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Climate Change Impacts on Projections of Excess Mortality at 2030 using Spatially-Varying Ozone-Temperature Risk Surfaces

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Abstract

We project the change in ozone-related mortality burden attributable to changes in climate between a historical (1995–2005) and near-future (2025–2035) time period while incorporating a nonlinear and synergistic effect of ozone and temperature on mortality. We simulate air quality from climate projections varying only biogenic emissions and holding anthropogenic emissions constant, thus attributing changes in ozone only to changes in climate and independent of changes in air pollutant emissions. We estimate nonlinear, spatially-varying, ozone-temperature risk surfaces for 94 US urban areas using observed data. Using the risk surfaces and climate projections we estimate daily mortality attributable to ozone exceeding 40 ppb (moderate level) and 75 ppb (US ozone NAAQS) for each time period. The average increases in city-specific median April-October ozone and temperature between time periods are 1.02 ppb and 1.94°F; however, the results varied by region. Increases in ozone due to climate change result in an increase in ozone-mortality burden. Mortality attributed to ozone exceeding 40 ppb increases by 7.7% (1.6%, 14.2%). Mortality attributed to ozone exceeding 75 ppb increases by 14.2% (1.6%, 28.9%). The absolute increase in excess ozone mortality is larger for changes in moderate ozone levels, reflecting the larger number of days with moderate ozone levels.

Keywords

Climate change; ozone; mortality; ozone-temperature interaction

Introduction

Epidemiological, clinical and toxicological literature has demonstrated a strong association between ozone and health effects. The US Environmental Protection Agency (EPA) concluded that there is a causal relationship between short-term ozone exposures and respiratory morbidity and a likely causal relationship between short-term ozone exposures and cardiovascular morbidity and total mortality, and between long-term exposures and respiratory effects (1). Ambient concentrations of ozone are tightly coupled with meteorological conditions because ozone is formed by photochemical reactions of volatile organic compounds with nitrogen oxides in the presence of solar radiation. Changes in meteorological conditions, induced by global climate change, are expected to alter the distribution of ozone (2), and the frequency of peak ozone concentrations exceeding the US National Ambient Air Quality Standard (US NAAQS). In this manuscript we estimate the mortality burden attributable to changes in ground-level ozone due to climate change and projected through 2030.

Climate and air quality models project that ozone levels will change non-uniformly in response to changes in meteorological conditions (3–5) with different average changes at different quantiles of the ozone distribution and at different locations. Because the ozone effect on mortality is nonlinear (6–8), projected ozone changes will result in a nonlinear change in health burden. Temperature has both a direct effect on mortality and indirect effects on mortality through two pathways: 1) increasing ground-level ozone concentrations (2,9) and 2) modifying the direct ozone effect (7,8,10–12). The combination of the nonlinear direct ozone effect and indirect temperature effects both contribute to a greater mortality burden at higher ozone concentrations and at higher temperatures. However, these nonlinear, synergistic ozone-temperature effects have not been accounted for when estimating the change in ozone-related health burden resulting from climate change.

We project the change in ozone-related mortality due to projected changes in ozone between historical (1995–2005) and future (2025–2035) climate conditions without considering potential changes in air pollutant emissions between those periods. Ozone levels were simulated using year 2006 observed emissions from anthropogenic sources of ozone precursors for both the historical and future periods. Biogenic emissions were allowed to respond to climate change, so all changes in ozone are attributable to climate change alone. The increase in ground-level ozone concentrations due to climate change is known as the “climate penalty” (5). We estimate spatially-varying ozone temperature risk surfaces (8) based on the observed data and use them to project excess mortality attributable to ozone exceeding 40 ppb (moderate level) and 75 ppb (US ozone NAAQS at the time of this analysis) within each time period. We then estimate the climate penalty on ozone-related mortality by comparing ozone-attributable mortality between the two time periods. We include the direct nonlinear ozone effect on mortality and the indirect effects of temperature by two ozone pathways: effect modification and increased ozone formation, but exclude the direct effect of temperature.

