The 1997 US EPA Standards for Particulate Matter and Ozone

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1 Introduction

Under the mandate of the Clean Air Act (CAA) of the United States, as amended in 1977, the Administrator of the Environmental Protection Agency (EPA) is supposed to review the basis for its National Ambient Air Quality Standards (NAAQS) every five years and, if necessary for the protection of public health and/or welfare, issue new or revised standards. In the past, as shown in Table 1, the ozone (O_3) and particulate matter (PM) NAAQS have been revised, albeit not at the specified five-year intervals. The 1997 revisions to the NAAQS for PM and O_3 were unusual in NAAQS revisions in a number of respects, including especially: (1) their simultaneous promulgations; (2) the historically tight timetables involved in their preparation and public review; (3) the extraordinary controversy they engendered during the review process and since their promulgation; and (4) the substantial increases in the number of additional communities that will not be in compliance with the NAAQS for both PM and O_3.

The CAA requires the EPA Administrator to promulgate NAAQS that protect the health of sensitive segments of the public with an adequate 'margin of safety'. In addition to the health-based (primary) standards, there are also welfare-based (secondary) standards, referring to human comfort, economic damage to materials, crops, livestock, and ecological balance. The CAA is unique among the statutes that EPA enforces in that the primary NAAQS are supposed to be set without regard to the costs they impose on society. However, EPA does perform cost–benefit analyses for NAAQS to conform with a Presidential directive requiring such analyses for all regulations that significantly affect the national economy.

In the process of considering the revision of a NAAQS, the National Center for Environmental Assessment (NCEA) of the EPA's Office of Research and Development (ORD) prepares a Criteria Document (CD). This is a massive and encyclopedic summary of the peer-reviewed literature that bears upon the chemical and physical nature of the pollutant, its measurement in the ambient air, its primary sources and secondary transformations in the atmosphere, its

Table 1 1997 Revisions: US National Ambient Air Quality Standards (NAAQS)

I. Ozone (*Revision of NAAQS Set in 1979 and Reaffirmed in 1993*)

	1979 NAAQS	*1997 NAAQS*
Daily concentration limit (ppb)	120	80
Averaging time	maximum: 1 h av.	maximum: 8 h av.
Basis for excessive concentration	4th highest over 3 year period	3 year av. of 4th highest in each year
Equivalent stringency for 1 h max in new format (ppb)	~90	
Number of US counties expected to exceed NAAQS	106	280
Number of people in counties exceeding NAAQS	74×10^6	113×10^6

II. Particulate Matter (*Revision of NAAQS Set in 1987*)

	1987 NAAQS	*1997 NAAQS*	
Index pollutant	PM_{10}	PM_{10}	$PM_{2.5}$
Annual av. concentration limit ($\mu g\,m^{-3}$)	50	50	15
Daily concentration limit ($\mu g\,m^{-3}$)	150	150	65
Basis for excessive daily concentration	4th highest over 3 year period	>99th percentile av. over 3 years	>98th percentile av. over 3 years
Number of US counties expected to exceed NAAQS	41	14	~150
Number of people in counties exceeding NAAQS	29×10^6	$\sim 9 \times 10^6$	$\sim 68 \times 10^6$

transport and fate, its geographic distribution, methods of sampling and analysis, the exposures of populations and other receptors to the pollutant (both outdoors and indoors), its dosimetry once inhaled or deposited, the effects that it produces, and, in the most recent CDs, an integrated summary of sources, exposures, and responses.

EPA's plans for the organization of the CD are reviewed by the Clean Air Scientific Advisory Committee (CASAC), an independent external advisory committee created in response to a mandate of the CAA amendments of 1977.

Subsequently, external review drafts of the CD are reviewed in public by CASAC, which then makes recommendation for changes in the draft. There is typically a second external review draft and further CASAC and public commentary. When CASAC is satisfied that the CD is a complete and balanced summary and analysis, it prepares a 'closure letter' to the Administrator endorsing the document as a suitable foundation for setting a NAAQS. CASAC also publicly reviews a companion document prepared by EPA's Office of Air Quality Planning and Standards (OAQPS). This document makes selective use of the contents of the CD that are most relevant to the selection of critical elements in the NAAQS, such as the indicator pollutant(s), averaging times, method(s) for analysis of concentration, sensitive segments of the population, exposure–response relationships, and margin-of-safety associated with several alternative NAAQS levels under consideration. The final draft of this companion document, known as a Staff Paper (SP), is prepared after CASAC Closure on the CD. After a final public review, the CASAC chair prepares a closure letter on the SP.

The preparation of an economic impact analysis and further internal reviews in EPA and the federal Office of Management and Budget (OMB) precedes a Federal Register proposal to change or reaffirm an existing NAAQS. After a further public comment period, the EPA Administrator announces the promulgation of the NAAQS in the Federal Register.

This exhaustive review process by EPA Staff (ECAO and OAQPS) and CASAC inevitably identifies important knowledge gaps that limit confidence in the NAAQS options as optimal choices. Recognizing these limitations, EPA Staff prepares lists of research recommendations for public review by CASAC, and CASAC summarizes and prioritizes research needs in a letter report to the EPA Administrator. It is hoped that EPA and other research sponsors will then arrange to support research that can close the critical knowledge gaps, so that the next cycle of NAAQS reviews can lead to more refined and well targeted NAAQS.

The author has served on all of the CASAC panels reviewing CDs, SPs, and research needs for the PM and O_3 NAAQS since 1980 in various capacities (consultant 1980–1982; member 1982–1987; chairman 1983–1987; consultant 1988–1997). The views expressed herein are those of the author, and do not necessarily reflect the views of the EPA.

2 Particulate Matter

In Europe, and elsewhere in the eastern hemisphere, particulate pollution has generally been measured as black smoke (BS) in terms of the optical density of stain caused by particles collected on a filter disc. However, it has been expressed in gravimetric terms ($\mu g\,m^{-3}$) based on standardized calibration factors. By contrast, US standards have specified direct gravimetric analyses of filter samples collected by a reference sampler built to match specific physical dimensions or performance criteria.

While justifications for the specific measurement techniques that have been used have generally been based on demonstrated significant quantitative associations between the measured quantity and human mortality, morbidity, or lung function differences, it is fair to say that we still lack established biological

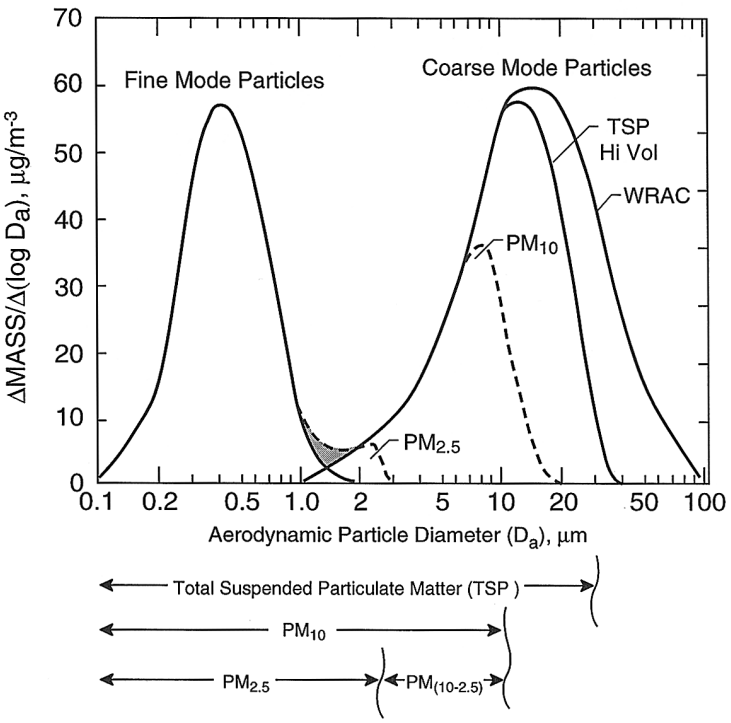


Figure 1 Representative example of a mass distribution of ambient PM as function of aerodynamic particle diameter. A wide ranging aerosol classifier (WRAC) provides an estimate of the full coarse mode distribution. Inlet restrictions of the TSP high volume sampler, the PM_{10} sampler, and the $PM_{2.5}$ sampler reduce the integral mass reaching the sampling filter (Adapted from ref. 5)

mechanisms that could account for these associations, and that we have too little information on the relative toxicities of the myriad specific constituents of airborne PM. In addition to chemical composition, airborne PM also varies in particle size distribution, which affects the number of particles that reach target sites as well as the particle surface area. To date, there are no NAAQS for PM constituents (other than lead) or for number or surface concentrations.

A broad variety of processes produce suspended particulate matter (PM) in the ambient air in which we live and breathe, and there is an extensive body of epidemiological literature that demonstrates that there are statistically significant associations between the concentrations of airborne PM and the rates of mortality and morbidity in human populations. The PM concentrations have almost always been expressed in terms of mass, although one recent study indicates that number concentration may correlate better with effects than does fine particle mass.[1] In those studies that reported on associations between health effects and more than one mass concentration, the strength of the association generally improves as one goes from total suspended particulate matter (TSP) to thoracic particulate matter, *i.e.*, PM less than 10 μm in aerodynamic diameter (PM_{10}), to fine particulate matter, *i.e.*, PM less than 2.5 μm in aerodynamic diameter ($PM_{2.5}$). The influence of a sampling system inlet on the sample mass collected is illustrated in Figure 1.

