American Journal of Epidemiology
Vol. 163, No. 6
DOI: 10.1093/aje/kwj078
Advance Access publication January 27, 2006

Original Contribution

The Effect of Ozone and PM_{10} on Hospital Admissions for Pneumonia and Chronic Obstructive Pulmonary Disease: A National Multicity Study

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Received for publication August 18, 2005; accepted for publication October 27, 2005.

A case-crossover study was conducted in 36 US cities to evaluate the effect of ozone and particulate matter with an aerodynamic diameter of $\leq$10 μm (PM_{10}) on respiratory hospital admissions and to identify which city characteristics may explain the heterogeneity in risk estimates. Respiratory hospital admissions and air pollution data were obtained for 1986–1999. In a meta-analysis based on the city-specific regression models, several city characteristics were evaluated as effect modifiers. During the warm season, the 2-day cumulative effect of a 5-ppb increase in ozone was a 0.27% (95% confidence interval (CI): 0.08, 0.47) increase in chronic obstructive pulmonary disease admissions and a 0.41% (95% CI: 0.26, 0.57) increase in pneumonia admissions. Similarly, a 10-$\mu g/m^3$ increase in PM_{10} during the warm season resulted in a 1.47% (95% CI: 0.93, 2.01) increase in chronic obstructive pulmonary disease at lag 1 and a 0.84% (95% CI: 0.50, 1.19) increase in pneumonia at lag 0. Percentage of households with central air conditioning reduced the effect of air pollution, and variability of summer apparent temperature reduced the effect of ozone on chronic obstructive pulmonary disease. The study confirmed, in a large sample of cities, that exposure to ozone and PM_{10} is associated with respiratory hospital admissions and provided evidence that the effect of air pollution is modified by certain city characteristics.

air pollution; effect modifiers (epidemiology); ozone; pneumonia; pulmonary disease, chronic obstructive

Abbreviations: CI, confidence interval; COPD, chronic obstructive pulmonary disease; PM_{10}, particulate matter with an aerodynamic diameter of $\leq$10 μm; $PM_{2.5}$, particulate matter with an aerodynamic diameter of $\leq$2.5 μm.

Air pollution has been associated with hospital admissions for respiratory disease in cities all over the world (1–7). The most common and consistent associations have been found with particulate matter and tropospheric ozone (4). The magnitude of the effect of these pollutants, however, has differed substantially across locations. Moreover, many studies have examined all respiratory admissions, possibly combining outcomes with different sensitivities to air pollution and different lags between exposure and hospitalization. In addition, the number of cities examined in individual studies of respiratory admissions has generally been modest, particularly for ozone, and a large national sample would avoid selection bias, especially in light of evidence for heterogeneity in results across individual cities.

Because of limited numbers of locations, few studies have addressed the issue of whether the observed variability in exposure-effect relations can be explained by differences in meteorology, pollution sources, or socioeconomic characteristics of the cities. A study conducted in 14 US cities did not find evidence of modification by sociodemographic characteristics of the association between particulate matter with an aerodynamic diameter of $\leq$10 μm (PM_{10}) and hospital admissions (8). A subsequent analysis using the same data found that the proportion of traffic-related particles in PM_{10} modified the effect of PM_{10} on

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hospital admissions for cardiovascular disease, but results were inconclusive for respiratory admissions (9). This study also found a stronger effect of PM_{10} on chronic obstructive pulmonary disease (COPD) admissions with decreasing proportion of central air conditioning, and similar results, although only marginally significant, were found for pneumonia admissions.

These studies had limited statistical power because they included a relatively small number of cities. In addition, they did not address modifiers of the association between ozone exposure and respiratory hospital admissions. We therefore conducted a large, multicity study in 36 US cities to evaluate the effect of daily PM_{10} and ozone concentrations on hospital admissions for COPD and for pneumonia and to assess whether the heterogeneity in the effect estimates across cities may be explained by differences in city characteristics such as meteorology, pollution sources, or socioeconomic factors.

MATERIALS AND METHODS

Study design

We conducted a case-crossover analysis using hospital admissions and air pollution data from 36 US cities during the period 1986–1999. We first selected cities that monitored PM_{10} daily and subsequently expanded the sample to include other large cities covering all regions of the United States. The cities selected were Albuquerque, New Mexico; Atlanta, Georgia; Baltimore, Maryland; Birmingham, Alabama; Boston, Massachusetts; Boulder, Colorado; Canton, Ohio; Chicago, Illinois; Cincinnati, Ohio; Cleveland, Ohio; Colorado Springs, Colorado; Columbus, Ohio; Denver, Colorado; Detroit, Michigan; Honolulu, Hawaii; Houston, Texas; Jersey City, New Jersey; Los Angeles, California; Minneapolis, Minnesota; Nashville, Tennessee; New Haven, Connecticut; New York City, New York; Palm Beach, Florida; Philadelphia, Pennsylvania; Pittsburgh, Pennsylvania; Provo, Utah; Sacramento, California; Salt Lake City, Utah; San Diego, California; San Francisco, California; Seattle, Washington; Steubenville, Ohio; St. Louis, Missouri; Spokane, Washington; Washington, DC; and Youngstown, Ohio.

