SPECIAL ARTICLE

American Thoracic Society and Marron Institute Report

Estimated Excess Morbidity and Mortality Caused by Air Pollution above American Thoracic Society–Recommended Standards, 2011–2013

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Abstract

Estimates of the health impacts of air pollution are needed to make informed air quality management decisions at both the national and local levels. Using design values of ambient pollution concentrations from 2011–2013 as a baseline, the American Thoracic Society (ATS) and the Marron Institute of Urban Management estimated excess morbidity and mortality in the United States attributable to exposure to ambient ozone (O_3) and fine particulate matter ($PM_{2.5}$) at levels above the American Thoracic Society–recommended standards. Within the subset of counties with valid design values for each pollutant, 14% had $PM_{2.5}$ concentrations greater than the ATS recommendation, whereas 91% had O_3 concentrations greater than the ATS recommendation. Approximately 9,320 excess deaths (69% from O_3; 31% from $PM_{2.5}$), 21,400 excess morbidities (74% from O_3; 26% from $PM_{2.5}$), and 19,300,000 adversely impacted days (88% from O_3; 12% from $PM_{2.5}$) in the United States each year are attributable to pollution exceeding the ATS-recommended standards. California alone is responsible for 37% of the total estimated health impacts, and the next three states (Pennsylvania, Texas, and Ohio) together contributed to 20% of the total estimates. City-specific health estimates are provided in this report and through an accompanying online tool to help inform air quality management decisions made at the local level. Riverside and Los Angeles, California have the most to gain by attaining the ATS recommendations for O_3 and $PM_{2.5}$. This report will be revised and updated regularly to help cities track their progress.

Keywords: ozone; particulate matter; risk assessment; environmental policy

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In an effort to protect public health with an adequate margin of safety, the Clean Air Act directs the U.S. Environmental Protection Agency (EPA) to establish National Ambient Air Quality Standards (NAAQS) for common air pollutants. The American Thoracic Society (ATS, New York, NY), a professional organization representing more than 15,000 physicians, research scientists, and other health professionals, has regularly provided scientific recommendations to the EPA regarding these standards. The ATS currently recommends more stringent primary standards than the EPA for two of the six "criteria" pollutants included in the NAAQS (available from www.epa.gov/criteria-air-pollutants/naaqs-table). Since 2007, the ATS has repeatedly recommended an 8-hour ozone (O_3) standard of 0.060 ppm and since 2012 it has recommended an annual fine particulate matter ($PM_{2.5}$) standard of 11 $\mu g/m^3$ (1–4).

Working in collaboration, the Marron Institute of Urban Management at New York University (New York, NY) and the ATS have compiled estimates of annual U.S. state- and city-level health burdens attributable to pollution concentrations that exceed ATS-recommended standards. This first annual "Health of the Air" report provides estimates of the health impacts attributable to concentrations of $PM_{2.5}$ and O_3 above ATS-recommended standards, based on air quality data measured nationwide in 2011–2013. An accompanying online tool, with searchable information by city, can be found at www.HealthoftheAir.org. The information provided by this report and accompanying online tool is intended to increase public awareness and better inform public decision-making regarding the management of outdoor air pollution.

County-specific information regarding ambient pollution concentrations relative to federal standards is widely available (5). However, this analysis goes further by providing estimates of the health impacts experienced by communities associated with elevated concentrations of outdoor air pollution, while also considering city-specific demographics and baseline health conditions. These previously unavailable health data should help inform air quality management decisions including the setting of national standards, state planning efforts to meet federal air quality regulations, and cities identifying opportunities to improve air quality.

Methods

Exposure Assessment

Current $PM_{2.5}$ and O_3 air pollution concentrations were estimated for each county with a valid design value for 2011–2013 (available from www3.epa.gov/airtrends/values.html). A design value is the 3-year average of pollution concentrations measured at each monitoring location and is used to determine whether a county is in attainment with federal air quality standards. For $PM_{2.5}$, the design value is the 3-year average of the annual mean concentration. For O_3, the design value represents the 3-year average of the fourth highest daily 8-hour maximum ozone concentration. In counties with more than one valid design value, the highest value was selected, which is government protocol when determining attainment status.