[1] A. Peters, E. Wichmann, T. Tuch, J. Heinrich, and J. Heyder, *Am. J. Respir. Crit. Care Med.*, 1997, in press.

The $PM_{2.5}$ distinction, while nominally based on particle size, is in reality a means of measuring the total gravimetric concentration of several specific chemically distinctive classes of particles that are emitted into or formed within the ambient air as very small particles. In the former category (emitted) are carbonaceous particles in wood smoke and diesel engine exhaust. In the latter category (formed) are carbonaceous particles formed during the photochemical reaction sequence that also leads to ozone formation, as well as the sulfur and nitrogen oxide particles resulting from the oxidation of sulfur dioxide and nitrogen oxide vapors released during fuel combustion and their reaction products.

The coarse particle fraction, *i.e.*, those particles with aerodynamic diameters larger than $\sim 2.5\,\mu$m, are largely composed of soil and mineral ash that are mechanically dispersed into the air. Both the fine and coarse fractions are complex mixtures in a chemical sense. To the extent that they are in equilibrium in the ambient air, it is a dynamic equilibrium in which they enter the air at about the same rate as they are removed. In dry weather, the concentrations of coarse particles are balanced between dispersion into the air, mixing with air masses, and gravitational fallout, while the concentrations of fine particles are determined by rates of formation, rates of chemical transformation, and meteorological factors. PM concentrations of both fine and coarse PM are effectively depleted by rainout and washout associated with rain. Further elaboration of these distinctions is provided in Table 2.

In the absence of any detailed understanding of the specific chemical components responsible for the health effects associated with exposures to ambient PM, and in the presence of a large and consistent body of epidemiological evidence associating ambient air PM with mortality and morbidity that cannot be explained by potential confounders such as other pollutants, aeroallergens, or ambient temperature or humidity, the EPA has established standards based on mass concentrations within certain prescribed size fractions (see Table 1).

As indicated in Table 2, fine and coarse particles generally have distinct sources and formation mechanisms, although there may be some overlap. Primary fine particles are formed from condensation of high temperature vapors during combustion. Secondary fine particles are usually formed from gases in three ways: (1) nucleation (*i.e.*, gas molecules coming together to form a new particle); (2) condensation of gases onto existing particles; and (3) by reaction of absorbed gases in liquid droplets. Particles formed from nucleation also coagulate to form relatively larger aggregate particles or droplets with diameters between 0.1 and $1.0\,\mu$m, and such particles normally do not grow into the coarse mode. Particles form as a result of chemical reaction of gases in the atmosphere that lead to products that either have a low enough vapor pressure to form a particle, or react further to form a low vapor pressure substance. Some examples include: (1) the conversion of sulfur dioxide (SO_2) to sulfuric acid droplets (H_2SO_4); (2) reactions of H_2SO_4 with ammonia (NH_3) to form ammonium hydrogensulfate (NH_4HSO_4) and ammonium sulfate [$(NH_4)_2SO_4$]; (3) the conversion of nitrogen dioxide (NO_2) to nitric acid vapor (HNO_3), which reacts further with NH_3 to form particulate ammonium nitrate (NH_4NO_3). Although some directly emitted particles are found in the fine fraction, particles formed secondarily from gases dominate the fine fraction mass.

Table 2 Comparisons of ambient fine and coarse mode particles

	Fine mode	*Coarse mode*
Formed from	Gases	Large solids/droplets
Formed by	Chemical reaction; nucleation; condensation; coagulation; evaporation of fog and cloud droplets in which gases have dissolved and reacted	Mechanical disruption (*e.g.*, crushing, grinding, abrasion of surfaces); evaporation of sprays; suspension of dusts
Composed of	Sulfate, SO_4^{2-}; nitrate, NO_3^-; ammonium, NH_4^+; hydrogen ion, H^+; elemental carbon; organic compounds (*e.g.*, PAHs, PNAs); metals (*e.g.*, Pb, Cd, V, Ni, Cu, Zn, Mn, Fe); particle-bound water	Resuspended dusts (*e.g.*, soil dust, street dust); coal and oil fly ash; metal oxides of crustal elements (Si, Al, Ti, Fe); $CaCO_3$, NaCl, sea salt; pollen, mold spores; plant/animal fragments; tire wear debris
Solubility	Largely soluble, hygroscopic, and deliquescent	Largely insoluble and non-hygroscopic
Sources	Combustion of coal, oil, gasoline, diesel, wood; atmospheric transformation products of NO_x, SO_2, and organic compounds including biogenic species (*e.g.*, terpenes); high temperature processes, smelters, steel mils, *etc.*	Resuspension of industrial dust and soil tracked onto roads; suspension from disturbed soil (*e.g.*, farming, mining, unpaved roads); biological sources; construction and demolition; coal and oil combustion; ocean spray
Lifetimes	Days to weeks	Minutes to hours
Travel distance	100s to 1000s of kilometers	<1 to 10s of kilometers

Source: EPA.

By contrast, most of the coarse fraction particles are emitted directly as particles, and result from mechanical disruption such as crushing, grinding, evaporation of sprays, or suspensions of dust from construction and agricultural operations. Basically, most coarse particles are formed by breaking up bigger masses into smaller ones. Energy considerations normally limit coarse particle sizes to greater than 1.0 μm in diameter. Some combustion-generated mineral particles, such as fly ash, are also found in the coarse fraction. Biological material such as bacteria, pollen, and spores may also be found in the coarse mode. As a result of the fundamentally different chemical compositions and sources of fine and coarse fraction particles, the chemical composition of the sum of these two fractions, PM_{10}, is more heterogenous than either mode alone.

Figure 2 presents a synthesis of the available published data on the chemical composition of $PM_{2.5}$ and coarse fraction particles in US cities by region. Each

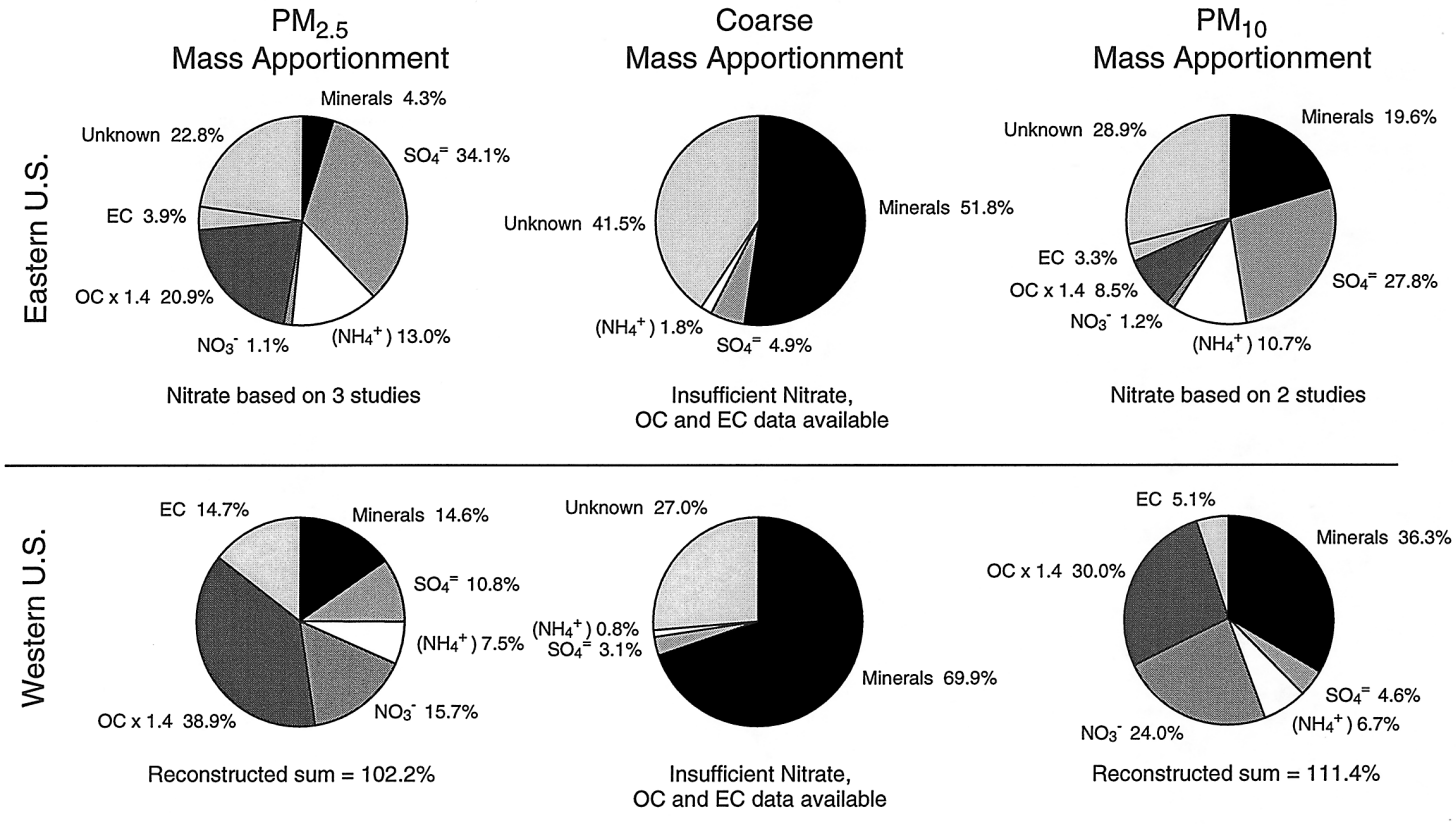


Figure 2 Representative distributions of major components of sampled PM in the fine fraction ($PM_{2.5}$), the coarse component of PM_{10}, and PM_{10} as a whole, respectively, in the eastern US (top) and western US (bottom) (Adapted from ref. 5)

fraction also has regional patterns resulting from the differences in sources and atmospheric conditions. In addition to the larger relative shares of crustal materials in the west, total concentrations of coarse fraction particles are generally higher in the arid areas of the western and south-western US.