The case-crossover design is a variant of the matched case-control design in which a case subject becomes a control subject on days when no event (hospital admission) occurs (10). By using control days close in time to the event day, there is no confounding by slowly varying personal characteristics since each subject is the perfect match for himself or herself. Bateson and Schwartz (11, 12) demonstrated that, by using such a matching scheme, even very strong seasonal confounding of exposure can be removed. If, in addition, as suggested by Levy et al. (13), we apply a time-stratified approach to choose the control days, a subtle selection bias can be avoided. A simulation study has shown that such an approach gives both unbiased estimates of effects and unbiased coverage probabilities (14). We followed this approach in our study by choosing control days only within the same month of the same year when the admission occurred (refer to the Statistical Analyses section of the text for more details on the specific selection criteria for days within that month).

Hospital admissions data

We extracted data on hospital admissions from the US Health Care Financing Administration (Medicare) billing records for the period 1986–1999. The Medicare system provides hospital coverage for all US citizens aged 65 years or older. The system includes data on type of admission, primary and secondary causes of admission, and other personal characteristics. Using this information, we selected for analyses those persons who had been admitted to the hospital on an emergency or urgent basis with a primary diagnosis of COPD (*International Classification of Diseases*, Ninth Revision: codes 490–496, except code 493) or pneumonia (*International Classification of Diseases*, Ninth Revision: codes 480–487).

Environmental data

We obtained air pollution data from the US Environmental Protection Agency's Aerometric Retrieval System (15). We estimated daily mean concentrations of ozone (8-hour) and PM_{10} (24-hour) for each city by using an algorithm that averaged levels reported by multiple monitoring locations (16). All cities except Minneapolis had daily measurements for ozone during the warm season (May–September), but only 16 had complete ozone measurements during the cold season (October–April). For all cities, PM_{10} levels were available throughout the year, although frequency of measurements varied across cities and within a city, with measurements typically made every 2, 3, or 6 days.

For each city, we obtained daily mean temperature and relative humidity from the nearest National Weather Service surface station (EarthInfo Inc., Boulder, Colorado). We used this information to calculate the apparent temperature, a composite index of perceived air temperature at a given humidity, previously used to control for weather in air pollution studies (17–20).

City characteristics

For each city, we calculated the mean and variance of the daily summer (June–August) apparent temperature by using the data described above. We calculated the percentage of people aged 65 years or older living in poverty by using data from the 1990 US Census. We calculated the percentage of households with central air conditioning by using data from the American Housing Survey of the US Census Bureau (21) for the period 1994–2002. We calculated the average annual mortality rate for emphysema among people aged 65 years or older by using data from the National Center for Health Statistics during 1989–2000. We took the latter as an indication of the smoking history of the population. Finally, we calculated the percentage of ambient PM_{10} from traffic (highway vehicles) by using data from the National Emission Trends 1996 (NET96) of the Environmental Protection Agency (22).

Statistical analyses

In the first stage of the analyses, we analyzed the association between exposure to air pollution and hospital admissions (for COPD and for pneumonia) by using city-specific conditional logistic regression models (the PROC PHREG procedure in SAS version 8.2 software, 2001; SAS Institute, Inc., Cary, North Carolina). We fitted separate models for ozone and PM_{10} exposure. For each hospitalization, we selected control days within the same month of the same year, leaving at least 2 days between each control day to minimize serial correlation. In all analyses, we controlled for day of the week and weather—the latter by using one cubic regression spline for the same day's apparent temperature and another for the previous day's apparent temperature.

For the ozone analysis, we examined the effect of exposure on the same day (lag 0) and on the day before (lag 1) admission. For each city, we first fitted a model including lag 0 and lag 1, then calculated the cumulative risk estimate for ozone exposure by summing the estimates from lag 0 and lag 1. We computed the overall standard error (SE) of the cumulative estimate as $SE = \text{sqrt}[\text{var}(\text{lag } 0) + \text{var}(\text{lag } 1) + 2 \times \text{cov}(\text{lag } 0, \text{lag } 1)]$.

Because PM_{10} was predominantly measured during nonconsecutive days, we assessed the effect of exposure to PM_{10} at lag 0 and lag 1 in separate models. We repeated the analyses of ozone and PM_{10} by examining separate effects for exposure in the warm season and in the cold season.

As a sensitivity analysis, we tested the use of an alternative matching scheme by selecting control days within the same month of the same year of the event and matching on apparent temperature (same rounded degrees Centigrade). In this analysis, we included in all models a cubic spline for the previous day's apparent temperature and indicator variables for day of the week. These models, by matching on temperature and month, control for potentially nonlinear temperature effects and for any interaction between temperature and month of the year (18). To study the effect of ozone exposure, we additionally tested a matching scheme that selected control days within the same month of the same year of the event and matched on day of the week. In this instance, models included a cubic spline for apparent temperature on the same day and another cubic spline for the previous day's apparent temperature. Because of the irregular sampling scheme for PM_{10}, we could not follow this approach for PM_{10}.