Design values provide stable estimates of pollution concentrations for a given location. However, these values do not provide information regarding the day-to-day variability in pollution concentrations, which is known to differ between cities. To incorporate city-specific variability of pollution concentrations, hourly O_3 and daily $PM_{2.5}$ values from design value monitors for 2013 (downloaded from www3.epa.gov/airdata/) were quantitatively adjusted by a uniform monitor specific factor to be equivalent to 2011–2013 design values. By using hourly measurements for O_3, we were also able to make exposure estimates corresponding to three different averaging metrics: 1-hour maximum, 8-hour maximum, and 24-hour average (6). An example of the pollution values used in this analysis can be found in Table E1 in the online supplement.

Baseline and control concentrations were inputted into BenMAP-CE, using nearest monitor interpolation. The baseline concentrations correspond to design values from 2011–2013. The control concentrations correspond to the ATS recommendations of 11 $\mu g/m^3$ for $PM_{2.5}$ and 0.060 ppm for O_3.

Health Impact Assessment

BenMAP-CE 1.1 was used to determine the health effects attributable to air pollution concentrations that exceed ATS recommendations. BenMAP is an open-source program, provided by the EPA, that is used in regulatory cost–benefit analysis as well as in academic research (7, 8). More information on this software can be found at www.epa.gov/benmap.

Estimated health impacts resulting from the difference in pollution concentrations between baseline and control conditions were divided into three general categories: mortality, major morbidity (including acute myocardial infarction, chronic bronchitis, cardiovascular and respiratory hospital admissions, and emergency department visits), and adverse impact days (including restricted activity days, acute respiratory symptom days, work loss days, and school loss days). We constrained our inclusion of health impact functions to the EPA standard health functions. Table E2 lists the epidemiology studies by pollutant and metric from which concentration–response functions were selected for use in this report.

In some cases only a single study was available in BenMAP for a given health end point. For other health end points with more than one available study, discretion was used as to which studies should be included in this analysis. For study inclusion, we prioritized most recent updates of large prospective cohort studies, multicity studies, studies that assessed associations for more general (e.g., all respiratory hospital admissions) as opposed to specific (e.g., pneumonia hospital admissions) health outcomes, studies that controlled for copollutant exposures, and studies assessing health impacts across wide age ranges. For the majority of health estimates, a single best study was identified for consistent application across all U.S. locations. However, for respiratory hospital admissions and emergency department visits associated with ozone exposures, there was no single preferred study. For these outcomes the coefficients from the multiple single-city studies were given uniform weight.

Health effect estimates were determined using all three calculated O_3 metrics (24-h average, 8-h maximum, and 1-h maximum) when available. Depending on the distribution of hourly O_3 concentrations in a given geographic location, the maximum estimated health outcomes varied by averaging metric (6). For this reason, health outcomes for which estimates were available from multiple metrics were arithmetically averaged for each county, thereby showing no preference for any one particular metric. This approach may result in conservative estimates for health end points that may be more strongly associated with peak O_3 levels.

For the online reporting tool, health impact estimates were aggregated at the 2010 census metropolitan statistical area (MSA) level, or on the basis of metropolitan divisions when available, for $PM_{2.5}$ and O_3. These aggregated values are simple summations from all counties in MSAs that have monitors with valid design values available. There are counties with estimated health impacts that are not included in any MSA boundary; the health effects estimated for counties outside an MSA are reported independently.

Results

There are more than 3,000 counties in the United States. Of the 483 counties with valid design values for $PM_{2.5}$ from 2011 to 2013, 66 counties (14% of monitored counties) across 20 states had concentrations higher than the ATS recommendation of 11 $\mu g/m^3$. For O_3, the percentage of monitored counties exceeding ATS recommendations is much higher. Of the 715 counties with valid design values for O_3 from 2011 to 2013, 654 (91% of monitored counties) across 46 states had concentrations greater than the ATS recommendation of 0.060 ppm. Figures 1 and 2 show design values by county for $PM_{2.5}$ and O_3, respectively. Because health effects estimates were made only for pollution concentrations exceeding ATS-recommended standards, the majority