In general, fine and coarse particles exhibit different degrees of solubility and acidity. With the exception of carbon and some organic compounds, fine particle mass is largely soluble in water and hygroscopic (*i.e.*, fine particles readily take up and retain water). Except under fog conditions, the fine particle mode also contains almost all of the strong acid. By contrast, coarse mineral particles are mostly insoluble, non-hygroscopic, and generally basic.

Fine and coarse particles typically exhibit different behavior in the atmosphere. These differences affect several exposure considerations, including the representativeness of central-site monitored values and the behavior of particles that were formed outdoors after they penetrate into homes and buildings where people spend most of their time.

Fine accumulation mode particles typically have longer atmospheric lifetimes (*i.e.*, days to weeks) than coarse particles, and tend to be more uniformly dispersed across an urban area or large geographic region, especially in the eastern US. Atmospheric transformations can take place locally, during atmospheric stagnation, or during transport over long distances. For example, the formation of sulfates from SO_2 emitted by power plants with tall stacks can occur over distances exceeding 300 kilometers and 12 hours of transport time; therefore, the resulting particles are well mixed in the air shed. Once formed, the very low dry deposition velocities of fine particles contribute to their persistence and uniformity throughout an air mass.

Larger particles generally deposit more rapidly than small particles; as a result, total coarse particle mass will be less uniform in concentration across a region than are fine particles. The larger coarse particles ($> 10\,\mu$m) tend to rapidly fall out of the air and have atmospheric lifetimes of only minutes to hours, depending on their size, wind velocity, and other factors. Their spatial impact is typically limited by a tendency to fall out in the nearby downwind area. The atmospheric behavior of the smaller particles within the 'coarse fraction' ($PM_{10-2.5}$) is intermediate between that of the larger coarse particles and fine particles. Thus, some of the smaller coarse fraction particles may have lifetimes on the order of days and travel distances of up to 100 km or more. In some locations, source distribution and meteorology affects the relative homogeneity of fine and coarse particles, and in some cases the greater measurement error in estimating coarse fraction mass precludes clear conclusions about relative homogeneity.

Up until the mid-1980s, available PM concentrations in the US were generally measured as TSP. Because TSP includes, and can be dominated by, particles too large to penetrate into the thorax, it is a poor index of inhalation hazard. Since the dispersion of large particles is limited, proximity of the sampler to local sources of dust has a major influence on measured TSP concentrations. The artifacts also vary with season and climate, and can be especially severe in the arid portions of the western US.

Despite the inherent limitations of (1) the assumption of equivalent toxicity of all sampled particles, and (2) the sampling and analytical artifacts that limit the

accuracy and precision of measured PM concentrations, there is a substantial body of epidemiological evidence for statistically significant associations between airborne PM concentrations and excess mortality and morbidity. Furthermore, the mortality and morbidity effects appear to be coherent and not explicable on the basis of known potential confounding factors or co-existing gas phase pollutants.[2,3]

Review of the Health Effects Literature that Influenced the New PM NAAQS

During the 1990s, there was a great increase in the number of peer-reviewed papers describing time-series studies of the associations between daily ambient air pollutant concentrations and daily rates of mortality and hospital admissions for respiratory diseases. Also, results of two prospective cohort studies of annual mortality rates were published. In terms of morbidity, there has been a rapid growth of the literature showing associations between airborne particle concentrations and exacerbation of asthma, increased symptom rates, decreased respiratory function, and restricted activities.

Table 3 shows an analysis of acute mortality studies in nine communities with measured PM_{10} concentrations.[4] As indicated in this table, the coefficients of response tend to be higher when the PM_{10} is expressed as a multiple-day average concentration, and lower when other air pollutants are included in multiple-regression analyses. In any case, the results in each city (except for the very small city of Kingston, TN) indicate a statistically significant association. It is also clear from recent research that the associations between PM_{10} and daily mortality are not seriously confounded by weather variables or the presence of other criteria pollutants.[5] Figure 3 shows that the calculated relative acute mortality risks for PM_{10} are relatively insensitive to the concentrations of SO_2, NO_2, CO, and O_3. The results are also coherent as described by Bates.[2] Figure 4 shows that the relative risks (RRs) for respiratory mortality are greater than for total mortality and hospital admissions.

While there is mounting evidence that excess daily mortality is associated with short-term peaks in PM_{10} pollution, the public health implications of this evidence are not yet fully clear. Key questions remain, including:

- Which specific components of the fine particle fraction ($PM_{2.5}$ and coarse particle fraction of PM_{10} are most influential in producing the responses?
- Do the effects of the PM_{10} depend on co-exposure to irritant vapors, such as ozone, sulfur dioxide, or nitrogen oxides?
- What influences do multiple-day pollution episode exposures have on daily responses and response lags?

[2] D. V. Bates, *Environ. Res.*, 1992, **59**, 336.
[3] C. A. Pope, Jr., D. W. Dockery, and J. Schwartz, *Inhal. Toxicol.*, 1995, **7**, 1.
[4] G. D. Thurston, personal communication of table prepared for draft EPA Criteria Document on Particulate Matter, 1995.
[5] US EPA, *Air Quality Criteria for Particulate Matter*, EPA/600/P-95/001F, US Environmental Protection Agency, Washington, DC, 1996.

Table 3 Comparison of time-series study estimates of total mortality relative risk (RR) for a 100 $\mu g\,m^{-3}$ PM_{10} increase

Study area (*reference*)	*Measured PM_{10} concentrations*		*RR for* 100 $\mu g\,m^{-3}$	*95% CI for* 100 $\mu g\,m^{-3}$
	Mean ($\mu g\,m^{-3}$)	*Maximum* ($\mu g\,m^{-3}$)		
1. Utah Valley, UT (Pope *et al.*, 1992)	47	297	1.16[a,d]	(1.10–1.22)
2. St. Louis, MO (Dockery *et al.*, 1992)	28	97	1.16[a,c]	(1.01–1.33)
3. Kingston, TN (Dockery *et al.*, 1992)	30	67	1.17[a,c]	(0.88–1.57)
4. Birmingham, AL (Schwartz, 1993)	48	163	1.11[a,d]	(1.02–1.20)
5. Athens, Greece (Touloumi *et al.*, 1994)	78	306	1.07[a,c] 1.03[b,c]	(1.05–1.09) (1.00–1.06)
6. Toronto, Canada (Özkaynak *et al.*, 1994)	40	96	1.07[a,c] 1.05[b,c]	(1.05–1.09) (1.03–1.07)
7. Los Angeles, CA (Kinney *et al.*, 1995)	58	177	1.05[a,c] 1.04[b,c]	(1.00–1.11) (0.98–1.09)
8. Chicago, IL (Ito *et al.*, 1995)	38	128	1.05[b,c]	(1.01–1.10)
9. Santiago, Chile (Ostro *et al.*, 1995)	115	367	1.08[a,c] 1.15[a,d]	(1.06–1.12) (1.08–1.22)

[a]Single pollutant model (*i.e.* PM_{10}). [b]Multiple pollutant model (*i.e.* PM_{10} and other pollutants simultaneously). [c]One-day mean PM_{10} concentration employed. [d]Multiple-day mean PM_{10} concentration employed.
From M. Lippmann, in *Aerosol Inhalation*: *Recent Research Frontiers*, ed. J.C.M. Marijnissen and L. Gradon, Kluwer Academic Publishers, 1996, pp. 1–25.

- Does long-term chronic exposure predispose sensitive individuals being 'harvested' on peak pollution days?
- How much of the excess daily mortality is associated with life-shortening measured in days or weeks *vs.* months, years, or decades?

The last question above is a critical one in terms of the public health impact of excess daily mortality. If, in fact, the bulk of the excess daily mortality were due to 'harvesting' of terminally ill people who would have died within a few days, then the public health impact would be much less than if it led to prompt mortality among acutely ill persons who, if they did not die then, would have recovered and lived productive lives for years or decades longer. An indirect answer to this question is provided by the results of two relatively recent prospective cohort studies of annual mortality rates in relation to long-term pollutant exposures.

Dockery *et al.*[6] reported on a 14- to 16-year mortality follow-up of 8111 adults in six US cities in relation to average ambient air concentrations of TSP $PM_{2.5}$, fine particle SO_4^{2-}, O_3, SO_2, and NO_2. Concentration data for most of these pollutant variables were available for 14–16 years. The mortality rates were adjusted for cigarette smoking, education, body mass index, and other influential

[6] D.W. Dockery, C.A. Pope III, X. Xu, J.D. Spengler, J.H. Ware, M.E. Fay, B.G. Ferris, Jr., and F.E. Speizer, *New Engl. J. Med.*, 1993, **329**, 1753.