In the second stage of the analyses, we combined the city-specific results in a meta-analysis by using restricted maximum likelihood random-effects models (REML in Stata version 8 software; Stata Corporation, College Station, Texas) (23). After estimating the overall effect of ozone and PM_{10} on hospital admissions, we assessed the potential for effect modification of several city characteristics by including them (one at a time) as covariates in the meta-regression models. Then, we used the estimated model coefficients to predict the effect of air pollution on hospital admissions at the 25th and the 75th percentiles of the distribution of each city characteristic. A significant difference between these two predicted values indicates that the city characteristic modifies the effect of air pollution on hospital admissions, that is, the existence of an interaction between air pollution and the city characteristic.

RESULTS

Our analyses included 578,006 COPD admissions and 1,384,813 pneumonia admissions. Table 1 shows the counts of hospital admissions and describes the main environmental variables for each city. Ozone levels were higher in the warm season (average across all cities, 45.8 ppb (standard deviation, 9.2)) than in the cold season (27.6 ppb (standard deviation, 6.3)). PM_{10} levels were similar during both seasons, with an average concentration across all cities of 30.4 $\mu g/m^3$ (standard deviation, 5.1). Los Angeles had the highest average levels of both ozone and PM_{10}.

Table 2 shows the results of the meta-analysis across all cities based on the city-specific models in which the baseline matching scheme was used. Overall, during the warm season, there was an increase in hospital admissions associated with ozone exposure, specifically with such exposure on the day before. The 2-day cumulative effect of a 5-ppb increase in 8-hour ozone levels was a 0.27 percent (95 percent confidence interval (CI): 0.08, 0.47) increase in COPD admissions and a 0.41 percent (95 percent CI: 0.26, 0.57) increase in pneumonia admissions. The distribution of the city-specific estimates for the warm season is shown in figure 1, with the extreme values corresponding to those estimates with larger standard errors. Results from the meta-analysis showed that hospital admissions during the cold season decreased with increasing ozone concentrations, which resulted in a weak positive association for the full-year period. Results were very similar when other matching schemes were used. The increases in COPD and pneumonia admissions during the warm season, for instance, were 0.26 percent (95 percent CI: −0.05, 0.57) and 0.32 percent (95 percent CI: 0.12, 0.52), respectively, when additionally matching on temperature, and 0.30 percent (95 percent CI: 0.10, 0.51) and 0.48 percent (95 percent CI: 0.30, 0.67), respectively, when matching on day of the week. After we restricted the analysis to those days for which PM_{10} measurements were available, adjustment for PM_{10} did not substantially modify the results (not shown).

As shown in table 2, there was also an overall increase in respiratory hospital admissions associated with PM_{10} concentrations throughout the year, with a more marked effect during the warm season. The between-city variability in the estimates for that season is presented in figure 1. Results from the meta-analysis showed that, for COPD, the larger increase during the warm season (1.47 percent for every 10-$\mu g/m^3$ increase in PM_{10}) was associated with PM_{10} exposure on the day before (lag 1); for pneumonia, the main increase (0.84 percent) occurred on the same day of exposure (lag 0). Repeating the analyses for the warm season matching also for apparent temperature led to similar results, with an increase of 1.30 percent (95 percent CI: 0.58, 2.02) in COPD admissions associated with PM_{10} at lag 1 and of 0.94 percent (95 percent CI: 0.41, 1.46) in pneumonia admissions associated with PM_{10} at lag 0. The associations remained unaltered when analyses were repeated adjusting for ozone (results not shown).

TABLE 1. Environmental variables and respiratory hospital admissions in 36 US cities during 1986–1999