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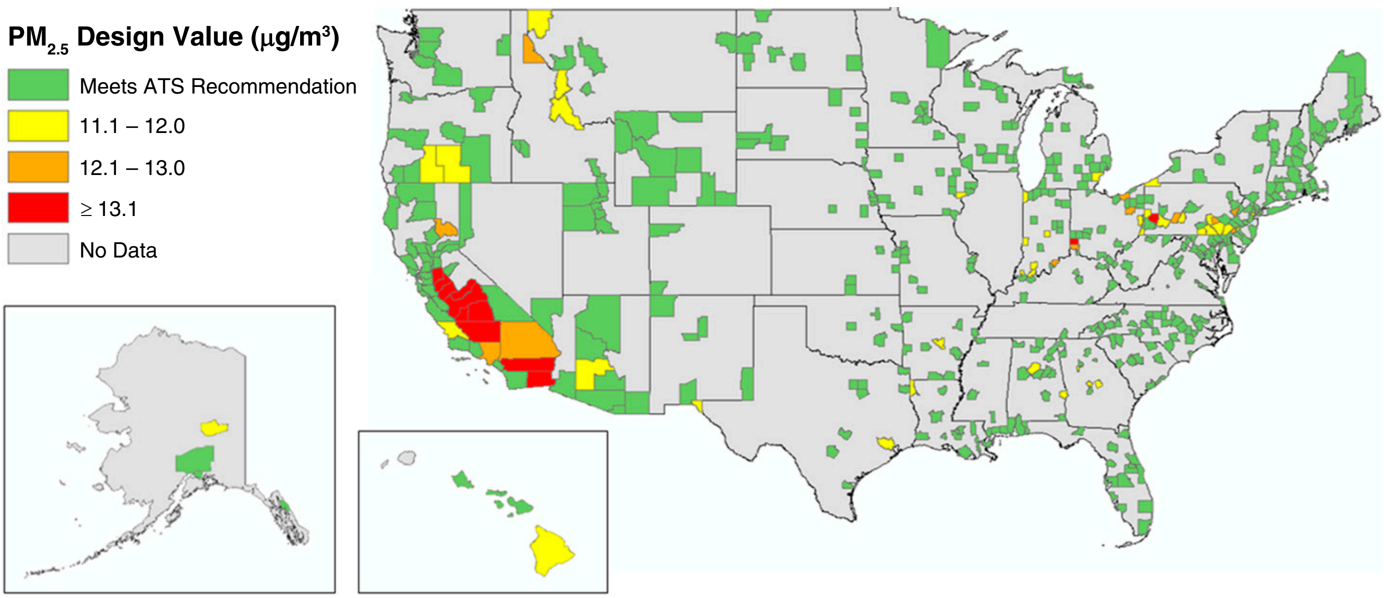

Figure 1. Nationwide design values for fine particulate matter ($PM_{2.5}$) by county, 2011–2013. Design values for $PM_{2.5}$ are the 3-year average of annual mean concentrations measured at monitoring locations and are used to determine attainment status with federal standards. The American Thoracic Society (ATS) recommends an annual $PM_{2.5}$ standard of 11 μg/m^3, which is lower than the current U.S. Environmental Protection Agency standard of 12 μg/m^3.

of total health impacts across the United States are attributable to O_3 in this analysis.

Across the entire country, the number of annual excess deaths attributable to pollution concentrations greater than ATS recommendations for O_3 and $PM_{2.5}$ are approximately 6,408 (95% confidence interval [CI], 2,517–10,217) and 2,913 (95% CI, 1,980–3,858), respectively. The percentage of total morbidities (74%) and impacted days (88%) attributable to O_3, as compared with $PM_{2.5}$, is greater than for mortality risk (69%). The annual number of morbidities attributable to O_3 and $PM_{2.5}$ levels above ATS recommendations is