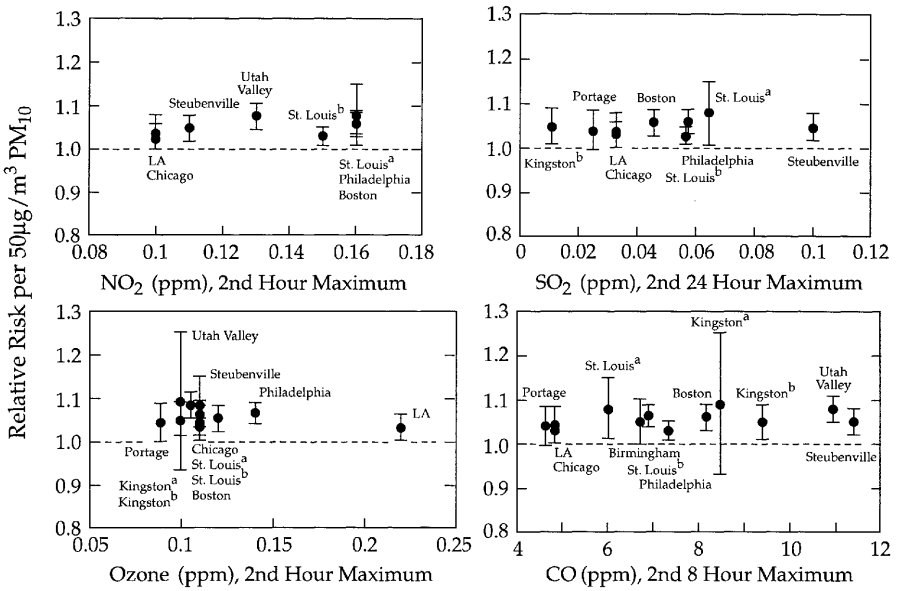


Figure 3 Relationship between RR associated with PM_{10} and peak daily levels of other criteria pollutants (Adapted from ref. 5)

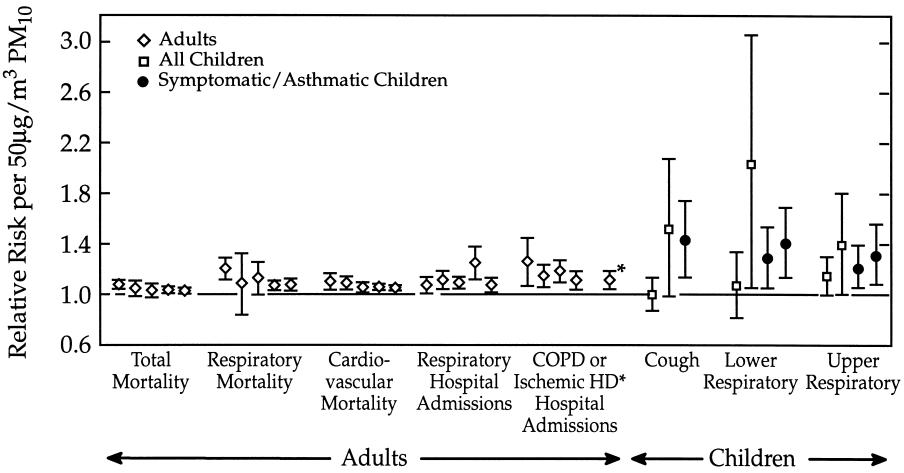
factors not associated with pollution. The two pollutant variables that best correlated with total mortality (which was mostly attributable to cardiopulmonary mortality) were $PM_{2.5}$ and SO_4^{2-}. The overall mortality rate ratios were expressed in terms of the range of air pollutant concentrations in the six cities. The rate-ratios (and 95% confidence intervals) for both $PM_{2.5}$ and SO_4^{2-} were 1.26 (1.08–1.47) overall and 1.37 (1.11–1.68) for cardiopulmonary. The mean life-shortening was in the range of 2–3 years.

Pope *et al.*[7] linked SO_4^{2-} data from 151 US metropolitan areas in 1980 with individual risk factors on 552 138 adults who resided in these areas when enrolled in a prospective study in 1982, as well as $PM_{2.5}$ data for 295 223 adults in 50 communities. Deaths were ascertained through December 1989. The relationships of air pollution to all-cause, lung cancer, and cardiopulmonary mortality were examined using multivariate analysis which controlled for smoking, education, and other risk factors. Particulate air pollution was associated with cardiopulmonary and lung cancer mortality, but not with mortality due to other causes. Adjusted relative risk ratios (and 95% confidence intervals) of all-cause mortality for the most polluted areas compared with the least polluted equaled 1.15 (1.09–1.22) and 1.17 (1.09–1.26) when using SO_4^{2-} and $PM_{2.5}$, respectively. The mean life-shortening in this study was between 1.5 and 2 years. The results were similar to those found in the previous cross-sectional studies of Özkaynak and Thurston[8] and Lave and Seskin.[9] Thus, the results of these earlier studies provide some confirmatory support for the findings of Pope *et al.*,[7] whose results indicate that the concerns about the credibility of the earlier results, due to their inability

[7] C. A. Pope, III, M. J. Thun, M. Namboodiri, D. W. Dockery, J. S. Evans, F. E. Speizer, and C. W. Heath, Jr., *Am. J. Respir. Crit. Care Med.*, 1995, **151**, 669.

[8] H. Özkaynak and G. D. Thurston, *Risk Anal.*, 1987, **7**, 449.

[9] L. B. Lave and E. P. Seskin, *Science*, 1970, **169**, 723.



Total, Respiratory, Cardiovascular Mortality
1. Pope et al. (1992) 2. Schwartz (1993) 3. Styer et al. (1995) 4. Ostro et al. (1995a)
5. Ito and Thurston (1996)

Respiratory Hospital Admissions
1. Schwartz (1995) New Haven, CT 2. Schwartz (1995) Tacoma,WA 3. Schwartz (1996) Spokane, WA
4. Ito and Thurston (1994) Toronto, Canada

COPD or Ischemic HD* Hospital Admissions
1. Schwartz (1994f) Minneapolis, MN 2. Schwartz (1994c) Birmingham, AL
3. Schwartz (1996) Spokane, WA 4. Schwartz (1994d), Detroit, MI
*5. Schwartz & Morris (1995), Detroit, MI, Ischemic HD

Cough, Lower Respiratory, Upper Respiratory
1. Hoek and Brunekreef (1993) 2. Styer et al. (1994) 3. Pope & Dockery (1992), symptomatic children

Figure 4 Relationships between relative risks per 50 $\mu g\,m^{-3}$ PM_{10} and health effects (Adapted from ref. 5)

to control for potentially confounding personal factors such as smoking and socioeconomic variables, can be eased.

The Dockery *et al.*[6] study had the added strength of data on multiple PM metrics. As shown in Figure 5, the association becomes stronger as the PM metric shifts from TSP to PM_{10}. Within the thoracic fraction (PM_{10}), the association is much stronger to the fine particle component ($PM_{2.5}$) than for the coarse component. Within the $PM_{2.5}$ fraction, both the SO_4^{2-} and non-SO_4^{2-} fractions correlate very strongly with annual mortality, suggesting a non-specific response to fine particles.

If, in fact, more people are dying of cardiopulmonary causes on a given day because of exposures to elevated concentrations of PM, it would be reasonable to expect higher daily rates of emergency hospital admissions and visits to emergency rooms and clinics for similar causes. This expectation is consistent with the results summarized in Figure 4. These studies indicate that indices of PM, such as daily concentrations of PM_{10}, SO_4^{2-}, and BS, are generally significantly associated with excess daily emergency admissions to hospitals for either respiratory diseases or cardiac diseases, or both. These studies have not shown associations with non-cardiopulmonary causes, and the influence of PM has generally been found to remain in multiple regression analyses that included other criteria pollutants. However, for respiratory diseases, the influence of

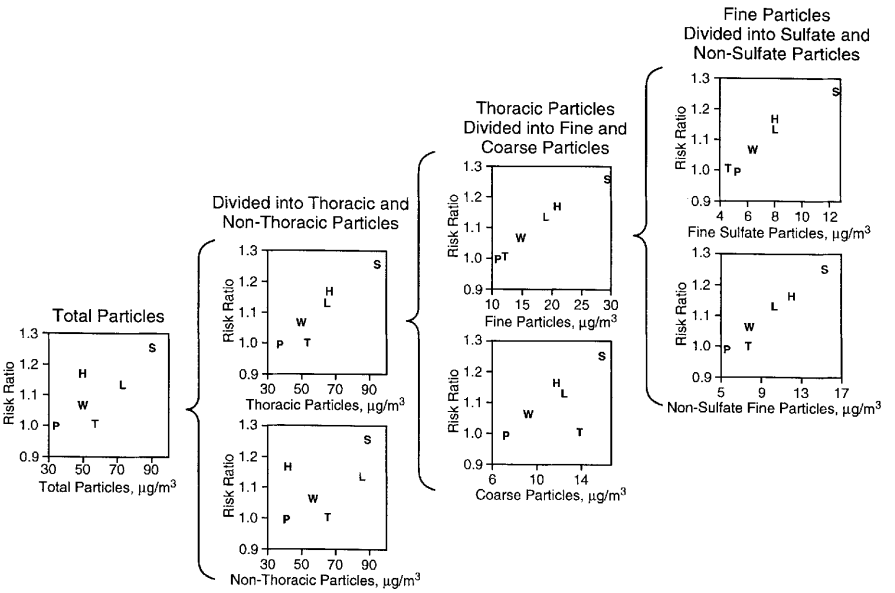


Figure 5 Adjusted relative risks for annual mortality are plotted against each of seven long-term average particle indices in the Six City Study, from largest size range [total suspended particulate matter (lower left), through sulfate and non-sulfate fine particle concentrations (upper right)]. Note that a relatively strong linear relationship is seen for fine particles, and for its sulfate and non-sulfate components. Topeka (T), which has a substantial coarse particle component of thoracic particle mass, stands apart from the linear relationship between relative risk and thoracic particle concentration (Adapted from ref. 5)

summertime O_3 has generally been greater than that of PM. This is in contradistinction to excess daily mortality, where the influence of PM is generally much greater than that of O_3. For hospitalizations for cardiac diseases, the most influential criteria pollutants appear to be PM and CO.

