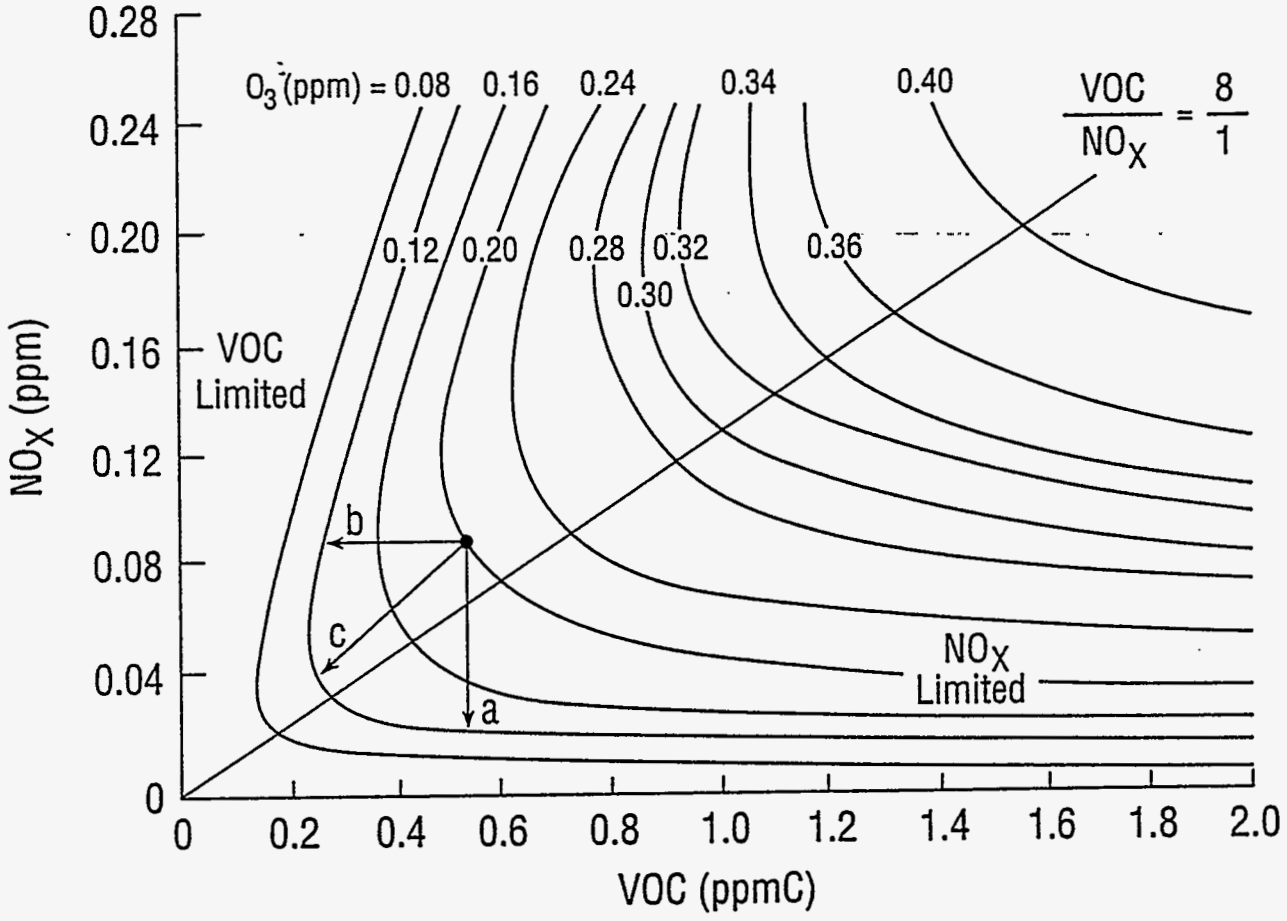

0.28
0.24
0.20
0.16
0.12
0.08
0.04
0
NO_X (ppm)
O_3 (ppm) = 0.08
0.16
0.24
0.34
0.40