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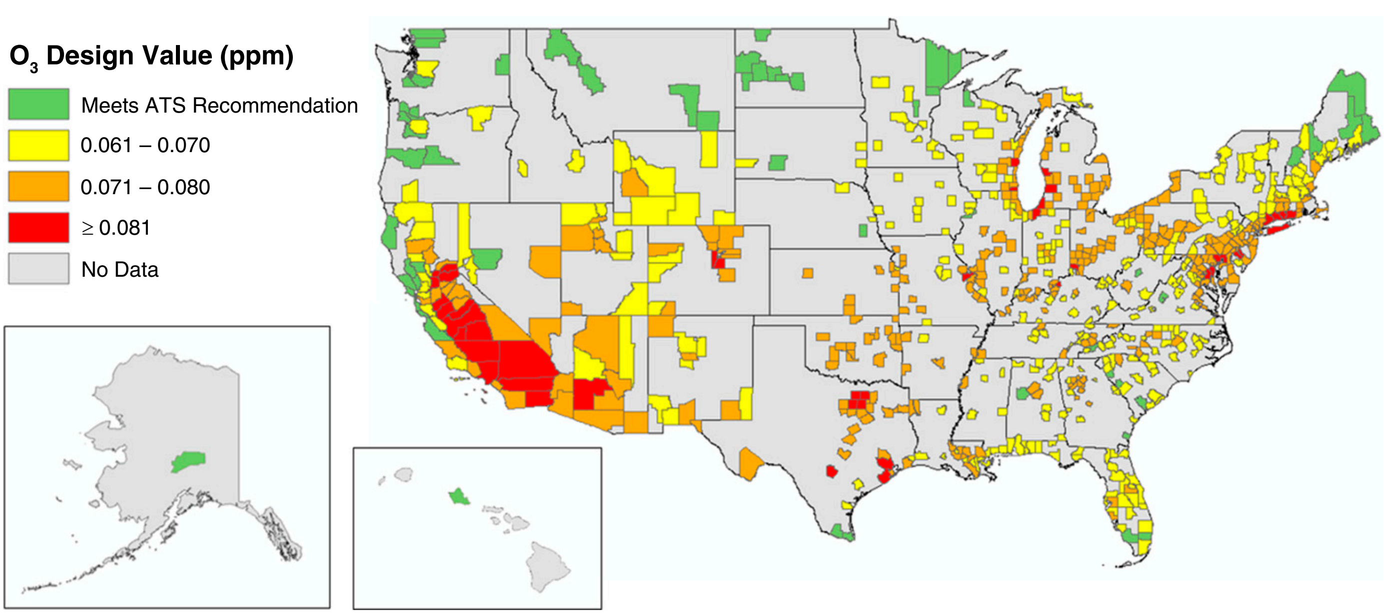


Figure 2. Nationwide design values for ozone (O_3) by county, 2011–2013. Design values for O_3 are the 3-year average of the fourth highest daily 8-hour maximum concentrations measured at monitoring locations and are used to determine attainment status with federal standards. The American Thoracic Society (ATS) recommends an 8-hour maximum standard of 0.060 ppm, which is lower than the current U.S. Environmental Protection Agency standard of 0.070 ppm.