The importance of the fine particles as a risk factor for subnormal vital capacity in children is illustrated in Figure 6, which shows data collected in the Harvard–Health Canada cross-sectional study of 22 US and Canadian communities.[10] There was a significant association between the percentage of children with forced vital capacity (FVC) < 85% of predicted and fine particle mass concentration, but no apparent association with the coarse component of PM_{10}. Actually, the strongest association observed in this comparison was for the H^+ component of the fine particles. Most of the recent epidemiological studies have not had the advantage of available $PM_{2.5}$, SO_4^{2-}, or H^+ data, and have had to rely on PM_{10} data. Summaries of such PM_{10} epidemiology are shown in Figure 4. There is coherence in the data, as defined by Bates,[2] in terms of the relative risk ratings, with mortality risks increasing from total to cardiovascular to respiratory, and with cough and respiratory conditions being more frequent than mortality.

The results of the carefully controlled prospective cohort studies[6,7] have been analyzed by the US EPA[11] and by Brunekreef,[12] and their analyses indicated that mean lifespan shortening is more than one year, and that the individuals whose lives are shortened are losing about 14 years. An important conclusion that can be drawn from these analyses is that there is considerably more excess

[10] M. Raizenne, L. M. Ware, and F. E. Speizer, *Environ. Health Perspect.*, 1996, **104**, 506.
[11] US EPA, *The Benefits and Costs of the Clean Air Act 1970 to 1990*, US Environmental Protection Agency, Washington, DC, 1997.
[12] B. Brunekreef, *Occup. Environ. Med.*, 1997, **781**.

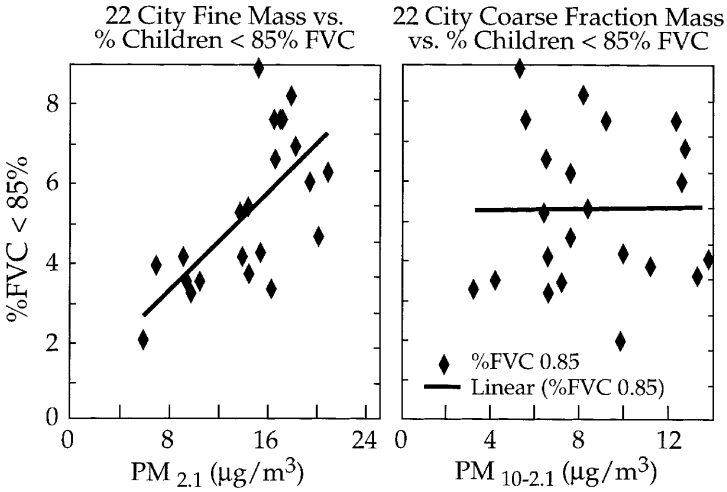


Figure 6 Plot appearing in PM Staff Paper[5] (Based on data reported by Raizenne *et al.*[10])

annual mortality associated with chronic fine particle exposure than from the cumulative impacts of daily peaks in exposure on daily mortality.

In the absence of any generally accepted mechanistic basis to account for the epidemiological associations between ambient fine particles on the one hand, and mortality, morbidity, and functional effects on the other, the causal role of PM remains questionable. However, essentially all attempts to discredit the associations on the basis of the effects being due to other environmental variables that may co-vary with PM have been unsuccessful. As shown in Figure 3, the relative risk for daily mortality in relation to PM_{10} is remarkably consistent across communities that vary considerably in their peak concentrations of other criteria air pollutants. The possible confounding influence of adjustments to models to account for weather variables has also been found to be minimal.[13,14]

While mechanistic understanding of processes by which ambient air PM causes human health effects remains quite limited, the credibility of ambient $PM_{2.5}$ as a cause of excess human mortality and morbidity has been enhanced by a series of animal inhalation studies in which rats were exposed to concentrated ambient accumulation mode PM. Godleski *et al.*[15] exposed both healthy and compromised rats to both filtered ambient air and to Boston winter ambient accumulation mode PM that was concentrated $\sim 25 \times$ (*i.e.*, $\sim 250\,\mu g\,m^{-3}$) for 6 h/day for 3 days. For two groups of compromised rats, one with SO_2-induced chronic bronchitis and one with monocrotaline-induced pulmonary hypertension, there was excess mortality during and/or immediately following these exposures. None of the healthy rats died and there was no lung inflammation and only minimal bronchoconstriction following exposure. The hypertensive rats had 19% mortality and evidence of acute inflammation in alveoli and lung interstitium. The bronchitic rats had 37% mortality and responses marked by

[13] J. M. Samet, S. L. Zeger, J. E. Kelsall, J. Xu, and L. S. Kalkstein, *Report on Phase 1.B of the Particle Epidemiology Project*, Health Effects Institute, Cambridge, MA, 1997.

[14] C. A. Pope, III and L. S. Kalkstein, *Environ. Health Perspect.*, 1996, **104**, 414.

[15] J. J. Godleski, C. Sioutas, M. Katler, and P. Koutrakis, *Am. J. Respir. Crit. Care Med.*, 1996, **153**, A15.

airway inflammation, increased mucus, marked bronchoconstriction, interstitial edema, and pulmonary vascular congestion. Compromised rats exposed to filtered ambient air had no comparable responses. Since the animals exposed to filtered air were still exposed to the same pollutant gases (O_3, SO_2, NO_2, CO), the effects are linked to the $PM_{2.5}$. Coarse particles were removed at the inlet to the concentrator, and ultrafine particles ($< 0.15\,\mu$m) were not concentrated.

In summary, excess daily mortality and morbidity have been related to ambient pollution at current levels in many communities in the US and around the world using available pollutant concentration data. However, it is not yet clear whether any of the pollutant indices used are causally related to the health effects or, if none of them are, which is the best index or surrogate measure of the causal factor(s). This gap can best be addressed by analyses of pollutant associations with mortality and morbidity in locations where a number of different pollutant metrics are available simultaneously, using analytic methods not dependent on arbitrary model assumptions.

PM Exposure Guidelines and Standards

While more research is needed on causal factors for the excess mortality and morbidity associated with PM in ambient air, and on the characterization of susceptibility factors, responsible public health authorities cannot wait for the completion and peer review of this research. It is already clear that the evidence for adverse health effects attributable to PM challenges the conventional paradigm used for setting ambient air standards and guidelines, *i.e.*, that a threshold for adversity can be identified, and a margin of safety can be applied. Excess mortality is clearly an adverse effect, and the epidemiological evidence is consistent with a linear non-threshold response for the population as a whole.

A revision of the Air Quality Guidelines of the World Health Organization-Europe (WHO-EURO) is currently nearing completion. The Working Group of WHO-EURO on PM, at meetings in October 1994 and October 1996 in Bilthoven, The Netherlands, determined that it could not recommend a PM Guideline. Instead, it prepared a tabular presentation of the estimated changes in daily average PM concentrations needed to produce specific percentage changes in: (1) daily mortality; (2) hospital admissions for respiratory conditions; (3) bronchodilator use among asthmatics; (4) symptom exacerbation among asthmatics; and (5) peak expiratory flow. The concentrations needed to produce these changes were expressed in PM_{10} for all five response categories. For mortality and hospital admissions, they were also expressed in terms of $PM_{2.5}$ and SO_4^{2-}. Using this guidance, each national or local authority setting air quality standards can decide how much adversity is acceptable for its population. Making such a choice is indeed a challenge.

In the US, the EPA Administrator promulgated the revised PM NAAQS shown in Table 1 in July 1997 (*Fed. Regis.*, 1997, **62**, 38762–38896) in recognition of the inadequate public health protection provided by enforcement of the 1987 NAAQS for PM_{10}. For PM_{10}, the $50\,\mu g\,m^{-3}$ annual average was retained without change, and the 24-h PM_{10} of $150\,\mu g\,m^{-3}$ was *relaxed* by applying it only to the 99th% value (averaged over 3 years) rather than to the 4th highest over 3

years. These PM_{10} standards were supplemented by the creation of new $PM_{2.5}$ standards. The annual average $PM_{2.5}$ is $15\,\mu g\,m^{-3}$, and the 24 hour $PM_{2.5}$ of $65\,\mu g\,m^{-3}$ applies to the 98th% value. Implementation of the new $PM_{2.5}$ NAAQS will advance the degree of public health protection for ambient air PM, especially in the eastern US and in some large cities in the west where fine particles are major percentages of PM_{10}.

In the author's view, the new PM NAAQS are not too strict. In terms of its introduction of a more relevant index of exposure and a modest degree of greater public health protection, it represents a prudent judgment call by the Administrator. These NAAQS may not be strict enough to fully protect public health, but there remain significant knowledge gaps on both exposures and the nature and extent of the effects that made the need for more restrictive NAAQS difficult to justify. It is essential that adequate research resources be applied to filling these gaps before the next round of NAAQS revisions during the next first decade of the next century.