Methods

Data

We first estimate parameters of the ozone-temperature risk surface based on the observed data. For that we use data from the National Morbidity, Mortality, and Air Pollution Study (NMMAPS) database (13,14) for daily mortality, ozone, and meteorological conditions in 94 US urban areas. We use daily maximum 8-hour average (MDA8) ozone and daily mean temperature from the NMMAPS database as measures of exposure. The NMMAPS study of the ozone effect on mortality (15) used daily mean temperature and daily mean ozone values. Here we calculate MDA8 from the 24 1-hour averages in the NMMAPS database as described by Smith et al. (7) because the primary NAAQS for ozone is based on the same metric¹. The NMMAPS data also contain daily counts of non-accidental mortality by age group (less than 65, 65–74, 75 and over), as well as daily average dew points for the years 1987 to 2000 for 108 urban areas. We limit the analysis to April through October, the months when photochemistry is most active and US ozone levels are highest, and to the 94 locations in the continental US with reported ozone levels for those months.

Second, we use the estimated risk surfaces to project ozone-related mortality based on simulated climate and air quality for two periods. Fields from the National Aeronautics and Space Administration Goddard Institute for Space Studies (NASA GISS) ModelE2 (16) were dynamically downscaled over North America (17,18) using the Weather Research and Forecasting (WRF) model (19). Two 11-year time slices from ModelE2 were downscaled: 1995–2005 from the “historical” run and 2025–2035 from the Representative Concentration Pathway (RCP) 6.0 projection (20). Ozone and other air pollutant concentrations were then simulated for the same time periods using the Community Multiscale Air Quality (CMAQ) modeling system (21). The air quality simulations assumed constant emissions of anthropogenic origin (based on 2006 values) but allowed biogenic emissions to vary in response to the changes in climate to isolate the climate penalty on ozone formation. Additional details on the regional climate and air quality simulations are described in (17).

The WRF projections of regional climate and the CMAQ projections of air quality were made at a horizontal grid spacing of 36 km × 36 km for the continental US. We interpolated the daily mean temperature and MDA8 ozone projections to county-level using a weighted average of the grid cells overlapping the counties used to define each urban area in the NMMAPS data, with weight proportional to the city area within each grid cell. This resulted in a single ozone and temperature projection for each city for each day. Like the NMMAPS data, we limited the WRF and CMAQ data to April through October. We calibrated the modeled temperature and ozone data from WRF and CMAQ to ensure that the distributions were the same for that overlap period (1995–2000) in each city. The calibration details are in the Supplemental Material.

¹In the original NMMAPS study, summaries of hourly pollutant data were trimmed for the highest and lowest observations because of the concern for the potential influence of outlying observations. While it is not clear how MDA8 ozone calculated from trimmed hourly averages impacts the estimation of risk we calculate MDA8 ozone to approximate the measure used in regulatory analyses by EPA.

Estimating the Bivariate Health Impact Function from the Observed Data

We estimate location-specific bivariate ozone-temperature health impact functions using the approach of Wilson et al. (8) and based on NMMAPS data for daily mortality, ozone, and meteorology. The bivariate risk function quantifies the risk of mortality at each ozone and temperature value with a nonlinear, non-additive ozone-temperature risk surface. With this approach, we impose a monotonicity constraint in the ozone direction ensuring a nonnegative risk estimate even in the presence of exposure misclassification or other modifying factors not included in the models but allow for null effects.

We follow the spatial, monotone model (8) to estimate the 94 city-specific ozone-temperature risk surfaces with the NMMAPS data. We assume the mortality count Y_{ct} in city c at time t is Poisson distributed with log mean

$$\log E(Y_{ct}) = f_c(\text{ozone}_{ct}, \text{temp}_{ct}) + g_c(\text{confounders}_{ct}).$$

The location-specific risk surface f_c is modeled as the outer product of Bernstein polynomial (BP) basis functions. The log risk surface at city c is assumed to be linear in the basis functions

$$f_c(x_1, x_2) = \sum_{j=0}^{M_1} \sum_{k=0}^{M_2} \psi_{jkc} B_{1j}(x_1, M_1) B_{2k}(x_2, M_2),$$

where $B_{lk}(x_l, M_l)$ is the k th BP of order M_l for exposure $l=1,2$ and x_1 denotes MDA8 ozone and x_2 denotes daily mean temperature. We estimate the unknown regression coefficients ψ_{jkc} with the spatial, monotone model of Wilson et al. (8). As in the original NMMAPS study we combine the city-specific risk estimates f_c with a two-stage model. The second stage Bayesian model uses a spatial prior (Gaussian process) on the unknown regression coefficients to allow city-specific risk surfaces to vary smoothly across space and encourage nearby cities to have similarly-shaped surfaces.