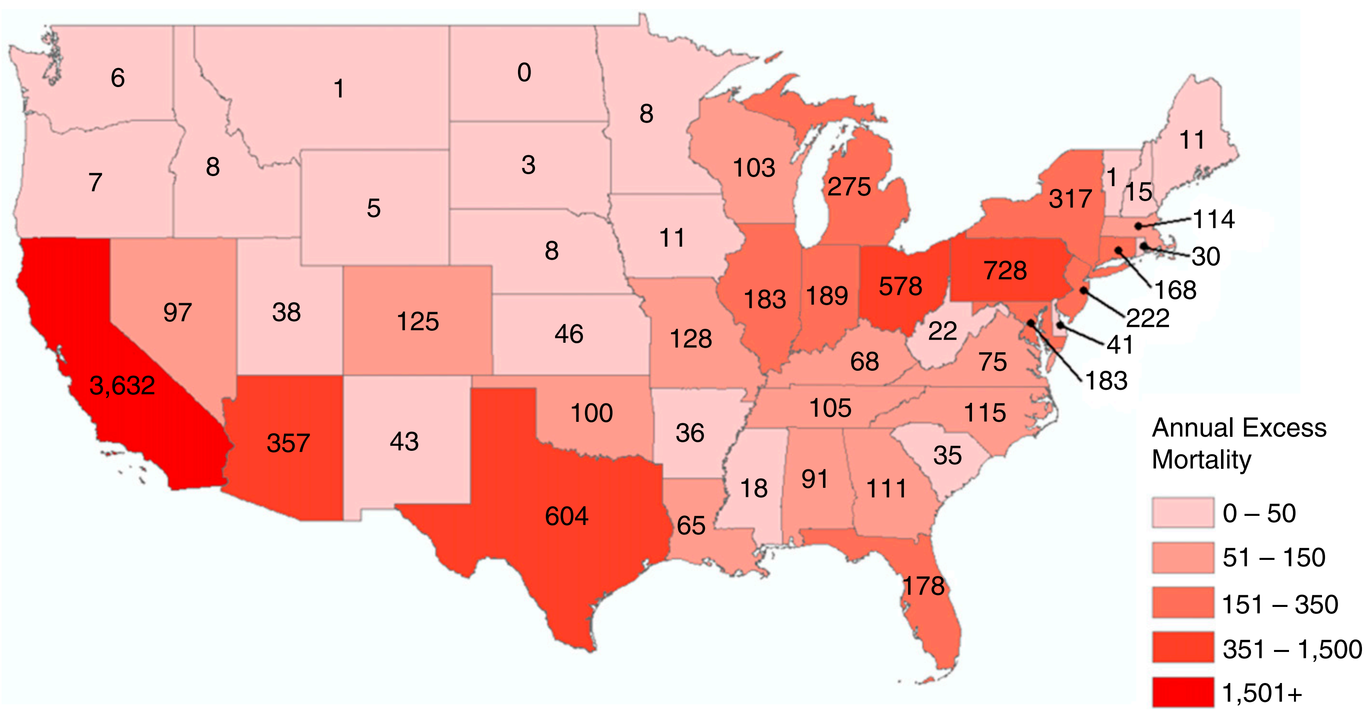
15,869 (95% CI, –4,966 to 36,023) and 5,543 (95% CI, 1,741–9,253), respectively. The total number of adverse impact days attributable to O_3 and $PM_{2.5}$ levels above ATS recommendations is 16,991,656 (95% CI, 4,651,415–28,781,244) and 2,348,094 (95% CI, 1,928,554–2,765,482), respectively.

As shown in Figure 3, the 9,321 deaths, 21,412 morbidities, and 19,339,750 adversely impacted days attributable to air pollution in excess of ATS-recommended standards are not uniformly distributed across the United States. For example, California alone is responsible for approximately 37% of the total estimated health impacts. The three states with the next highest impacts (Pennsylvania, Texas, and Ohio) are together responsible for approximately 20% of the total estimated health impacts. Table 1 lists the number of deaths, morbidities, and impacted days for each state. Table E3 provides a full breakdown of design values and health impacts, separated by pollutant, county, and state.

Individual cities (defined in this analysis by MSAs) with the highest air pollution–related impacts have large populations and relatively high concentrations of at least one of the two pollutants considered in this analysis. A ranking of the cities with the most to gain by improving air quality sufficiently to meet the ATS-recommended standards is shown in Table 2. Of particular note is the relative contribution of health impacts from $PM_{2.5}$ and O_3 across different cities. For example, the total number of excess deaths attributable to air pollution in Phoenix, Arizona and Pittsburgh, Pennsylvania is nearly identical. However, the number of estimated excess morbidities and adversely impacted days is much higher in Phoenix than Pittsburgh, primarily due to higher O_3 concentrations in Phoenix. Detailed results for every individual city, as well as comparisons of multiple user-defined cities, are available using the web tool accompanying this report at www.HealthoftheAir.org.

Discussion

The national estimates in this report indicate that there are substantial public health impacts from air pollution in the United States. For comparison, the estimated 9,320 deaths attributable to air pollution in this report are quantitatively comparable to the 10,076 alcohol-related traffic deaths that occurred in the United States in 2013 (9). In addition to providing



NATIONWIDE TOTALS

Ozone (O_3)			Fine Particulate Matter ($PM_{2.5}$)		
Health Category	*Estimate*	*95% CI*	*Health Category*	*Estimate*	*95% CI*
Mortality	6,408	(2,517 – 10,217)	Mortality	2,913	(1,980 – 3,858)
Morbidity	15,869	(–4,966 – 36,023)	Morbidity	5,543	(1,741 – 9,253)
Impacted Days	16,991,656	(4,651,415 – 28,781,244)	Impacted Days	2,348,094	(1,928,554 – 2,765,482)

Figure 3. Nationwide health impacts attributable to air pollution exceeding ATS recommendations for O_3 and $PM_{2.5}$. Annual excess health impacts are estimated as a function of outdoor pollution concentrations, size of the exposed population, and baseline health risks. States with high estimates of air pollution–related health impacts typically have a combination of high pollution concentrations and large populations. For state level estimates of excess mortality, excess morbidity, and impacted days, *see* Table 1. For county-level estimates, *see* Table E3.