3 Ozone

Ozone (O_3) is the indicator for photochemical pollutants. The NAAQS is intended to prevent the health and welfare effects associated with short-term peaks in exposure and to provide protection against more cumulative damage that is suspected, but not clearly established.

Ozone is almost entirely a secondary air pollutant, formed in the atmosphere through a complex photochemical reaction sequence requiring reactive hydrocarbons, nitrogen dioxide (NO_2), and sunlight. It can only be controlled by reducing ambient air concentrations of hydrocarbons, NO_2, or both. Both NO and NO_2 are primary pollutants, known collectively as NO_x. In the atmosphere, NO is gradually converted to NO_2. Motor vehicles, one of the major categories of sources of hydrocarbons and NO_x, have been the target of control efforts, and major reductions ($> 90\%$) have been achieved in the US in hydrocarbon emissions per vehicle. However, there have been major increases in vehicle miles driven. Reductions in NO_x from motor vehicles and stationary-source combustion have been much smaller. The net reduction in exposure has been modest at best, with some reductions in areas with more stringent controls, such as California, and some increases in exposure in other parts of the US. In 1988, there were record high levels of ambient O_3 with exceedance of the former 1-hour maximum 120 ppb limit in 96 communities containing over 150 million people. Since then, there has been a gradual decrease of ambient O_3 in most of the US.

We know a great deal about O_3 chemistry and have developed highly sophisticated O_3 air quality models.[16] Unfortunately, the models, and their applications in control strategies, have clearly been inadequate in terms of community compliance with the NAAQS. We also know a great deal about some of the health effects of O_3. However, much of what we know relates to transient, apparently reversible, effects that follow acute exposures lasting from 5 minutes to 6.6 hours. These effects include changes in lung capacity, flow resistance, epithelial permeability, and reactivity to bronchoactive challenges; such effects

[16] J. H. Seinfeld, *J. Air Pollut. Control Assoc.*, 1988, **38**, 616.

can be observed within the first few hours after the start of the exposure and may persist for many hours or days after the exposure ceases. Repetitive daily exposures over several days or weeks can exacerbate and prolong these transient effects. There has been a great deal of controversy about the health significance of such effects and whether such effects are sufficiently adverse to serve as a basis for the O_3 NAAQS.[17–19]

Decrements in respiratory function such as forced vital capacity (FVC) and forced expiratory volume in the first second of a vital capacity maneuver (FEV_1) fall into the category where adversity begins at some specific level of pollutant-associated change. However, there are clear differences of opinion on what the threshold of adversity ought to be. The 1989 Staff Paper[20] included a table in which the responses were categorized as mild, moderate, severe, and incapacitating. The judgement was that mild responses are not adverse, but the other categories were. The 1996 O_3 Staff Paper[21] made numerous elaborations on these gradations, and focused them on persons with impaired respiratory symptoms, as well as on healthy people, because NAAQS are generally set to protect sensitive subgroups of the population. These more elaborate gradations were presented for both healthy persons and persons with impaired respiratory symptoms.

With respect to adversity, the 1996 Staff Paper concluded that responses listed as large or severe were clearly adverse. For responses listed as moderate, it was concluded that they could be considered adverse if there were repetitive exposures.

Review of the Health Effects Literature that Influenced the New O_3 NAAQS

Although we know a great deal about the transient effects on respiratory mechanical function following single exposures to O_3, as shown in Table 4, our current knowledge about the chronic health effects of O_3 is much less complete. The chronic effects include alterations in baseline lung function and structure. Such effects may result from cumulative damage and/or from the side effects of adaptive responses to repetitive daily or intermittent exposures.

In terms of functional effects, we know that single O_3 exposures to healthy nonsmoking young adults at concentrations in the range of 80–200 ppb produce a complex array of pulmonary responses including decreases in respiratory function and athletic performance, and increases in symptoms, airway reactivity, neutrophil content in lung lavage, and rate of mucociliary particle clearance. The respiratory function responses to O_3 in purified air in chambers that occur at concentrations of 80 or 100 ppb when the exposures involve moderate exercise over 6 h or more are illustrated in Figure 7. Comparable responses require

[17] M. Lippmann, *J. Air Pollut. Control Assoc.*, 1988, **38**, 881.

[18] M. Lippmann, in *The Handbook of Environmental Chemistry, Volume 4: Part C: Air Pollution*, ed. O. Hutzinger, Springer, Heidelberg, 1991, p. 31.

[19] M. Lippmann, *J. Exp. Anal. Environ. Epidemiol.*, 1993, **3**, 103.

[20] US EPA, *Review of the National Ambient Air Quality Standards for Ozone—Assessment of Scientific and Technical Information—OAQPS Staff Paper*, EPA-450/2-92/001, NTIS, Springfield, VA, 1989.

[21] US EPA, *Review of National Ambient Air Quality Standards for Ozone—Assessment of Scientific and Technical Information—OAQPS Staff Paper*, EPA-452/A-96-007, US EPA-OAQPS, Research Triangle Park, NC, 1996.

Table 4 Mean functional changes per part per billion ozone after moderate or heavy exercise: comparison of results from field and chamber exposure studies

Investigator *Subjects, age* (yr)	*Minute ventilation* (L)	*Exposure (exercise) time* (min)	*Ozone concentration* (ppb)	*Mean rate of functional change*		
				FVC [mL (ppb)$^{-1}$]	FEV_1 [mL (ppb)$^{-1}$]	$FEF_{25-75\%}$ [mL s^{-1} (ppb)$^{-1}$]
Folinsbee (1988) 10 M, 18–33	40	395 (300)	120[c]	−3.8	−4.5	−5.0
McDonnell (1983) 22 M, 22.3 H 3.1[a]	65	120 (60)	120[c]	−1.4	−1.3	−2.9
20 M, 23.3 H 2[a]	65	120 (60)	180[c]	−1.8	−1.6	−3.0
Kulle (1985) 20 M, 25.3 H 4.1[a]	68	120 (60)	150[c]	−0.5	−0.2	−2.1
Linn (1980) 24 M, 18–33	68	120 (60)	160[c]	−0.7	−0.6	−1.1
Spektor (1988b) 1 M, 9 F, 28–44	38.4 ± 12.3[a]	34.4 ± 9.9[a]	21–124[b]	−1.9	−1.8	−6.7
7 M, 3 F, 22–40	64.6 ± 10.0[a]	26.7 ± 8.7[a]	21–124[b]	−2.9	−3.0	−9.7
Spektor (1988a) 53 M, 38 F, 7–13	—	150–550	19–113[b]	−1.0	−1.4	−2.5
Avol (1987) 33 M, 33 F, 8–11	22	60 (60)	113[b]	−0.3	−0.3	—
Avol (1985) 46 M, 13 F, 12–15	32	60 (60)	150[c]	−0.7	−0.8	−0.7
McDonnell (1985b) 23 M, 8–11	39	150 (60)	120[c]	−0.3	−0.5	−0.6

[a]Mean ± SD. [b]Ozone concentration within ambient mixture. [c]Ozone concentration within purified air.
From M. Lippmann, in *Environmental Toxicants*, ed. M. Lippmann, Van Nostrand Reinhold, 1992, pp. 465–519.

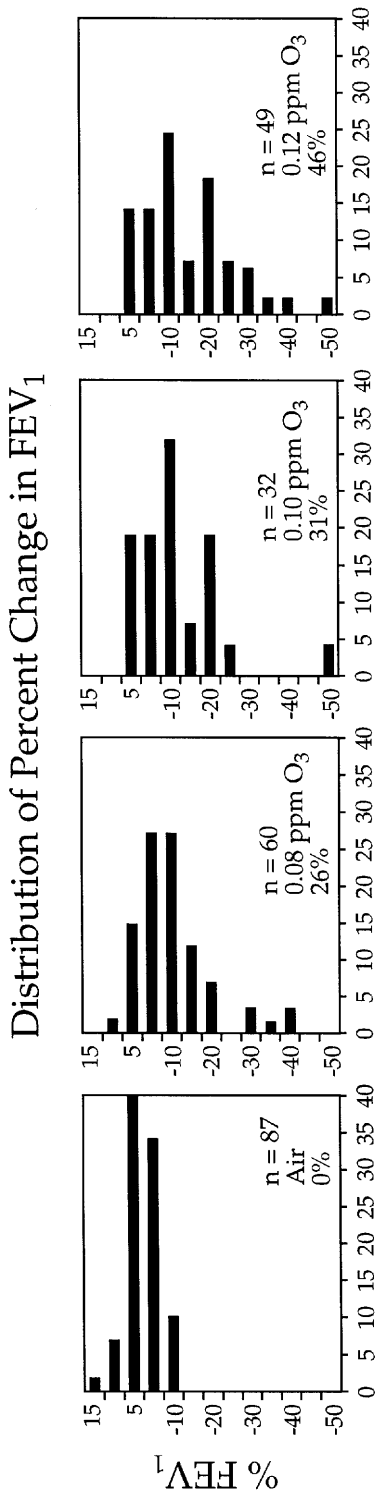


Figure 7 Percentage change in FEV_1 in healthy nonsmokers following 6.6 h exposures to clean air and O_3 at 80, 100, and 120 ppb during exercise lasting 50 minutes of each hour for studies performed at EPA's Clinical Research Laboratory at Chapel Hill, NC. Each box shows the number of subjects studied and the percentage of subjects with reductions in FEV_1 that were greater than 10% (Adapted from ref. 41)

concentrations of 180 or 200 ppb when the duration of exposure is 2 h or less. On the other hand, Table 4 shows that mean FEV_1 decrements $> 5\%$ have been seen at 100 ppb of O_3 in ambient air for children at summer camps and for adults engaged in outdoor exercise for only 0.5 h. The apparently greater responses to O_3 in ambient air may be related to the presence of, or prior exposures to, acidic aerosol.