Each city's risk surface is constrained so that the risk of mortality is non-decreasing in ozone for any given temperature. The temperature effect is not constrained. The constraint on the city-specific risk surface is implemented by an order constraint on the regression coefficients. This constraint on regression coefficients corresponds to our biological understanding of the health effects of ozone (1) and reduces variability in the estimate while informing the model in areas where the data are sparse. Wilson et al. (8) found that this spatial-monotone model fit the NMMAPS data better than both non-spatial and non-monotone models as well as additive models.

The confounder model g_c is a generalized additive model that includes linear and nonlinear effects for potential confounders including age, dew point, and smooth trends of time. This is the confounder model used in Wilson et al. (8) and is a modification of the confounder model used in Bell et al. (15) with daily mean temperature now incorporated in the exposure model f_c . Full details for the confounder model and computational approach are in the Supplemental Material All analysis was done in R statistical software version 3.0.1.

Projecting Excess Mortality due to Climate Penalty

We use the estimated bivariate impact functions to project excess ozone-attributable mortality for each city and climate period separately. The difference in excess mortality between two periods is the climate penalty on ozone-related mortality. More specifically, we estimated the relative risk (RR) for ozone-related mortality attributed to ozone levels exceeding a moderate level ($q=40$ ppb) and the US NAAQS ozone level at the time of this analysis ($q=75$ ppb) for each city and climate period separately. It is difficult to isolate the effect of ozone in the presence of nonlinear ozone-temperature interaction. We use a counterfactual approach within each time period to estimate the RR due to the direct effect of ozone and the ozone effect modification by temperature but exclude the direct temperature effect.

The RR in city c and time period m , where $m = 0$ denotes 1995–2005 and $m = 1$ denotes 2025–2035, is defined as the ratio of average risk in that period to the average risk in the same time period and city with counterfactual ozone reduced to not exceed the moderate or regulatory level q ,

$$RR_{cm} = \frac{\sum_{t \in T_m} \exp \{f_c(x_{1t}, x_{2t})\}}{\sum_{t \in T_m} \exp \{f_c[\min(x_{1t}, q), x_{2t}]\}},$$

where T_m is the set of April through October days in time period m . Hence, the quantity summed in the numerator is proportional to the ozone-related mortality in the simulated data while the denominator is proportional to the risk in the same city and time period under the counterfactual where MDA8 ozone values do not exceed a concentration q . Because daily mean temperature values are the same in the numerator and denominator, RR_{cm} does not include the direct temperature effect. Rather, by matching temperature and varying ozone between the numerator and denominator we isolate the direct effect of ozone and the modification of the ozone effect by temperature.

Using the city and time period specific relative risk, the total annual excess mortality (EM) for each city c and time period m was calculated as $EM_{cm} = B_c L (RR_{cm} - 1)$, where $L = 214$ days is the length of time exposed each year (April through October) and B_c is the baseline incidence rate, i.e., the mean number of mortalities in the NMMAPS data per day for a given city. We calculated the baseline incidence rate separately for each city and for each of the three age high-risk groups (less than 65, 65–74, 75 and over), then summed the age-specific rates to get the city-specific baseline incidence rates B_c . The age-specific incidence rate is the mean daily mortality count across all NMMAPS days included in the sample described in the Methods Section.

The excess mortality rate (EMR) per 100,000 persons is calculated as $EMR_{cm} = 100,000 \times EM_{cm} / \text{population}_c$. We use the 2000 population for the historical time frame and calculate estimates for the future time period with both the 2000 population and population projections for 2030 (22). These projections are used in US EPA Environmental Benefits

Mapping and Analysis Program (23) analyses to evaluate potential benefits of various proposed air pollution regulations.

We compare EM between time periods with the percent change in EM and compare EMR with the absolute difference between time periods. The between-time-period comparison estimates the climate penalty on ozone mortality, i.e., the change in ozone mortality attributable to changes in climate conditions assuming emissions remain constant. This captures the changes attributable to changing ozone levels and modification of the ozone effect caused by collocated changes in temperature.

Results

Table 1 gives the projected increases in temperature and ozone distributions between the two time periods for April through October and following this climate change scenario.