Table 1. Annual health impacts, by state, attributable to pollution concentrations exceeding ATS recommendations

State	Excess Mortality (95% CI)	Excess Morbidity (95% CI)	Impacted Days (95% CI)
Alabama	91 (45–137)	179 (–7 to 361)	145,126 (52,025–236,028)
Alaska	N/A	N/A	N/A
Arizona	357 (164–546)	733 (–99 to 1,541)	767,664 (251,303–1,261,339)
Arkansas	36 (18–54)	76 (–6 to 156)	56,788 (17,261–95,027)
California	3,632 (1,988–5,236)	7,686 (–126 to 15,140)	6,741,955 (2,772,068–10,459,550)
Colorado	125 (49–199)	250 (–84 to 573)	388,492 (104,795–660,459)
Connecticut	168 (65–267)	472 (–115 to 1,039)	344,567 (96,304–581,861)
Delaware	41 (16–65)	104 (–15 to 220)	88,347 (24,478–149,606)
District of Columbia	18 (8–29)	57 (–14 to 124)	58,901 (18,685–97,717)
Florida	178 (69–286)	374 (–80 to 824)	337,847 (94,580–577,573)
Georgia	111 (45–176)	285 (–88 to 649)	366,860 (102,287–622,899)
Hawaii	N/A	N/A	N/A
Idaho	8 (4–12)	14 (–2 to 30)	16,635 (4,979–28,088)
Illinois	183 (71–293)	638 (–168 to 1,427)	487,198 (132,116–833,207)
Indiana	189 (90–286)	471 (–32 to 964)	315,834 (107,825–518,039)
Iowa	11 (4–17)	24 (–2 to 49)	20,330 (5,490–34,970)
Kansas	46 (18–74)	129 (–36 to 290)	117,515 (30,460–201,654)
Kentucky	68 (25–110)	132 (–17 to 278)	89,869 (24,301–153,503)
Louisiana	65 (28–102)	143 (–27 to 310)	142,346 (39,581–242,821)
Maine	11 (4–17)	24 (–5 to 53)	17,171 (4,851–29,229)
Maryland	183 (71–293)	435 (–164 to 1,017)	451,924 (125,779–765,792)
Massachusetts	114 (44–184)	293 (–81 to 661)	234,713 (66,322–399,788)
Michigan	275 (116–431)	640 (–161 to 1,421)	565,414 (158,112–958,845)
Minnesota	8 (3–12)	18 (–5 to 42)	26,619 (6,939–46,045)
Mississippi	18 (7–29)	39 (–8 to 87)	40,817 (10,679–70,363)
Missouri	128 (51–205)	258 (–93 to 599)	276,068 (76,266–468,190)
Montana	1 (1–1)	2 (0 to 3)	579 (474–683)
Nebraska	8 (3–12)	17 (–3 to 37)	22,068 (5,704–38,132)
Nevada	97 (38–155)	194 (–48 to 428)	244,434 (66,750–415,287)
New Hampshire	15 (6–24)	31 (–7 to 69)	29,809 (8,331–50,976)
New Jersey	222 (90–351)	651 (–204 to 1,482)	551,606 (154,435–934,428)
New Mexico	43 (17–68)	90 (–16 to 194)	97,395 (25,805–166,932)
New York	317 (125–505)	979 (–493 to 2,414)	862,754 (245,801–1,458,884)
North Carolina	115 (44–184)	268 (–104 to 633)	320,964 (86,513–548,951)
North Dakota*	—	—	—
Ohio	578 (294–858)	1,269 (–150 to 2,653)	799,658 (280,495–1,301,555)
Oklahoma	100 (39–159)	253 (–39 to 535)	215,870 (57,168–368,068)
Oregon	7 (3–10)	9 (–1 to 18)	7,935 (2,420–13,416)
Pennsylvania	728 (385–1,066)	1,450 (–102 to 2,966)	1,033,571 (416,925–1,630,655)
Rhode Island	30 (12–48)	80 (–23 to 180)	64,558 (18,247–109,524)
South Carolina	35 (14–57)	76 (–24 to 174)	74,950 (20,214–128,668)
South Dakota	3 (1–6)	12 (–2 to 25)	8,785 (2,335–15,111)
Tennessee	105 (41–169)	277 (–65 to 612)	237,774 (64,768–405,531)
Texas	604 (274–928)	1,691 (–350 to 3,672)	1,936,377 (601,716–3,214,225)
Utah	38 (15–61)	63 (–17 to 142)	161,349 (38,035–281,176)
Vermont	1 (0–1)	1 (0 to 2)	881 (247–1,513)
Virginia	75 (29–121)	233 (–61 to 520)	265,166 (72,871–451,524)
Washington	6 (2–10)	12 (–4 to 28)	19,189 (5,483–32,819)
West Virginia	22 (9–35)	55 (–2 to 111)	32,109 (9,308–54,424)
Wisconsin	103 (41–165)	219 (–72 to 503)	241,479 (65,426–411,854)
Wyoming	5 (2–8)	10 (–2 to 22)	11,486 (3,009–19,800)