City, state	Mean (SD*) ozone level (ppb)		Mean (SD) PM_{10}* level ($\mu g/m^3$)	Mean (SD) apparent temperature (°C)	Total population aged ≥65 years (no.)	COPD* admissions (no.)	Pneumonia admissions (no.)
	Warm season	Cold season					
Albuquerque, New Mexico	50.5 (9.3)	34.5 (10.2)	27.9 (16.5)	12.2 (8.9)	50,379	3,115	9,035
Atlanta, Georgia	55.9 (21.4)		33.0 (16.4)	17.1 (10.2)	155,955	15,503	36,488
Baltimore, Maryland	52.3 (20.2)	26.8 (13.0)	32.4 (17.1)	13.0 (11.1)	197,438	19,950	40,858
Birmingham, Alabama	49.7 (17.0)		36.1 (21.0)	17.4 (10.5)	119,809	13,134	33,011
Boston, Massachusetts	42.3 (17.8)	28.3 (11.3)	25.4 (11.7)	10.0 (10.3)	342,322	34,700	88,936
Boulder, Colorado	51.3 (14.2)		24.2 (15.5)	8.5 (9.7)	17,048	1,678	3,427
Canton, Ohio	52.6 (17.8)		26.1 (12.6)	9.3 (11.2)	53,216	7,534	12,965
Chicago, Illinois	40.0 (16.1)	22.7 (9.8)	33.6 (17.4)	9.5 (11.9)	631,826	49,581	142,576
Cincinnati, Ohio	50.0 (17.8)		32.2 (15.6)	11.9 (11.5)	115,000	10,797	33,323
Cleveland, Ohio	44.6 (17.6)		37.1 (19.1)	9.8 (11.3)	220,659	29,947	50,262
Colorado Springs, Colorado	45.5 (11.3)	30.4 (11.6)	23.3 (13.4)	7.8 (9.0)	31,674	2,497	5,729
Columbus, Ohio	49.8 (18.1)		30.5 (14.6)	11.1 (11.5)	92,485	12,571	21,900
Denver, Colorado	44.0 (14.0)	22.1 (12.7)	33.2 (18.8)	8.5 (9.7)	64,152	4,219	11,820
Detroit, Michigan	41.7 (17.2)		33.7 (19.7)	9.3 (11.5)	263,997	5,751	12,393
Honolulu, Hawaii	15.0 (8.4)		15.9 (6.2)	27.5 (2.9)	91,485	28,404	57,682
Houston, Texas	44.9 (22.1)	32.9 (17.1)	30.3 (16.0)	22.2 (10.1)	196,474	3,798	14,463
Jersey City, New Jersey	50.3 (23.4)		32.2 (17.0)	12.4 (11.1)	70,014	18,863	41,754
Los Angeles, California	63.0 (23.4)	31.4 (20.2)	44.0 (19.3)	16.5 (4.3)	855,666	9,211	12,645
Minneapolis, Minnesota			27.3 (14.6)	7.4 (12.5)	175,854	63,316	174,241
Nashville, Tennessee	44.9 (16.8)	23.9 (13.5)	32.2 (14.9)	15.5 (11.3)	59,235	9,805	26,923
New Haven, Connecticut	45.4 (19.5)		26.0 (16.1)	9.6 (10.8)	117,863	5,962	14,719
New York City, New York	41.0 (19.5)	19.7 (10.0)	28.9 (13.9)	12.5 (10.8)	952,731	8,082	22,954
Palm Beach, Florida	28.6 (12.7)	33.7 (12.0)	20.0 (8.1)	27.1 (6.3)	210,389	70,181	187,043
Philadelphia, Pennsylvania	47.8 (21.0)	23.0 (13.0)	32.1 (15.8)	12.9 (11.1)	241,206	10,626	22,170
Pittsburgh, Pennsylvania	48.4 (19.9)		30.3 (20.0)	10.3 (10.9)	232,505	26,604	47,126
Provo, Utah	54.6 (10.9)		35.1 (26.7)	9.6 (10.4)	18,429	33,408	52,148
Sacramento, California	55.6 (15.7)	32.7 (14.2)	31.1 (19.7)	14.4 (7.0)	109,674	718	4,081
Salt Lake City, Utah	54.0 (12.5)		35.7 (23.9)	9.6 (10.4)	61,079	8,680	21,840
San Diego, California	47.6 (12.1)	40.4 (15.2)	33.3 (13.1)	17.0 (4.4)	272,348	2,090	9,348
San Francisco, California	22.8 (8.1)	19.3 (10.2)	27.7 (16.8)	12.6 (3.8)	105,263	17,632	43,446
Seattle, Washington	35.0 (14.2)		28.8 (18.6)	9.5 (6.3)	167,328	4,711	18,139
Steubenville, Ohio	46.1 (17.3)		34.7 (19.9)	10.3 (10.9)	23,878	9,334	23,732
St. Louis, Missouri	48.4 (17.1)		27.7 (12.7)	13.7 (12.3)	214,492	4,039	9,412
Spokane, Washington	44.6 (10.4)		32.2 (28.3)	6.5 (9.0)	47,877	5,633	8,976
Washington, DC	48.4 (20.2)	20.1 (12.3)	27.7 (13.4)	14.2 (11.2)	77,672	17,665	54,386
Youngstown, Ohio	47.1 (20.3)		31.2 (15.6)	8.9 (11.0)	61,122	8,267	14,862

* SD, standard deviation; PM_{10}, particulate matter with an aerodynamic diameter of ≤10 μm; COPD, chronic obstructive pulmonary disease.

We further examined the potential for effect modification of the city characteristics presented in table 3. In general, all characteristics varied from city to city. The proportion of households with central air conditioning was especially variable, ranging from 6.2 percent in Seattle to 93.3 percent in Houston. Central air conditioning and poverty were more common in those cities with a higher summer apparent temperature (Spearman's $r = 0.74$ and $r = 0.52$, respectively) but were unrelated to the variance of summer apparent temperature ($r = 0.07$ and $r = 0.23$, respectively).