Averaged over all cities and for those months, the increase in median ozone is 1.02 ppb and the increase in median temperature is 1.94°F between 2000 and 2030. These changes vary considerably by percentile and spatial region. Nationally, the greatest projected ozone increase is 1.70 ppb in the 99th percentile, consistent with an increase in the severity of extreme ozone episodes. In contrast, the increase in extreme warm temperature (a 1.15°F increase at the 99th percentile) is less than the increase over the more moderate temperature ranges. The regions with the largest increases in median ozone are the Industrial Midwest (2.28 ppb), Upper Midwest (1.77 ppb), and Southeast (1.41 ppb). These regions also have large temperature increases. In contrast, the Southwest (−0.73 ppb) and Northwest (−0.07 ppb) are projected to incur decreases in median ozone related to climate change. While the largest increases in ozone concentrations occur at the higher end of the ozone distribution, the largest changes in population exposure levels are projected for moderate levels of ozone because the moderate levels occur much more frequently. The largest between-time-period change in the number of days exceeding a given ozone level occurs at ozone levels between 35 and 65 ppb, which experience a small net increase (5–10 days per year) in days exceeding those levels nationally at 2030 (Supplemental Figure S2).

The projected changes in MDA8 ozone distribution affect the projected number of days exceeding the NAAQS at the time of this analysis of 75 ppb. Figure 1 plots the projected increase in number of days exceeding 75 ppb between 1995–2005 and 2025–2035. Several cities in the Industrial Midwest, Upper Midwest, and Southeast are projected to have an additional 5 or more days per year with ozone exceeding the ozone standard. Among the cities in the NMMAPS study, Muskegon, Michigan has the highest projected increase of 7.8 days per year. In contrast, most cities in the Southwest are projected to have a decrease in high ozone days, with Phoenix, Arizona having the largest projected reduction of 4.4 days per year.

Regional differences in projected changes in mortality are also displayed in Figure 1. These changes reflect both local changes in ozone and local variation in health effect estimates. Most cities are projected to have a modest increase in the mortality rate. Changes of 0.05 or more additional urban mortalities per year per 100,000 people are concentrated in a few cities in the Industrial Midwest and Northeast: St. Louis, Missouri, Muskegon, Michigan,

and Worcester, Massachusetts; however, none are statistically significant. The uncertainty of estimated effects is given in Supplemental Figure S3. There are also projected increases in parts of the Southeast, including Knoxville, Tennessee and Tulsa, Oklahoma. Importantly, the cities with large changes in the mortality rate tend to be small or medium size cities and despite the large increase in excess mortality rate these cities make a relatively small impact on the total change in excess mortality. On the other hand, the largest cities (Chicago; Illinois; Los Angeles, California; and New York, New York) tend to have smaller changes in excess mortality rates.

The estimated joint health effects of temperature and ozone are summarized in Figure 2 using the national average of the 94 city-specific risk surfaces. The risk is allowed to vary by city and uses a statistical model designed to allow for flexibility in the risk surface at higher ozone concentrations. The risk estimate is centered on zero at median temperature and ozone; hence, the y-axis can also be interpreted as the log RR relative to median temperature and ozone. The effects of both ozone and temperature on mortality are nonlinear. For example, the risk is similar for all temperatures at or below the 50th percentile, but increases dramatically at the 75th and 95th temperature percentiles. Similarly, for each temperature, the ozone risk curves show a steeper increase above 70 ppb. The main ozone effect is nonlinear at all temperatures. The effect modification of the nonlinear ozone effect by temperature is present but negligible in comparison to the main effects. The negligible effect of modification of the nonlinear main effect of ozone is demonstrated through nearly parallel concentration-response relationship at moderate percentiles (e.g. 25th, 50th, and 75th percentiles of temperature in the national concentration-response functions in Figure 1). At higher temperatures and ozone levels the effect modification is more pronounced but impacts very few days, thus resulting in negligible effect on burden.

Combining the projected changes in temperature and ozone with health effect estimates gives the projected changes in EM in Table 1. The estimated number of excess deaths attributed to days with ozone above 75 ppb increases 14.2% (1.6%, 28.9%) nationally, from 47.2 per year in 1995–2005 to 53.7 per year in 2025–2035. Similarly, the estimated number of excess deaths attributed to days with ozone above 40 ppb increases 7.7% (1.6%, 14.2%) nationally, from 399.5 per year in 1995–2005 to 430.4 per year in 2025–2035. The increases are larger, 50.5% (26.6%, 57.5%) for 75 ppb and 39.8% (23.0%, 57.5%) for 40 ppb, when accounting for population changes. Consistent with projected population trends these expected increases are disproportionately focused in the Southeast, Southwest, and Southern California where population growth is expected to be largest. In comparison, the Industrial Midwest is the only region where population growth does not impact projected mortality. Supplement Figure S4 compares the regional estimates.