Definition of abbreviations: CI = confidence interval; ATS = American Thoracic Society; N/A = not available.
*Indicates that the state already meets ATS recommendations for ozone (O_3) and fine particulate matter ($PM_{2.5}$), and therefore no health estimates are made here.

nationwide estimates, the city-specific information contained in this report provides a valuable tool for air quality managers. It is important to note that these results represent population-level impacts and are not directly representative as an estimate of individual risk. Thus, cities with high estimates of air pollution–related health impacts are typically those with both large populations and elevated pollution concentrations.

There are also additional adverse health impacts from air pollution not included in our analysis, particularly chronic outcomes such as cancer, new-onset asthma, and diabetes (10–12). Deaths from these causes, and longer term nonfatal exacerbations of chronic illness, were not captured in this analysis but would add to the total health burden of air pollution outcomes.

This report uses a similar approach as the EPA to estimate health impacts attributable to air pollution. We limited health functions to the subset of those used

Table 2. Cities with the most to gain by attaining the ATS recommendations for O_3 and $PM_{2.5}$*

Rank	City	Avoided Deaths	Avoided Morbidities	Fewer Impacted Days
1	Los Angeles (Long Beach-Glendale), CA	1,341	3,255	2,892,029
2	Riverside (San Bernardino-Ontario), CA	808	1,416	1,321,762
3	New York (Jersey City-White Plains), NY-NJ	282	977	818,666
4	Phoenix (Mesa-Scottsdale), AZ	283	598	636,730
5	Pittsburgh, PA	285	533	281,858
6	Fresno, CA	260	672	390,551
7	Bakersfield, CA	241	333	220,722
8	Houston (The Woodlands-Sugar Land), TX	229	661	636,211
9	Cleveland (Elyria), OH	196	487	231,859
10	Cincinnati, OH-KY-IN	173	298	192,989
11	Dallas (Plano-Irving), TX	142	431	572,502
12	San Diego (Carlsbad), CA	132	281	381,631
13	Sacramento (Roseville-Arden-Arcade), CA	128	244	304,876
14	Modesto, CA	130	262	177,349
15	Philadelphia, PA	126	284	232,031
16	St. Louis, MO-IL	119	267	253,545
17	Visalia (Porterville), CA	117	185	117,646
18	Baltimore (Columbia-Towson), MD	103	261	215,646
19	Stockton (Lodi), CA	100	189	120,598
20	Chicago (Naperville-Arlington Heights), IL	91	326	259,480
21	Detroit (Dearborn-Livonia), MI	92	205	151,113
22	Atlanta (Sandy Springs-Roswell), GA	87	239	325,874
23	Fort Worth (Arlington), TX	87	245	287,029
24	Indianapolis (Carmel-Anderson), IN	86	217	150,422
25	Las Vegas (Henderson-Paradise), NV	82	167	214,947
26	Warren (Troy-Farmington Hills), MI	79	211	176,917
27	Nassau County (Suffolk County), NY	78	188	192,408
28	Montgomery County (Bucks County-Chester County), PA	75	157	165,203
29	Denver (Aurora-Lakewood), CO	71	139	222,540
30	Washington (Arlington-Alexandria), DC-VA-MD-WV	67	183	235,256