Further research will be needed to establish the interrelationships between small transient functional decrements, such as FEV_1, PEFR, and mucociliary clearance rates, which may not in themselves be adverse effects, and changes in symptoms, performance, reactivity, permeability, and neutrophil counts. The latter may be more closely associated with adversity in themselves or in the accumulation or progression of chronic lung damage.

Successive days of exposure of adult humans in chambers to O_3 at current high ambient levels leads to a functional adaptation in that the responses are attenuated by the third day, and are negligible by the fifth day.[22,23] On the other hand, a comparable functional adaptation in rats[24] does not prevent the progressive damage to the lung epithelium. Daily exposures of animals also increase other responses in comparison to single exposures, such as a loss of cilia, a hypertrophic response of Clara cells, alterations in macrophage function, and alterations in the rates of particle clearance from the lungs.

For children exposed to O_3 in ambient air there was a week-long baseline shift in peak flow following a summer haze exposure of 4 days' duration, with daily peak O_3 concentrations ranging from 125 to 185 ppb.[25] Since higher concentrations used in adult adaptation studies in chambers did not produce such effects, it is possible that baseline shifts require the presence of other pollutants in the ambient air. A baseline shift in peak flow in camp children was also seen following a brief episode characterized by a peak O_3 concentration of 143 ppb and a peak acidic aerosol concentration of 559 nmol m^{-3}.[26]

The clearest evidence that current ambient levels of O_3 are closely associated with health effects in human populations comes from epidemiological studies focused on acute responses. The 1997 revision to the O_3 NAAQS relied heavily for its quantitative basis on a study of emergency hospital admissions for asthma in New York City[27] and its consistency with other time-series studies of hospital admissions for respiratory diseases in Toronto, all of Southern Ontario, Montreal, Detroit, and Buffalo, NY (see Table 5 and Figure 8). However, other acute responses, while less firmly established on quantitative bases, are also occurring. In order to put them in perspective, Thurston[28] prepared a graphic

[22] S. M. Horvath, J. A. Gliner, and L. J. Folinsbee, *Am. Rev. Respir. Dis.*, 1981, **123**, 496.
[23] T. J. Kulle, L. R. Sauder, H. D. Kerr, B. P. Farrell, M. S. Bermel, and D. M. Smith, *Am. Ind. Hyg. Assoc. J.*, 1982, **43**, 832.
[24] J. S. Tepper, D. L. Costa, J. R. Lehmann, M. F. Weber, and G. E. Hatch, *Am. Rev. Respir. Dis.*, 1989, **140**, 493.
[25] P. J. Lioy, T. A. Vollmuth, and M. Lippmann, *J. Air Pollut. Control Assoc.*, 1985, **35**, 1068.
[26] M. E. Raizenne, R. T. Burnett, B. Stern, C. A. Franklin, and J. D. Spenger, *Environ. Health Perspect.*, 1989, **79**, 179.
[27] G. D. Thurston, K. Ito, P. L. Kinney, and M. Lippmann, *J. Exp. Anal. Environ. Epidemiol.*, 1992, **2**, 429.
[28] G. D. Thurston, testimony submitted to US Senate Committee on Environment and Public Works, Subcommittee on Clean Air, Wetlands, Private Property, and Nuclear Safety, February 1977.

Table 5 Summary of effect estimates for ozone in recent studies of respiratory hospital admissions

Location	*Reference*	*Respiratory admission category*	*Effect size* ($\pm$ *SE*) (admissions/100 ppb O_3/ day/10^6 persons)	*Relative risk* (95% *CI*)[a] (RR of 100 ppb O_3, 1-h max)
New York City, NY[b]	Thurston *et al.*, (1992)	All	1.4 ($\pm$ 0.5)	1.14 (1.06–1.22)
Buffalo, NY[b]	Thurston *et al.*, (1992)	All	3.1 ($\pm$ 1.6)	1.25 (1.04–1.46)
Ontario, Canada[b]	Burnett *et al.*, (1994)	All	1.4 ($\pm$ 0.3)	1.10 (1.06–1.14)
Toronto, Canada[b]	Thurston *et al.*, (1994)	All	2.1 ($\pm$ 0.8)	1.36 (1.13–1.59)
Montreal, Canada[c]	Delfino *et al.*, (1994a)	All	1.4 ($\pm$ 0.5)	1.22 (1.09–1.35)
Birmingham, AL[d]	Schwartz (1994a)	Pneumonia in elderly	0.73 ($\pm$ 0.54)	1.11 (0.97–1.26)
Birmingham, AL[d]	Schwartz (1994a)	COPD in elderly	0.83 ($\pm$ 0.33)	1.13 (0.92–1.39)
Detroit, MI[d]	Schwartz (1994b)	Pneumonia in elderly	0.82 ($\pm$ 0.26)	1.22 (1.12–1.35)
Detroit, MI[d]	Schwartz (1994b)	COPD in elderly	0.90 ($\pm$ 0.41)	1.25 (1.07–1.45)
Minneapolis, MD[d]	Schwartz (1994c)	Pneumonia in elderly	0.41 ($\pm$ 0.19)	1.117 (1.03–1.39)
Minneapolis, MN[d]	Schwartz (1994c)	COPD in elderly	—[e]	—[e]

[a]One-way ($\beta \pm 1.65$ SE). [b]1-h daily maximum ozone data employed in analysis. [c]8-h daily maximum ozone data employed in analysis.
[d]24-h daily average ozone data employed in analysis (1 h/24 h av. ratio = 2.5 assumed to compute effects and RR estimates).
[e]Not reported (nonsignificant).
From *Review of National Ambient Air Quality Standards for Ozone*, OAQPS Staff Paper, EPA-452/R-96-007, US EPA, Research Triangle Park, NC, 1996.

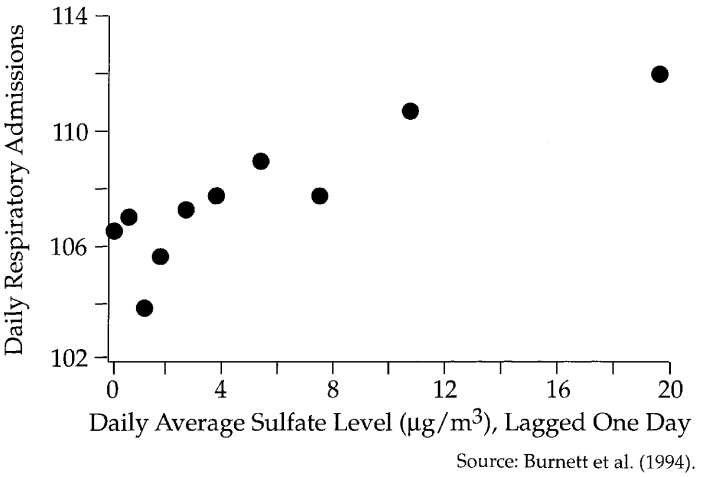


Figure 8 Average number of adjusted respiratory admissions among all 168 hospitals in Lower Ontario by decile of the daily 1-h maximum O_3 level (ppm), lagged 1 day (Adapted from ref. 41)

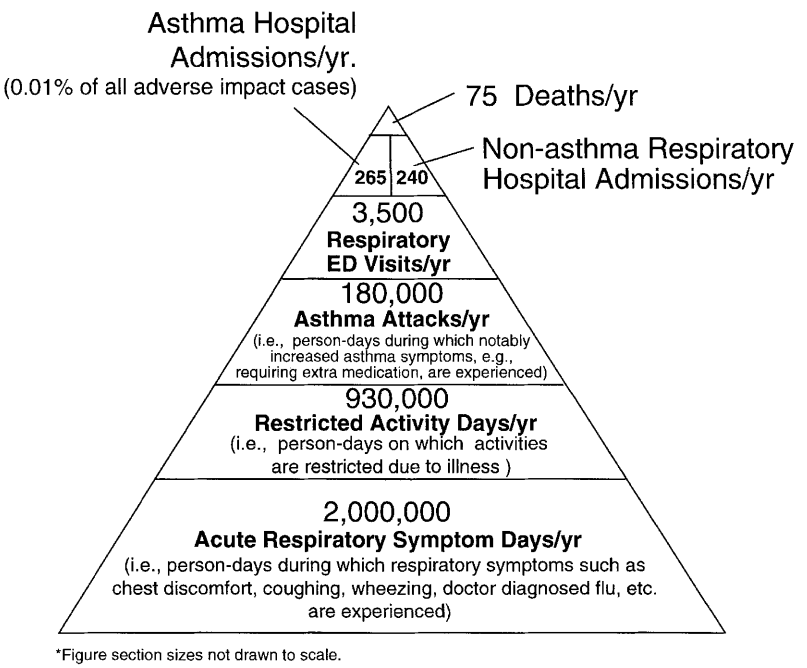


Figure 9 Pyramid summarizing the adverse effects of ambient O_3 in New York City that can be averted by reduction of mid-1990s levels to those meeting the 1997 NAAQS revision (Data assembled by Dr. G. D. Thurston for testimony to US Senate Committee on Public Works)

presentation showing the extent of related human responses based on the exposure–response relationships established in a variety of studies reviewed earlier in this chapter. This is shown, for New York City, in Figure 9. It estimates the extent of the human health responses to ambient ozone exposures in New York City that could be avoided by full implementation of the new O_3 NAAQS of 80 ppb averaged over 8 h. The extent of effects avoided on a national scale would be much larger. However, they have not been estimated, and would require knowledge of current (1997) O_3 levels and populations at risk in other parts of the country.