The projected change in EM attributed to ozone exceeding 75 ppb (14.2%) is larger than for ozone exceeding a more moderate level of 40 ppb (7.7%) because it includes larger projected changes in ozone levels at higher ozone percentiles (Table 1) and the nonlinear effect at higher ozone concentrations (Figure 2). However, Table 2 also shows that the total increase in EM attributed to ozone exceeding 40 ppb results in a larger overall contribution to the health burden (430 compared to 54 annual mortalities). Despite the relatively small per-day impact of days that exceed 40 ppb, the cumulative effect of moderate ozone level

days is substantial because there are far more days that exceed 40 ppb (135.6 days per city per year in 2000) compared to days that exceed 75 ppb (14.6 days per city per year). In other words, ozone exceeding 75 ppb has a larger per day effect on the health burden, but the overall number of days with ozone exceeding 40 ppb results in a larger overall health burden. Over moderate temperatures and ozone values the ozone effect is nonlinear but there is a small effect modification by temperature. Because of the preponderance of days with lower ozone concentrations and moderate temperatures the nonlinearity of ozone dominates the effect modification by temperature when projecting the change in EM. Figure 4 shows that there is no lower threshold of MDA8 ozone at which the climate penalty does not impact ozone-related mortalities. Similar analyses can be done for any ozone level. However, the uncertainty of the estimates increases substantially below 30 ppb because many cities never or rarely experience these low levels. For this reason we do not present estimates for concentrations below 30 ppb in Figure 3 and focus our analysis on the moderate but more precisely estimated level of 40 ppb.

Discussion

This study estimated the impacts of climate change on ambient ozone levels and the resulting impacts on the ozone-related mortality between the periods 1995–2005 and 2025–2035. We used a nonlinear, monotone, bivariate health impact function for MDA8 ozone and daily mean temperature to associate ozone and temperature with mortality. Median ozone concentrations increased by 1.02 ppb between the two time periods, while the magnitude and direction of changes varied considerably across regions of the US and across percentiles of ozone distributions. The results suggest that for the 2025–2035 time period, the added burden of short-term exposures is not only attributable to increases in the highest concentrations but to moderate increases in ozone across the overall distribution.

We introduced the use of a bivariate health impact function to quantify the health burden from projected changes in ozone. This health impact function allowed us to estimate the direct nonlinear ozone effect and modification of the direct effect by temperature within each time period, but exclude the direct effect of temperature. Because the direct temperature effect is excluded from the ozone-related EM estimates, all changes in ozone mortality between the two time periods are attributed to climate changes. The difference in ozone mortality burden between the two time periods quantifies the effect of climate change on ozone mortality burden that results from climate-induced changes in ozone levels in combination with changes in temperature which modifies the ozone effect. We call the difference in mortality between two time periods a ‘climate penalty’ on ozone-related mortality. The direct effect of temperature changes between the two periods would also add to the mortality burden resulting from climate change, but is not the focus of this analysis.

Regional analysis revealed important differences in EMR due to future ozone concentrations. The largest changes in EM were estimated within the regions that saw the largest increases in ozone across the entire distribution (Industrial Midwest and Southeast). However, the largest increases in EM after accounting for projected population growth were found for the Southeast and Southern California. Additionally, when evaluating EM with

respect to various reference levels of ozone at the national level we did not detect an apparent threshold below which a climate penalty on health burden was not estimated.

There are several limitations to our analysis that should be considered in interpretation of results. When possible, we took measures to reduce some uncertainties, such as the time horizon and emission scenarios used in modeling of climate projections. Climate models are mathematical representations of complex systems of atmospheric processes that inherently include a number of uncertainties. This study used ozone concentrations forced by dynamically downscaled regional climate fields that were based on global climate projections. Dynamic downscaling is used to represent atmospheric processes, topography, and patterns of land use at a higher spatial resolution than is available from global climate models. While some of the climate changes suggested by the scenario used here are robust, it is important to note that the ozone projections modeled in this study are the result of only one global climate model forced by one greenhouse gas scenario. There are four RCPs developed for IPCC AR5 (RCP 8.5, RCP 6.0, RCP 4.5, and RCP 2.6; see IPCC (24)) that differ by the amount of radiative forcing in Wm^{-2} at 2100 relative to pre-industrial conditions. We used RCP 6.0 which considers a moderate increase in energy use from coal and oil and a significant increase in use of natural gas, nuclear energy, solar, wind, geothermal and hydropower (25). However, the differences between the RCPs are relatively minor out to 2030. A significant divergence takes place among emission scenarios for simulations beyond the 2030s, adding uncertainties to air quality projections extending beyond that time frame.