Definition of abbreviations: ATS = American Thoracic Society; O_3 = ozone; $PM_{2.5}$ = fine particulate matter.
*Overall rankings are a composite of the three individual health categories.

by the EPA in regulatory analysis despite the availability of additional epidemiology studies of sufficient quality to potentially merit consideration for inclusion in this analysis (13–15). One important difference is our decision to include mortality health impact functions in our analysis for both short-term (16) and long-term ozone exposures (17). This is somewhat in contrast to the approach used in the core analysis of the most recent Regulatory Impact Assessment for the O_3 NAAQS, which estimates monetary benefits only for short-term exposures; monetary benefits due to long-term exposure is included only as an additional sensitivity analysis (*see* Section 6.3, available at: https://www3.epa.gov/ttn/naaqs/standards/ozone/data/20151001ria.pdf).

Unlike what is observed for mortality impact associated with $PM_{2.5}$, where impacts from long-term pollution exposures are much greater than those observed in time-series studies of mortality risk from short-term exposure (18), the magnitude of mortality risk estimated in this analysis for short- and long-term O_3 exposures is highly similar. The combined central estimate for excess mortality was 7,033 from short-term O_3 exposures and 6,123 from long-term O_3 exposures. Even though it is not the focus of this analysis, this observation is noteworthy given the limited understanding of the relative contributions to adverse health outcomes between short-term and long-term exposures for different pollutants (19).

There are a number of caveats to consider when interpreting the results of this analysis. Health impacts are estimated only in counties with valid pollution monitoring for the years 2011–2013. For counties without a pollution monitor, health impacts are not estimated here even though concentrations may exceed the ATS-recommended levels. In addition, some counties have pollution monitors that provided measurements that were later invalidated (e.g., $PM_{2.5}$ monitoring in Illinois). Without valid pollution concentrations we were unable to accurately estimate the potential impacts of air quality in these counties.

In some regions of the United States, short-term $PM_{2.5}$ concentrations frequently exceed the current EPA 24-hour standard of 35 $\mu g/m^3$, despite having annual concentrations below the ATS recommendation of 11 $\mu g/m^3$. As a result of the decision to only estimate health impacts in counties that exceed the recommended annual standard, health estimates for $PM_{2.5}$ are not available in this study for cities that exceed the 24-hour but not the annual standard. Therefore, there are health effects from short-term increases in $PM_{2.5}$ above the daily standard of 35 $\mu g/m^3$ that are not included in this analysis. This analysis also does not include the public health benefits expected with further improvements in outdoor air quality below the ATS-recommended standards. Use of this approach does not imply that further health benefits would not be achieved by still further reductions in pollution levels. In the 2013 rulemaking adopting the revised annual $PM_{2.5}$ NAAQS standard, the EPA explained that there is no epidemiological evidence of a threshold below which $PM_{2.5}$ effects do not occur (20).

The ATS recommendations for O_3 and $PM_{2.5}$, which are the basis for the health estimates in this report, are more stringent than the current NAAQS determined by the EPA. Although the numeric difference between the ATS-recommended standards and current NAAQS may seem small, the expected health benefits of more stringent standards are substantial. For example, full compliance with the current O_3 standard, revised to 0.070 ppm in 2015, would result in approximately 2,650 avoided deaths and 7,560,000 fewer impacted days, which is substantially less than the 6,410 avoided deaths and 17,000,000 fewer impacted days expected in meeting the ATS recommendation for O_3 (*see* data table from www.HealthoftheAir.org). However, regardless of actions taken by the EPA in setting national standards, cities should be encouraged to improve air quality to avoid the adverse health impacts that have been quantified in this report.

This analysis will be updated as new data become available, taking into account the most recent pollution data and estimates of population growth. The annual release of the Marron Institute–ATS "Health of the Air" report will also serve as a marker for public health progress as cities work to reduce ambient pollution concentrations. ■

Author disclosures are available with the text of this article at www.atsjournals.org.

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