$\frac{VOC}{NO_X} = \frac{8}{1}$
0.12
0.20
0.28
0.32
0.36
0.30
VOC
Limited
b
c
a
NO_X
Limited
0
0.2
0.4
0.6
0.8
1.0
1.2
1.4
1.6
1.8
2.0
VOC (ppmC)

Figure 2

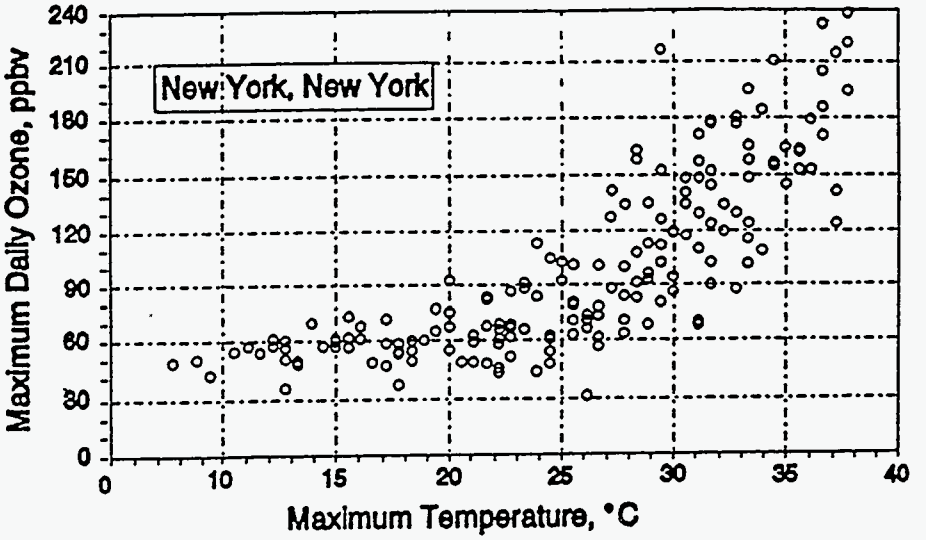


Figure 3

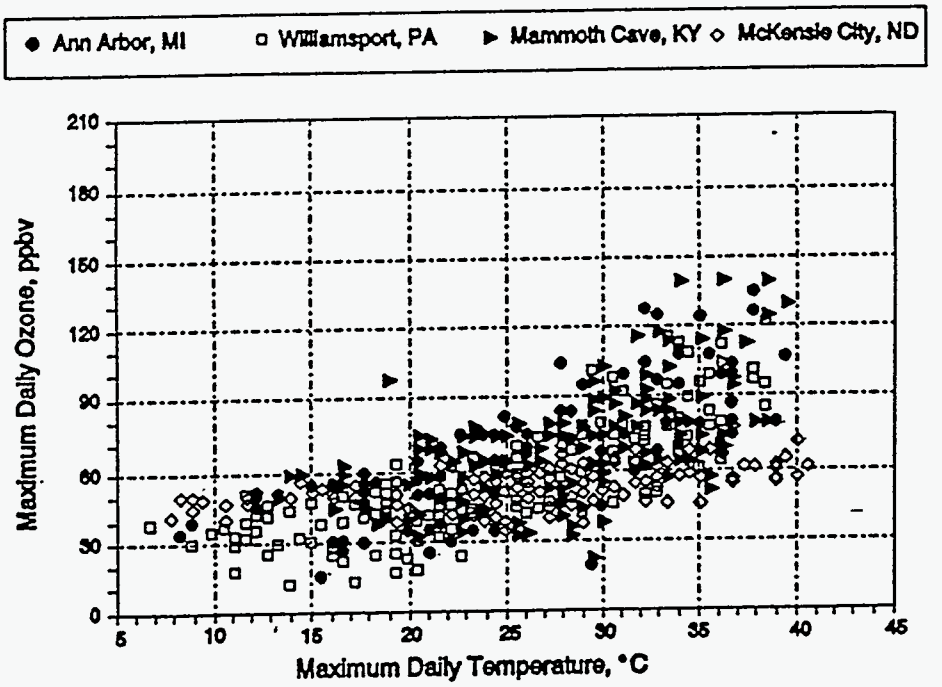
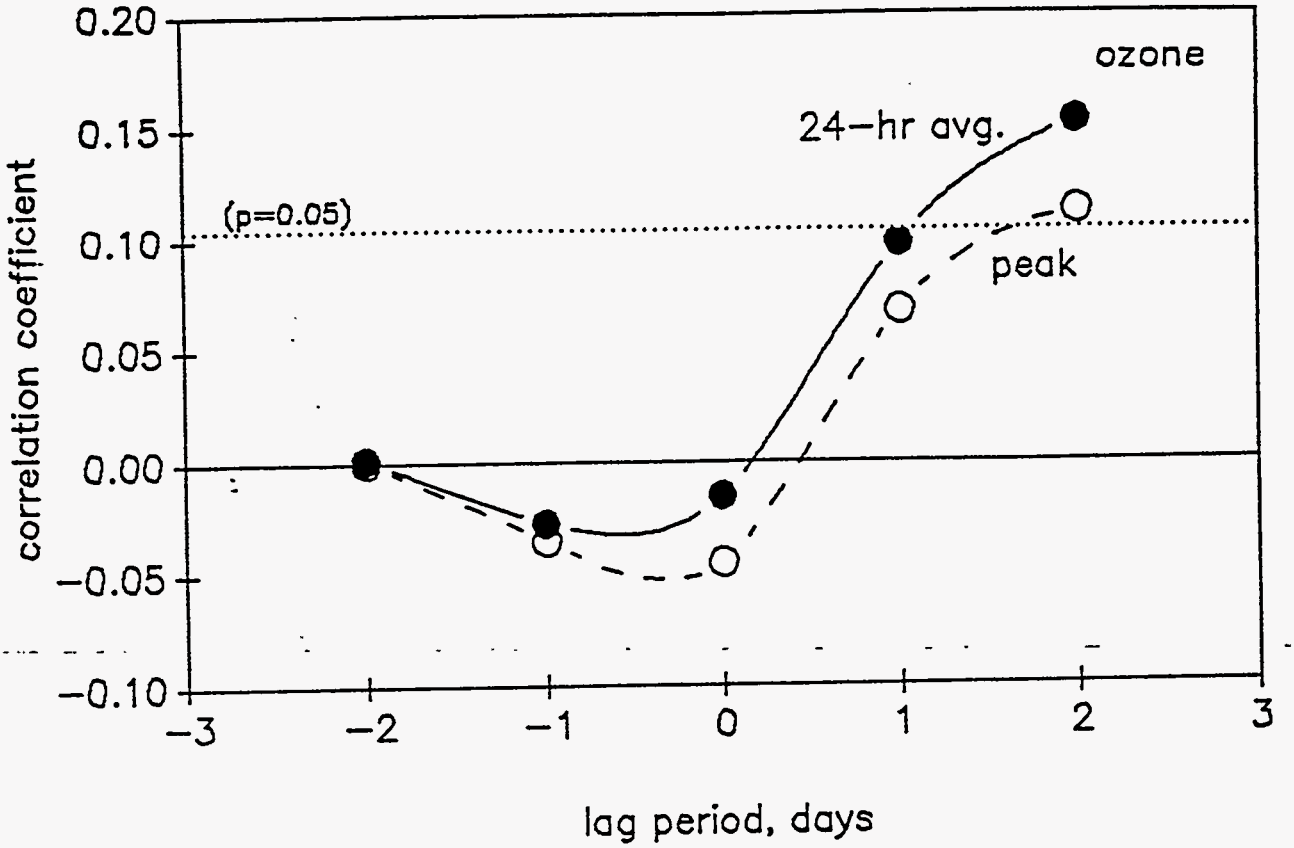