The association between hospital admissions and air pollution during the warm season was modified by some of the city characteristics (table 4). Overall, in cities with a large proportion of households with central air conditioning, the effect of air pollution on hospital admissions was milder, especially for pneumonia admissions. For instance, in a city

TABLE 2. Percentage change in respiratory hospital admissions associated with air pollution exposure across 36 US cities during 1986–1999*

	COPD†						Pneumonia					
	Lag 0		Lag 1		2-day cumulative estimates		Lag 0		Lag 1		2-day cumulative estimates	
	%	95% CI†	%	95% CI	%	95% CI	%	95% CI	%	95% CI	%	95% CI
Ozone‡	−0.32	−0.49, −0.15	0.33	0.19, 0.47	0.04	−0.13, 0.20	−0.23	−0.32, −0.13	0.21	0.11, 0.30	0.30	−0.12, 0.08
Warm season	−0.25	−0.46, −0.04	0.48	0.30, 0.66	0.27	0.08, 0.47	0.01	−0.12, 0.13	0.42	0.29, 0.55	0.41	0.26, 0.57
Cold season§	−0.48	−0.75, −0.21	0.14	−0.13, 0.42	−0.31	−0.61, −0.01	−0.61	−0.85, −0.36	−0.17	−0.34, −0.004	−0.83	−1.13, −0.53
PM_{10}¶	0.29	−0.01, 0.58	0.59	0.30, 0.88			0.45	0.27, 0.64	0.26	0.01, 0.52		
Warm season	0.81	0.22, 1.41	1.47	0.93, 2.01			0.84	0.50, 1.19	0.79	0.45, 1.13		
Cold season	0.06	−0.40, 0.51	0.10	−0.30, 0.49			0.30	0.07, 0.53	0.14	−0.17, 0.45		

* Estimates are presented as percentage increase in hospital admissions associated with a 5-ppb increase in ozone or a 10-μg/m^3 increase in particulate matter with an aerodynamic diameter of ≤10 μm (PM_{10}).
† COPD, chronic obstructive pulmonary disease; CI, confidence interval.
‡ Estimates for ozone exposure at lag 0 and lag 1 were obtained by including both variables in the same model.
§ Estimates for ozone exposure during the cold season are based on 16 cities.
¶ Estimates for PM_{10} exposure at lag 0 and lag 1 were obtained by using separate models for each lag.

with a low proportion of central air conditioning (in the 25th percentile of the distribution), the estimated percentage increase in pneumonia admissions for every 10-μg/m^3 increase in PM_{10} was 1.47 percent; whereas, in a city with a high proportion of central air conditioning (in the 75th percentile), the change in pneumonia admissions was negligible (−0.11 percent). The effect of air pollution was consistently lower for those cities with a higher summer apparent temperature, although the interaction was statistically significant for the association between PM_{10} and pneumonia admissions only and disappeared after adjusting for percentage of central air conditioning ($p = 0.96$, results not shown). Similarly, the milder effect of PM_{10} on cities with a large proportion of people living in poverty disappeared after adjusting for percentage of central air conditioning ($p = 0.38$, results not shown). The variability of summer apparent temperature modified the effect of ozone on COPD (about a fourfold decrease when the 25th percentile was compared with the 75th percentile) and remained the same after adjustment for other city characteristics. When we examined modification of the association between PM_{10} and hospital admissions for the entire year, we obtained results similar to the ones presented in table 4 but weaker associations (results not shown).

DISCUSSION

In a large, multicity study, we found effects of ozone and PM_{10} on hospital admissions for COPD and pneumonia and identified several city characteristics as effect modifiers. We found increased risks of COPD and pneumonia admissions associated with ambient ozone and PM_{10} levels, predominantly during the warm season. The proportion of households with central air conditioning was the most important effect modifier in that season. The variance of summer apparent temperature was associated with a milder effect of ozone on COPD.

Most of the literature about ozone effects on respiratory hospital admissions is based on data from single cities (6). One of the strengths of our study is that we included a large number of cities across the United States involving great diversity in weather, geography, and other characteristics that enabled us to analyze how these characteristics may modify the effect of air pollution. In addition, we analyzed more years of follow-up than previous multicity studies on the respiratory effects of ozone (3, 7, 24) and PM_{10} (25, 26). These two circumstances provided us with a larger sample size and more stable estimates than prior investigations offered.

In our study, we found that the risk of daily hospital admissions for COPD and for pneumonia increased with ozone concentration during the warm season but not during the cold season. Consistently, epidemiologic studies conducted in the United States and in other industrialized countries have also found an increased risk of both respiratory hospital admissions (3, 27) and total mortality (18, 28) associated with ozone exposure during the warm season only. Differences in the effect of ozone during these two seasons could be explained by the higher ozone concentrations and the larger amount of time spent outdoors during the warm