Several studies have evaluated ozone induced health impacts under climate predictions for more distant future time periods. For a future period centered around 2050, ozone-related mortality was estimated to increase by a median 4.5% across the 31 counties in the New York metropolitan region (26) while a total mortality was estimated to increase by 0.11–0.27% across 50 eastern US cities (27). Nationally, 300 additional ozone-related premature deaths annually were attributed to changes in ozone levels by Tagaris et al. (28). While most studies have found projected changes in ozone to be associated with an additional burden, others have found that uncertainties in future climate, the concentration-response function, and population projections are large enough to encompass this burden. For example, Post et al. (29) estimated future mortality due to changes in ozone levels as ranging from >600 deaths avoided to >2,500 additional deaths, depending on the underlying modeling system. In addition, these studies have used linear concentration response functions. Wilson et al. (8) compared the bivariate approach with linear additive models and found that the nonlinear bivariate approach resulted in larger ozone effect estimates at higher concentrations but lower effect estimates at moderate concentrations. This suggests that EM attributable to ozone exceeding 75 ppb would be lower under a linear model within each time period. The study also found that the bivariate model fit the observed data better than the linear additive or nonlinear additive models.

The timeframe of 2030 is considered a policy relevant period because it encompasses the current air-quality planning horizon. However, it is important to note that these estimates of the future impacts of climate change on ozone levels and the associated health burden are based on emissions for the year 2006 and reflect neither improvements in air quality due to

emissions controls promulgated to attain compliance with the NAAQS nor changes in background ozone concentrations transported into the U.S. Between 1998 and 2013, trends in 50th, 75th, and 95th percentile summer ozone levels were predominantly decreasing at most urban and suburban sites in the U.S., while trends in 5th and 25th percentile ozone were increasing, particularly in the West and Southwest (30). Emissions of nitrogen oxides from electrical generating units and mobile sources are expected to continue to decline significantly by 2030 due to the full implementation of the Cross-State Air Pollution Rule and the Tier 2 and Tier 3 standards for on-road mobile sources, regulations on nonroad mobile sources, and various additional regulations on point sources such as industrial boilers and cement kilns. These emissions changes are likely to continue to compress the ozone distribution, which will increase concentrations at low and median percentiles and decrease concentrations at upper percentiles, potentially increasing the importance of applying our bivariate approach to the full ozone distribution.

The choice of health impact function is another source of uncertainty in evaluating the health burden (29). Biases can be introduced when impacts are extrapolated beyond the subpopulations for which the risk was estimated. To reduce the uncertainty due to geographic effect heterogeneity we restricted our analysis to the 94 locations used to estimate the impact function and applied city-specific impact functions rather than the national averages. The bivariate health impact function used in this analysis led to lower estimates of relative risk for low and moderate ozone levels in comparison to linear and additive models. In using our approach, the estimated effect of ozone increases for higher levels of ozone; however, as previously noted there are relatively few days with very high ozone levels, and those days result in a comparatively small impact on overall EM. Our impact function may reflect added uncertainties in association between daily ozone levels and mortality due to calculation of MDA8 ozone from the historic NMMAPS hourly data that used trimmed hourly means. It is therefore important to consider methods introduced here and replicate analysis with impact functions developed on more recent data, which were not available to us. Our analysis did not consider the impacts of between-time-period changes in population age and demographics with respect to heat stress and ozone. Accounting for future changes in age distribution would have an impact on mortality burden estimates. The duration of the ozone season may lengthen as well, further increasing the health burden, but this was not considered in this study. The role of increased use of air conditioners and acclimatization to heat and ozone was also not accounted for in this study.

Conclusions

We evaluated the impacts of climate change on ambient ozone by modeling air quality under the recent historical climate (1995–2005) and a near future climate projection (2025–2035) following RCP 6.0 and varying only biogenic emissions. In evaluating the mortality burden we considered synergism between ozone and temperature and the nonlinear direct effect of ozone. The results have important implications toward our understanding of the added burden due to changes in ozone, suggesting that the largest burden of short term exposures is attributable to moderate increases in ozone across the overall distribution rather than to the highest concentrations alone. This was true even after accounting for a nonlinear concentration-response relationship.

Supplementary Material

Refer to Web version on PubMed Central for supplementary material.