Chronic human exposures to ambient air appear to produce a functional

adaptation that persists for at least a few months after the end of the O_3 season but dissipates by the spring.[29] Several population-based studies of lung function[30,31] indicate that there may be an accelerated aging of the lung associated with living in communities with persistently elevated ambient O_3, but the limited ability to accurately assign exposure classifications of the various populations in these studies makes a cautious assessment of these provocative data prudent.

Some evidence for chronic effects of O_3 were reported from an analysis of pulmonary function data in a national population study in 1976–80, *i.e.*, the second National Health and Nutrition Examination Survey (NHANES II).[32] Using ambient O_3 data from nearby monitoring sites, Schwartz reported a highly significant O_3-associated reduction in lung function for people living in areas where the annual average O_3 concentrations exceeded 40 ppb.

An autopsy study of 107 lungs from 14–25 year old accident victims in Los Angeles County by Sherwin and Richters[33] reported that 27% had what were judged to be severe degrees of structural abnormalities and bronchiolitis not expected for such young subjects, while another 48% of them had similar, but less severe, abnormalities. In the absence of corresponding analyses of lungs of comparable subjects from communities having much lower levels of air pollution, the possible association of the observed abnormalities with chronic O_3 exposure remains speculative. Some of the abnormalities observed could have been due to smoking and/or drug abuse, although the authors noted that published work on the association between smoking and small airway effects showed lesser degrees of abnormality.[34]

Although the results of these epidemiological and autopsy studies are strongly suggestive of serious health effects, they have been found wanting as a basis for standards setting by EPA staff. The basis for skepticism lies in the uncertainty about the exposure characterization of the populations and the lack of control of possibly important confounding factors. Some of these limitations are inherent in large-scale epidemiological studies. Others can be addressed in more carefully focused study protocols.

The plausibility of accelerated aging of the human lung from chronic O_3 exposure is greatly enhanced by the results of chronic animal exposure studies at near ambient O_3 concentrations in rats and monkeys.[35–39] There is little reason

[29] W. S. Linn, E. L. Avol, D. A. Shamoo, R. C. Peng, L. M. Valencia, D. E. Little, and J. D. Hackney, *Toxicol. Ind. Health*, 1988, **4**, 505.

[30] R. Detels, D. P. Tashkin, J. W. Sayre, S. N. Rokaw, A. H. Coulson, F. J. Massey, and D. H. Wegman, *Chest*, 1987, **92**, 594.

[31] K. H. Kilburn, R. Warshaw, and J. C. Thornton, *Am. J. Med.*, 1985, **79**, 23.

[32] J. Schwartz, *Environ. Res.*, 1989, **50**, 309.

[33] R. P. Sherwin and V. Richters, in *Tropospheric Ozone and the Environment* (*TR-19*), ed. R. L. Berglund, D. R. Lawson, and D. J. McKee, Air & Waste Management Assoc., Pittsburgh, 1991, p. 178.

[34] M. G. Cosio, K. A. Hole, and D. E. Niewohner, *Am. Rev. Respir. Dis.*, 1980, **122**, 265.

[35] W. S. Tyler, N. K. Tyler, J. A. Last, M. J. Gillespie, and T. J. Barstow, *Toxicology*, 1988, **50**, 131.

[36] D. M. Hyde, C. G. Plopper, J. R. Harkema, J. A. St. George, W. S. Tyler, and D. L. Dungworth, in *Atmospheric Ozone Research and Its Policy Implications*, ed. T. Schneider, S. D. Lee, G. J. R. Wolters, and L. D. Grant, Elsevier, Nijmegen, The Netherlands, 1989.

[37] Y. Huang, L. Y. Chang, F. J. Miller, J. A. Graham, J. J. Ospital, and J. D. Crapo, *Am. J. Aerosol Med.*, 1988, **1**, 180.

to expect humans to be less sensitive than rats or monkeys. On the contrary, humans have a greater dosage delivered to the respiratory acinus than do rats for the same exposures. Another factor is that the rat and monkey exposures were to confined animals with little opportunity for heavy exercise. Thus humans who are active outdoors during the warmer months may have greater effective O_3 exposures than the test animals. Finally, humans are exposed to O_3 in ambient mixtures. The potentiation of the characteristic O_3 responses by other ambient air constituents seen in the short-term exposure studies in humans and animals may also contribute toward the accumulation of chronic lung damage from long-term exposures to ambient air containing O_3.

The lack of a more definitive database on the chronic effects of ambient O_3 exposures on humans is a serious failing that must be addressed with a long-term research program. The potential impacts of such exposures on public health deserve serious scrutiny and, if they turn out to be substantial, strong corrective action. Further controls on ambient O_3 exposure will be extraordinarily expensive and will need to be very well justified.

Ozone Exposure Standards

The US Occupational Safety and Health Administration's (OSHA) permissible exposure limit (PEL) for O_3 is 100 parts per billion (ppb), equivalent to 235 $\mu g\,m^{-3}$, as a time-weighted average for 8 h/day, along with a short-term exposure limit of 300 ppb for 15 min.[40] The American Conference of Governmental Industrial Hygienists[41] threshold limit value (TLV) for occupational O_3 exposure is 100 ppb as an 8-hour time-weighted average for light work, 80 ppb for moderate work, and 50 ppb for heavy work.

The initial primary (health-based) NAAQS established by the EPA in 1971 was 0.80 ppb of total oxidant as a 1-hour maximum not to be exceeded more than once per year. The NAAQS was revised in 1979 to 120 ppb of O_3 as a 1-hour maximum not to be exceeded more than four times in three years.

EPA initiated a review of the 1979 NAAQS in 1983, and completed a Criteria Document for Ozone in 1986 and updated it in 1992.[42] However, the Agency did not decide either to retain the 1979 standard or to promulgate a new one until it was compelled to do so by a August 1992 court order. In response, the EPA decided, in March 1993, to maintain the existing standard and to proceed as rapidly as possible with the next round of review. This expedited review was

[38] J. S. Tepper, M. J. Wiester, M. F. Weber, S. Fitzgerald, and D. L. Costa, *Fundam. Appl. Toxicol.*, 1991, **17**, 52.

[39] L.-Y. Chang, Y. Huang, B. L. Stockstill, J. A. Graham, E. C. Grose, M. G. Menache, F. J. Miller, D. L. Costa, and J. D. Crapo, *Toxicol. Appl. Pharmacol.*, 1992, **115**, 241.

[40] US DOL, *Fed. Regist.*, 1989, **54**, 2332.

[41] ACGIH, *Threshold Limit Values and Biological Exposure Indices for 1997*, American Conference of Governmental Industrial Hygienists, Cincinnati, 1997.

[42] US EPA, *Summary of Selected New Information on Effects of Ozone on Health and Vegetation: Supplement to 1986 Air Quality Criteria for Ozone and Other Photochemical Oxidants*, EPA/600/8-88/105F, ECAO, NTIS, Springfield, VA, 1992.

completed with the publication of both a new criteria document[43] and staff paper[21] in 1996. In July 1997 the EPA Administrator promulgated a revised primary O_3 NAAQS of 80 ppb as an 8-hour time-weighted average daily maximum, with no more than four annual exceedances, and averaged over three years (*Fed. Regist*., 1997, **62**, 38762–38896) (see Table 1). The reason for the switch from one allowable annual exceedance to four was to minimize the designation of NAAQS non-attainment in a community that was triggered by rare meteorological conditions especially conducive to O_3 formation. The goal was to have a more stable NAAQS that allowed for extremes of annual variations of weather. The switch to an 8-hour averaging time was in recognition that ambient O_3 in much of the US has broad daily peaks, and that human responses are more closely related to total daily exposure than to brief peaks of O_3 exposure. Since the 120 ppb, 1-h average, one exceedance NAAQS was approximately equivalent to an 8-h average, four exceedance NAAQS at a concentration a little below 90 ppb in average stringency in the US as a whole, the 1997 NAAQS represents about a 10% reduction in permissible O_3 exposure. While this new NAAQS will be difficult to achieve in much of the southern and eastern US, it represents a prudent public health choice considering the extent and severity of the effects that O_3 produces in sensitive segments of the population.

The effects of concern with respect to acute response in the population at large are reductions in lung function and increases in respiratory symptoms, airway reactivity, airway permeability, and airway inflammation. For persons with asthma, there are also increased rates of medication usage and restricted activities. Margin-of-safety considerations included: (1) the influence of repetitive elicitation of such responses in the progression of chronic damage to the lung of the kinds seen in chronic exposure studies in rats and monkeys; and (2) evidence from laboratory and field studies that ambient air co-pollutants potentiate the responses to O_3.

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[43] US EPA, *Air Quality Criteria for Ozone and Related Photochemical Oxidants*, EPA/600/P-93/004F, US EPA, National Center for Environmental Assessment, Research Triangle Park, NC, 1996.