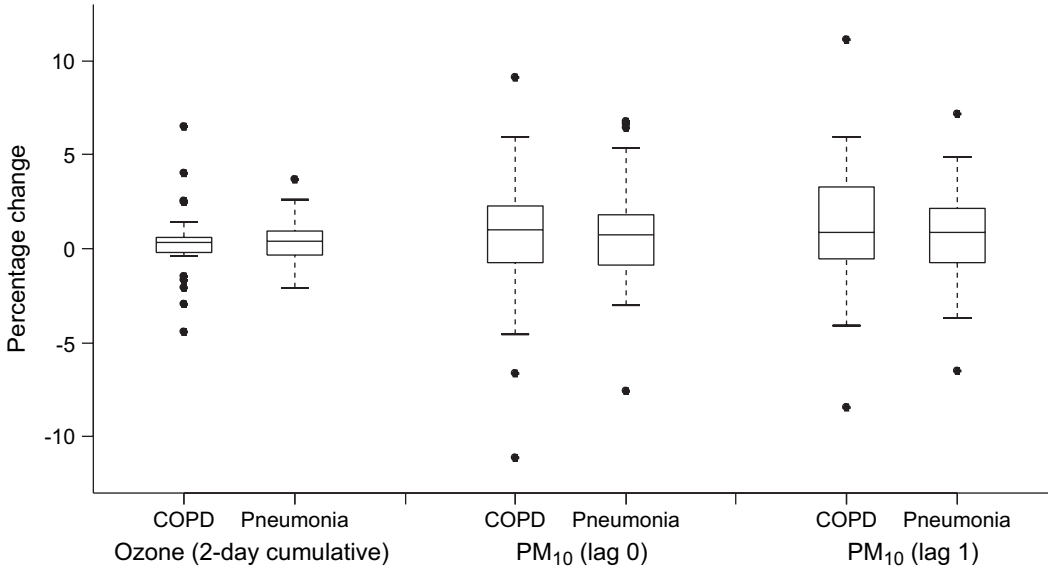

FIGURE 1. Distribution of the city-specific estimates of the percentage change in respiratory hospital admissions associated with air pollution during the warm season in 36 US cities during 1986–1999. Estimates are presented as the percentage increase in hospital admissions associated with a 5-ppb increase in ozone or a 10-$\mu g/m^3$ increase in particulate matter with an aerodynamic diameter of $\leq$10 μm (PM_{10}). COPD, chronic obstructive pulmonary disease.

season (29). In addition, ozone is a highly reactive gas whose indoor concentrations are extremely low in buildings with low ventilation (closed windows, etc.), which occurs more often in the winter. In the summer, open windows can result in some indoor exposure as well, and this exposure difference could also partially explain why positive effects are seen during the warm season only.

We also found an increased risk of COPD and pneumonia admissions associated with PM_{10} levels, which closely agrees with findings by Zanobetti et al. (2) using data from 10 US cities and also with results from the APHEA-2 study in eight European cities (26). The effect of PM_{10} on respiratory admissions was evident throughout the year but was stronger during the warm season. Because PM_{10} levels are usually similar during the warm and cold seasons, few studies have examined the effect of PM_{10} separately for these two seasons. A stronger effect during the warm season may be related to an increase in individual exposure (i.e., more time spent outdoors, higher ventilation rates, etc.) rather than to an overall increase in the outdoor ambient concentration. These results are consistent with those of Peng et al. (30), who recently reported that the effect of PM_{10} on mortality was higher in the summer. In our study, exposure to PM_{10} seemed to have an earlier effect on pneumonia admissions, occurring predominantly on the same day of exposure, than on COPD admissions, occurring predominantly 1 day after the exposure. An earlier effect of PM_{10} on pneumonia than on COPD was also observed in a mortality study of 20 US cities (31) and could be due to differences in the biologic mechanisms involved in the aggravation of these two respiratory disorders.

The effect of air pollution on respiratory hospital admissions was smaller in those cities with a larger proportion of central air conditioning. In a meta-analysis of 14 US cities, Janssen et al. (9) studied the role of air conditioning as an effect modifier of the relation between PM_{10} and hospital admissions and found a smaller effect for COPD, and possibly pneumonia, in cities with a larger proportion of central air conditioning. Several studies have shown that homes with air conditioning typically have lower exchange rates than homes that use open windows for ventilation (32, 33), suggesting a higher penetration of outdoor pollutants in homes without air conditioning. Therefore, those living in a home with central air conditioning are likely to be less exposed to outdoor air pollutants, resulting in attenuation of the association between ambient air pollution and health effects.

We observed a milder effect of ozone on COPD admissions during the warm season for those cities with a large variability in their summer temperature. Levels of ozone are generally related to temperature (18); thus, cities with a variable summer temperature are expected to also have more variable ozone levels. In our study, the variance of summer temperature was significantly correlated with the variance of ozone levels during the warm season (Spearman's $r = 0.41$). Therefore, a possible explanation for the observed effect modification would be that persons with COPD are more likely to be affected by high ozone concentrations when high levels have been sustained for several consecutive days. However, we did not find a statistically significant effect modification by ozone variability in our study ($p = 0.20$, results not shown), suggesting that there must also be

TABLE 3. City characteristics included in the analysis of effect modification, United States, 1986–1999