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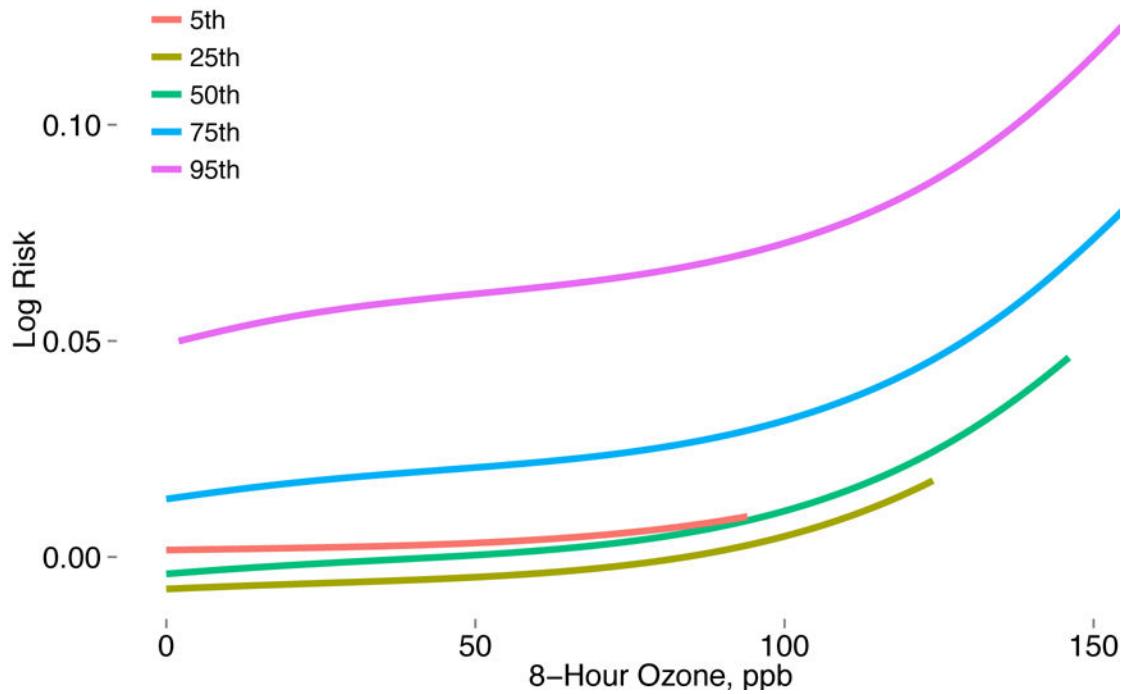


Figure 2. Cross-section of the national ozone-temperature log risk surface. The cross-sections are presented so that median ozone (44.75 ppb) and median temperature (70.50 °F) are centered on 0. Hence, this shows the log relative risk compared to median MDA8 ozone and daily average temperature. Each curve represents the effects of increasing ozone for a specific temperature percentile.