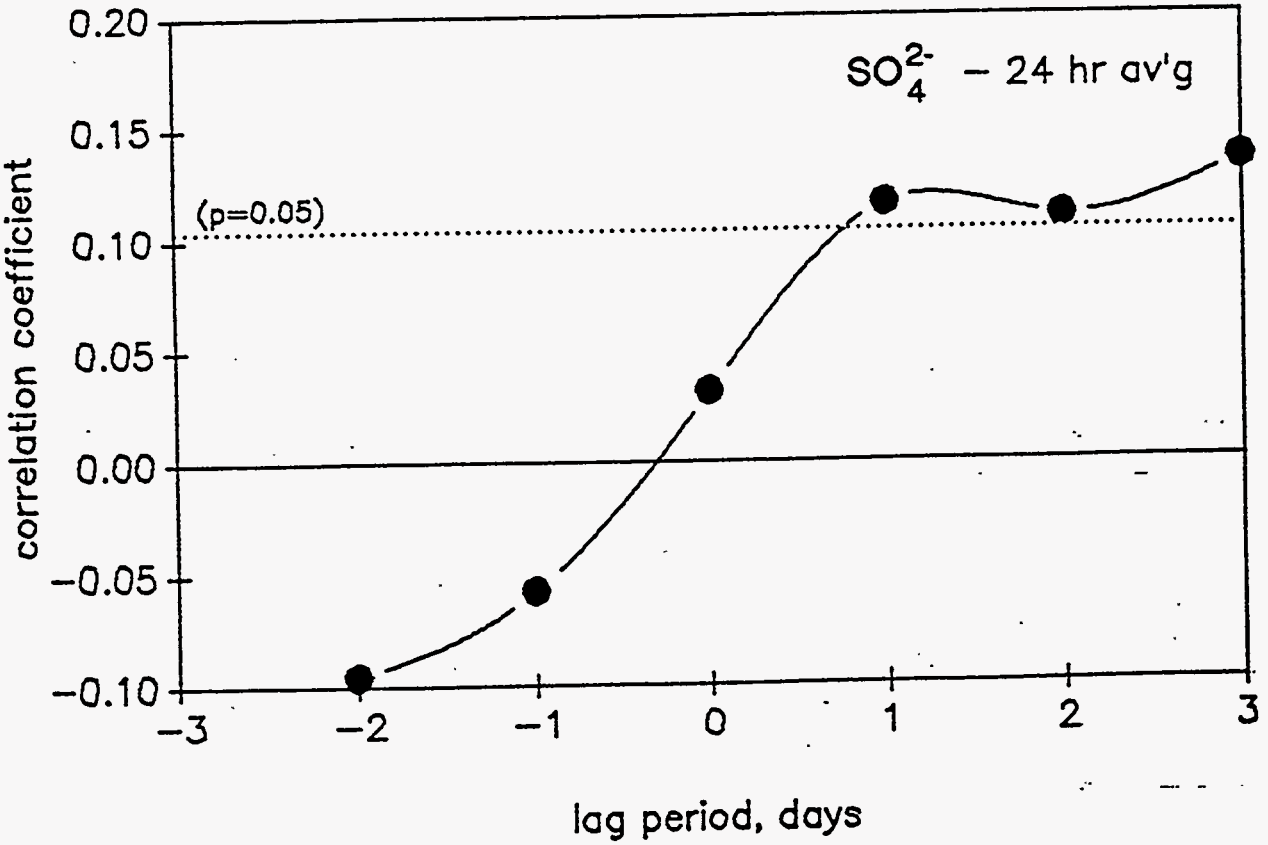


Figure 4


0.20
0.15
0.10
0.05
0.00
−0.05
−0.10
correlation coefficient
(p=0.05)
ozone
24−hr avg.