City, state	People aged ≥65 years		Daily summer apparent temperature (°C)		Percentage of households with central air conditioning	Percentage of PM_{10}* from traffic
	Percentage living in poverty	Emphysema mortality rate (deaths/100,000)	Mean	Variance		
Albuquerque, New Mexico	10.7	45.8	22.9	6.3		1.0
Atlanta, Georgia	13.7	58.7	29.0	10.5	82.6	2.8
Baltimore, Maryland	13.2	23.5	26.6	22.0	67.3	3.2
Birmingham, Alabama	18.3	29.5	29.5	10.9	70.2	1.5
Boston, Massachusetts	9.6	34.9	22.6	25.0	17.0	2.5
Boulder, Colorado	8.7	46.9	20.0	11.3	6.3	1.7
Canton, Ohio	8.7	49.5	22.6	23.1	29.9	2.3
Chicago, Illinois	11.2	34.3	23.7	29.6	43.2	5.0
Cincinnati, Ohio	11.2	29.3	25.4	20.7	66.2	3.5
Cleveland, Ohio	10.2	33.0	23.3	24.4	39.3	5.5
Colorado Springs, Colorado	8.1	65.5	18.2	10.6	13.4	1.6
Columbus, Ohio	10.3	61.0	24.7	22.7	63.7	3.4
Denver, Colorado	12.7	87.9	20.0	11.3	25.7	3.0
Detroit, Michigan	13.5	40.2	23.2	25.7	41.0	7.5
Honolulu, Hawaii	7.8	25.4	29.8	2.0		2.8
Houston, Texas	15.8	46.9	32.9	6.3	93.3	1.1
Jersey City, New Jersey	15.2	27.9	26.1	23.3	35.1	3.2
Los Angeles, California	9.2	28.4	20.7	6.2	32.1	11.7
Minneapolis, Minnesota	8.0	32.7	22.5	25.1	48.4	3.2
Nashville, Tennessee	14.5	63.9	28.7	14.7	72.2	2.5
New Haven, Connecticut	7.7	17.8	22.9	23.1	23.9	5.6
New York City, New York	16.5	17.2	25.6	22.1	11.1	2.4
Palm Beach, Florida	7.1	35.5	32.9	2.7		2.2
Philadelphia, Pennsylvania	16.3	19.9	26.6	23.5	38.8	3.5
Pittsburgh, Pennsylvania	10.1	34.9	23.2	21.3	33.4	5.0
Provo, Utah	6.7	38.4	22.2	17.4	26.9	2.2
Sacramento, California	6.8	46.3	22.2	13.1	71.7	5.6
Salt Lake City, Utah	7.8	42.0	22.2	17.4	39.7	2.8
San Diego, California	6.3	39.3	21.2	7.5	26.6	3.5
San Francisco, California	9.9	29.5	15.8	4.6	13.2	9.0
Seattle, Washington	7.3	34.7	16.7	12.1	6.2	3.0
Steubenville, Ohio	11.4		23.2	21.3	71.6	1.2
St. Louis, Missouri	10.7	37.5	28.2	24.0	76.5	1.7
Spokane, Washington	10.9	73.8	17.3	20.1	28.2	2.2
Washington DC	17.2	23.4	27.9	21.2	86.7	7.9
Youngstown, Ohio	11.3	43.6	21.8	23.7	22.8	3.0
Median (25th, 75th percentiles)	10.5 (8.0, 13.4)	35.5 (29.3, 46.9)	23.2 (21.9, 26.6)	20.4 (10.6, 23.3)	38.8 (24.8, 68.8)	3.0 (2.2, 4.6)

* PM_{10}, particulate matter with an aerodynamic diameter of ≤10 μm.

other explanations for the interaction between ozone and variance of summer temperature.

Our results suggest that the observed interaction between PM_{10} and both the percentage of poverty and the mean summer apparent temperature may be explained by the high correlation of these variables with the proportion of central air conditioning rather than effect modification by the variables themselves. Although air pollution may affect different sociodemographic groups in different ways, in a study conducted in 14 US cities, the association between PM_{10}

TABLE 4. Modification by city characteristics of the effect of air pollution on hospital admissions in 36 US cities during 1986–1999: comparison of the predicted percentage change in hospital admissions at the 25th and the 75th percentiles of the effect-modifier distribution†

	Ozone exposure in the warm season‡							
	Change in COPD§ admissions at the				Change in pneumonia admissions at the			
	25th percentile		75th percentile		25th percentile		75th percentile	
	%	95% CI§	%	95% CI	%	95% CI	%	95% CI
Percentage of people aged ≥65 years living in poverty	0.38	0.06, 0.70	0.24	0.04, 0.45	0.50	0.24, 0.76	0.38	0.21, 0.55
Emphysema mortality rate for people aged ≥65 years (deaths/100,000)	0.28	0.08, 0.49	0.20	−0.10, 0.49	0.47	0.33, 0.61	0.32	0.12, 0.53
Daily summer apparent temperature (°C)								
Mean	0.29	0.04, 0.54	0.26	0.05, 0.48	0.51	0.31, 0.72	0.36	0.19, 0.53
Variance	0.45*	0.20, 0.70	0.12*	−0.12, 0.36	0.42	0.21, 0.63	0.40	0.21, 0.59
Percentage of households with central air conditioning	0.29	0.04, 0.53	0.23	−0.05, 0.51	0.54*	0.38, 0.70	0.30*	0.10, 0.49
	PM_{10} exposure in the warm season¶							
	Change in COPD admissions at the				Change in pneumonia admissions at the			
	25th percentile		75th percentile		25th percentile		75th percentile	
	%	95% CI	%	95% CI	%	95% CI	%	95% CI
Percentage of people aged ≥65 years living in poverty	1.61	0.65, 2.58	1.40	0.69, 2.11	1.37*	0.77, 1.98	0.52*	0.06, 0.98
Emphysema mortality rate for people aged ≥65 years (deaths/100,000)	1.78**	1.10, 2.46	1.03**	0.30, 1.77	0.91	0.49, 1.32	0.85	0.39, 1.31
Daily summer apparent temperature (°C)								
Mean	1.56	0.91, 2.22	1.33	0.52, 2.13	1.13*	0.71, 1.54	0.37*	−0.15, 0.88
Variance	0.96	−0.27, 2.19	1.53	0.95, 2.12	1.07	0.33, 1.80	0.84	0.50, 1.19
Percentage of households with central air conditioning	1.74	0.92, 2.55	1.07	0.03, 2.12	1.47*	0.94, 2.00	−0.11*	−0.79, 0.57
Percentage of PM_{10} from traffic	1.37	0.53, 2.20	1.47	0.93, 2.00	0.53	−0.01, 1.06	0.83	0.49, 1.18