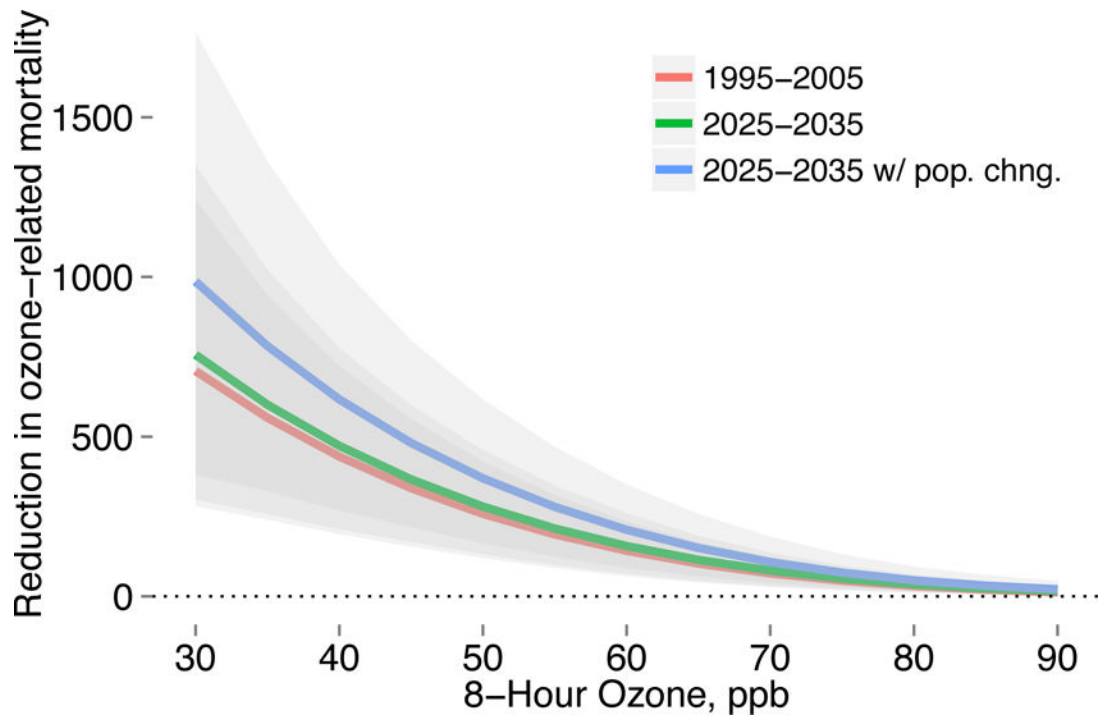


Figure 3. Premature mortalities attributable to ozone exceeding a given value for a historical and a future time period. The three overlapping grey bands are 95% credible regions for each estimate.

Summary of changes in MDA8 ozone and daily mean temperature from 1995–2005 to 2025–2035, averaged over cities within each region for the mean and several percentiles across the distributions. Region definitions follow Samet et al. (2000a,b).

Table 1

	Mean	1 st	10 th	25 th	50 th	75 th	90 th	99 th
	Maximum 8-Hour Average Ozone (ppb)							
National	0.89	0.58	0.49	0.64	1.02	1.15	0.88	1.70
Industrial Midwest	2.07	1.70	1.55	1.79	2.28	2.19	2.14	3.57
Northeast	0.77	-0.15	0.11	0.49	0.84	1.25	0.73	2.92
Northwest	-0.01	-0.12	0.08	-0.10	-0.07	0.18	-0.22	0.30
Southern CA	-0.17	0.55	-0.63	-0.50	0.01	-0.24	-0.27	2.30
Southeast	1.14	0.69	0.57	0.78	1.41	1.51	1.25	0.77
Southwest	-0.59	0.05	-0.71	-0.93	-0.73	-0.21	-0.60	-0.82
Upper Midwest	1.56	0.69	1.48	1.70	1.77	1.69	1.45	2.31
	Mean Daily Temperature (°F)							
National	1.82	0.84	1.73	2.43	1.94	1.66	1.46	1.15
Industrial Midwest	2.40	-0.74	2.25	3.76	2.66	1.99	1.64	1.76
Northeast	1.95	-1.05	1.50	2.95	2.25	1.73	1.49	2.04
Northwest	1.37	2.73	2.17	2.08	1.13	0.92	0.77	0.29
Southern CA	0.93	1.96	1.49	1.20	0.70	0.83	0.32	0.59
Southeast	1.59	1.34	1.29	1.58	1.75	1.73	1.64	0.50
Southwest	1.57	1.73	1.60	1.82	1.93	1.34	1.09	0.10
Upper Midwest	2.79	1.93	2.09	3.36	2.64	2.86	3.09	3.32

Annual average (April-October) excess ozone-related mortality and excess ozone-related mortality rate per 100,000 persons derived from national estimate of ozone and temperature health impact function.

Table 2

	Attributable to Ozone > 40 ppb		Attributable to Ozone > 75 ppb	
	Mean	Cred. Int.	Mean	Cred. Int.
	Excess Mortality			
1995–2005	399	(162, 688)	47	(16, 87)
2025–2035	430	(176, 7456)	54	(18, 99)
2025–2035 w/pop chng	560	(219, 986)	71	(22, 135)
	Percent Change in Excess Mortality			
2025–2035	7.7	(1.6, 14.2)	14.2	(1.6, 28.9)
2025–2035 w/pop chng	39.8	(23.0, 57.5)	50.5	(26.6, 69.8)
	Excess Mortality Rate per 100,000			
1995–2005	0.388	(0.157, 0.668)	0.046	(0.015, 0.084)
2025–2035	0.418	(0.170, 0.724)	0.052	(0.017, 0.096)
2025–2035 w/pop chng	0.418	(0.163, 0.735)	0.053	(0.017, 0.101)
	Change in Excess Mortality Rate per 100,000 People			
2025–2035	0.030	(0.005, 0.067)	0.006	(0.00, 0.015)
2025–2035 w/pop chng	0.030	(–0.016, 0.095)	0.007	(–0.001, 0.019)
	Mean number of days exceeding ozone level per city-year			
1995–2005	135.6		14.6	
2025–2035	139.1		16.1	
Pct. change	2.5%		9.0%	