peak
−3
−2
−1
0
1
2
3
lag period, days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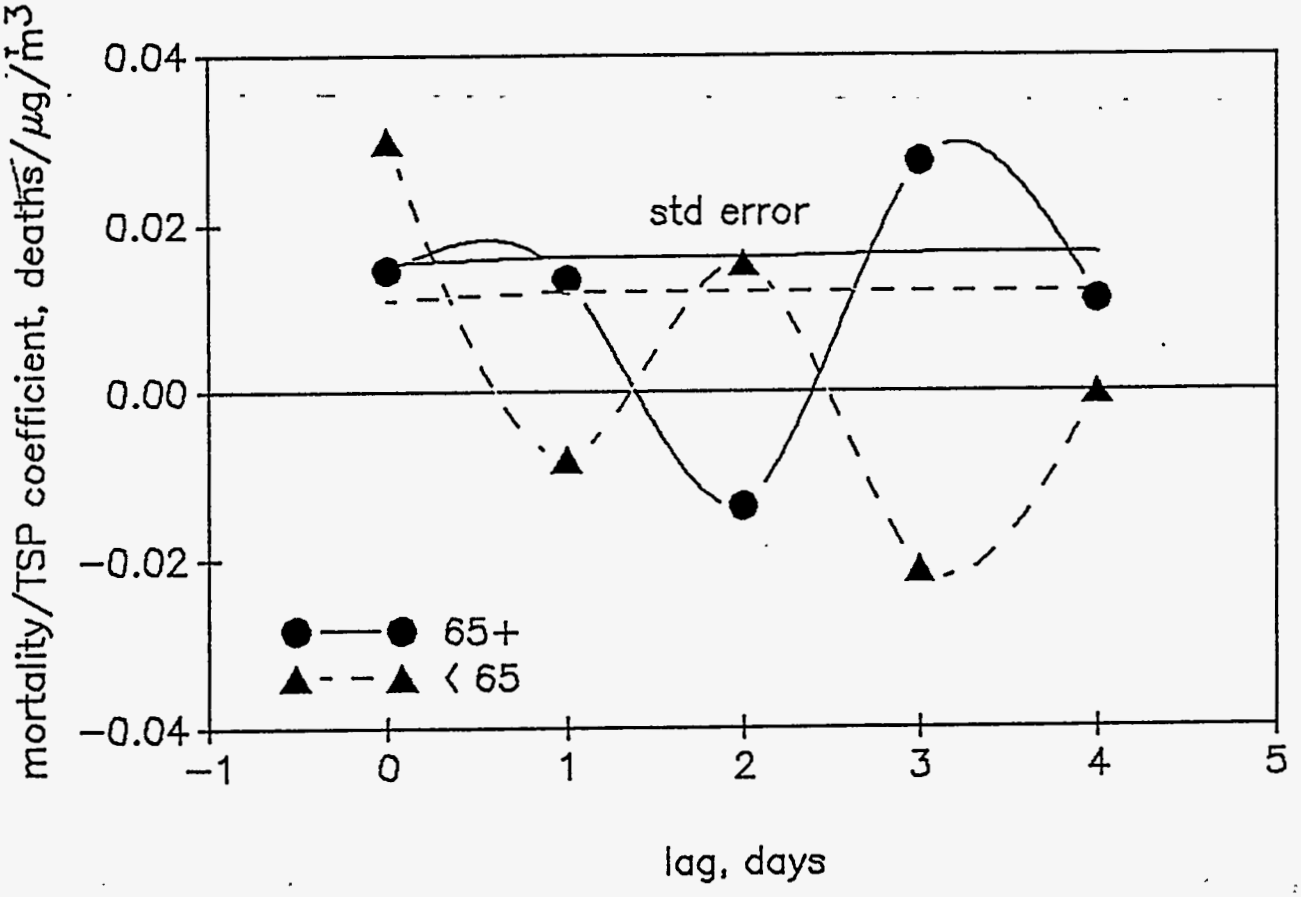
0.10
0.05
0.00
−0.05
−0.10
correlation coefficient
(p=0.05)
SO_4^{2-} − 24 hr av'g
−3
−2
−1
0
1
2
3
lag period, days


0.04
0.02
0.00
−0.02
−0.04