* Two-sided $p < 0.05$ in the meta-regression model; **two-sided $p < 0.1$ in the meta-regression model.

† Estimates are presented as percentage increase in hospital admissions associated with a 5-ppb increase in ozone or a 10-μg/m^3 increase in particulate matter with an aerodynamic diameter of ≤10 μm (PM_{10}).

‡ Modification of the association between ozone during the warm season and hospital admissions was examined by using the 2-day cumulative-effect models.

§ COPD, chronic obstructive pulmonary disease; CI, confidence interval.

¶ Modification of the association between PM_{10} during the warm season and hospital admissions was examined by using the lag 1 model for COPD admissions and the lag 0 model for pneumonia admissions.

and respiratory hospital admissions was not modified by the percentage of poverty or by the percentage of non-White population (8). However, both the latter study and our study used county-level data, which may be too ecologic to be meaningful given that variation in socioeconomic status may be larger within a county than between counties.

In this study, the percentage of PM_{10} from traffic did not modify the risk of respiratory admissions due to PM_{10}. Two different studies in the United States showed that PM_{10} from traffic (31) and particulate matter with an aerodynamic diameter of ≤2.5 μm ($PM_{2.5}$) from mobile sources (34) were associated with a higher mortality risk than particulate matter from other sources. However, one of these studies (34) looked at cause-specific mortality and found no adverse effects of $PM_{2.5}$ from mobile sources for deaths due to COPD or pneumonia. In a morbidity study in 14 US cities, a higher proportion of PM_{10} from traffic was marginally significantly associated with a stronger effect of PM_{10} on pneumonia admissions but not on COPD admissions (9).

The mechanisms behind the adverse respiratory effects of exposure to ozone and PM_{10} are unclear. Some authors have suggested that air pollution may act as an irritant and induce defensive responses in the airways, such as increased mucus secretion and increased bronchial hyperreactivity (35). Both ozone (36) and PM_{10} (37) are potent oxidants that have been shown to produce free radicals and oxidative stress on lung cells. Experimental chamber studies have shown

decrements in forced expiratory volume in 1 second (FEV_1) after exposure to ozone (38), but large interindividual differences have been observed in the responsiveness of both healthy subjects and those with COPD (27). Animal studies have shown an increased vulnerability to PM_{10} in animals with cardiopulmonary disease (39) and exacerbations of ongoing pneumococcal infection after exposure to concentrated ambient $PM_{2.5}$ (40). Exposure to concentrated air particles has also been shown in vivo to increase reactive oxygen species in the lung (41).

The results presented here are unlikely to be due to inadequate control for weather and temporal trends, given that the results from the sensitivity analysis using alternative matching schemes (including control days matched on temperature) showed very similar associations. Confounding by individual characteristics is also unlikely because case-crossover sampling matches perfectly on individual characteristics (10–12). In our study, we did not address confounding by other air pollutants because of previous evidence that the effect of PM_{10} (2) and ozone (3) on respiratory hospital admissions remains practically unaltered after adjusting for other gaseous pollutants. A limitation of our study is that we used ambient air pollution as a surrogate for personal exposure, which may have resulted in a measurement error. Nevertheless, a recent article suggested that this measurement error would generally tend to bias estimates downward (25). Because of the irregular sampling scheme for PM_{10}, we could not assess the cumulative effect of PM_{10} exposure, which may have led to either an overestimation or an underestimation of the risk estimates for each lag. Finally, although the emission estimates from the Environmental Protection Agency should be a reasonable indicator of the actual composition of ambient PM_{10} (9), they may have changed throughout the study period. The same is applicable to data on the percentage of central air conditioning, which was estimated by using data from a population sample during several different years.

In summary, our study confirmed, in a considerably larger sample of cities than, to our knowledge, has been previously examined, that short-term increases in PM_{10} and ozone ambient concentrations are related to hospital admissions for COPD and pneumonia, especially during the warm season. Our findings suggest that some city characteristics modify the effect of air pollution on respiratory hospital admissions. In particular, we found evidence that use of central air conditioning decreases the effect of air pollution and that variability of summer apparent temperature decreases the effect of ozone on COPD. On the other hand, we did not find evidence of a higher toxicity of PM_{10} from traffic as other studies observed for cardiovascular diseases (9, 34).

ACKNOWLEDGMENTS

This study was funded by Environmental Protection Agency grant R-82735301-7 and National Institute of Environmental Health Sciences grant EH-0002.

Conflict of interest: none declared.

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