−1
0
1
2
3
4
5
std error
65+
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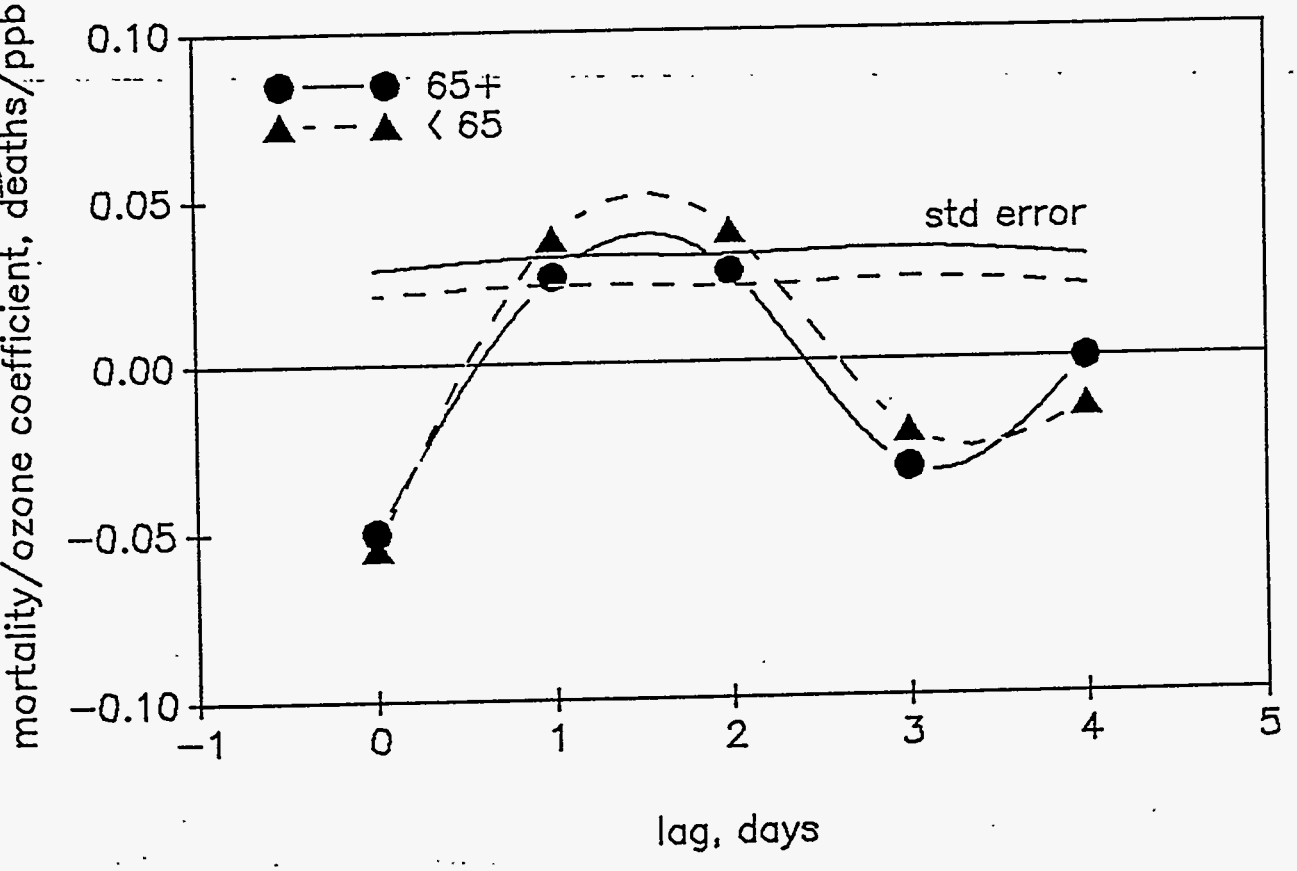
mortality/TSP coefficient, deaths/μg/m3
lag, days


0.10
0.05
0.00
−0.05
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−1
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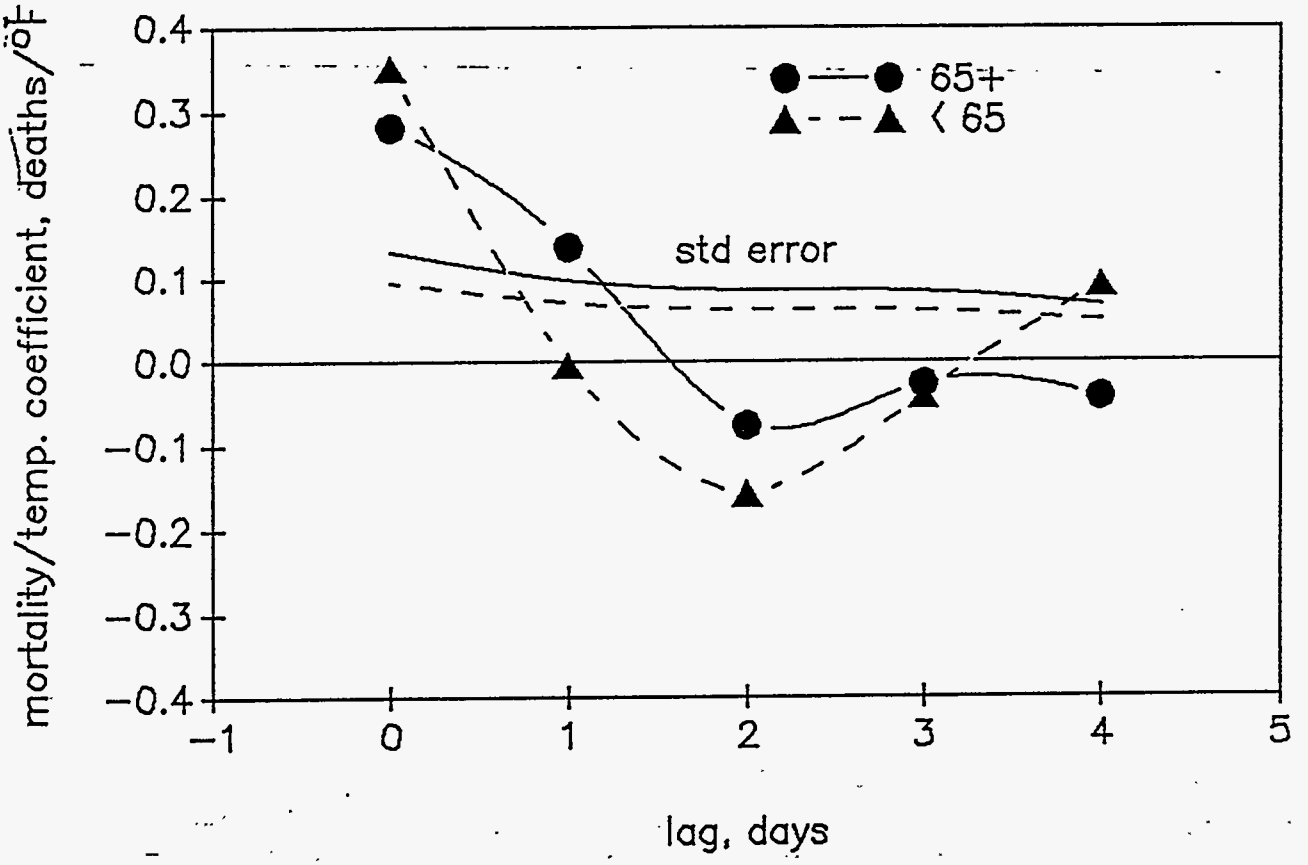
1
2
3
4
5
65+
< 65
std error
mortality/ozone coefficient, deaths/ppb
lag, days


mortality/temp. coefficient, deaths/°F
0.4
0.3
0.2
0.1
0.0
−0.1
−0.2
−0.3
−0.4
65+
< 65
std error
−1
0
1
2
3
4
5
lag, days