The benefits of lower ozone due to air pollution emissions reductions (2002-2011) in the Eastern US during extreme heat

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ABSTRACT

Using the Community Multiscale Air Quality (CMAQ) model and the Benefits Mapping and Analysis Program – Community Edition (BenMAP-CE) tool, we estimate the benefits of anthropogenic emissions reductions between 2002 and 2011 in the Eastern United States (US) with respect to surface ozone concentrations and ozone-related health and economic impacts, during a month of extreme heat, July 2011. Based on CMAQ simulations using emissions appropriate for 2002 and 2011, we estimate that emissions reductions since 2002 likely prevented 10 - 15 ozone exceedance days (using the 2011 maximum eight hour average ozone standard of 75 ppbv) throughout the Ohio River Valley and 5 - 10 ozone exceedance days throughout the Washington, DC – Baltimore, MD metropolitan area during this extremely hot month. CMAQ results were fed into the BenMAP-CE tool to determine the health and health-related economic benefits of anthropogenic emissions reductions between 2002 and 2011. We estimate that the concomitant health benefits from the ozone reductions were significant for this anomalous month: 160-800 mortalities (95% confidence interval (CI): 70-1,010) were avoided in July 2011 in the Eastern U.S, saving an estimated \$1.3 – \$6.6 billion (CI: \$174 million – \$15.5 billion). Additionally, we estimate that emissions reductions resulted in 950 (CI: 90-2,350) less hospital admissions from respiratory symptoms, 370 (CI: 180-580) less hospital admissions for pneumonia, 570 (CI: 0-1650) less Emergency Room (ER) visits from asthma symptoms, 922,020 (CI: 469,960-1,370,050) less minor restricted activity days (MRADs), and 430,240 (CI: -280,350-963,190) less symptoms of asthma exacerbation during July 2011.

KEYWORDS

Air Quality
Ozone
Emissions
CMAQ
BENMAP

IMPLICATIONS

We estimate the benefits of air pollution emissions reductions on surface ozone concentrations and ozone-related impacts on human health and the economy between 2002 and 2011 during an extremely hot month, July 2011, in the eastern United States (US) using the CMAQ and BenMAP-CE models. Results suggest that, during July 2011, emissions reductions prevented 10-15 ozone exceedance days in the Ohio River Valley and 5-10 ozone exceedance days in the Mid Atlantic; saved 160-800 lives in the Eastern US, saving $1.3 - $6.5 billion; and resulted in 950 less hospital admissions for respiratory symptoms, 370 less hospital admissions for pneumonia, 570 less Emergency Room visits for asthma symptoms, 922,020 less minor restricted activity days, and 430,240 less symptoms of asthma exacerbation.

INTRODUCTION

Exposure to ozone has been linked to premature mortality (Bell et al., 2004; Bell et al., 2005), exacerbation of respiratory symptoms, such as asthma (Anenberg et al. 2018), and increases in hospital admissions and emergency room visits (Burnett et al., 1997; Medina-Ramon et al., 2006). Consequently, anthropogenic emissions of nitrogen oxides ($NO_x = NO + NO_2$) have been controlled over the last few decades in the eastern United States (US) with a goal to lower concentrations of surface ozone (US EPA, 2018). A primary driver of these decreases was the NO_x State Implementation Plan (SIP) Call Rule, finalized in 1998 and fully implemented by May 31, 2004, which required 22 states and the District of Columbia (D.C.) to regulate NO_x emissions under the US Environmental Protection Agency (EPA) NO_x Budget Trading Program. Nearly 100% compliance for major sources was achieved by 2008 (US EPA, 2009). Since then, the EPA Clean Air Interstate Rule (CAIR), as well as additional court-orders and state regulations, have gone into effect to continue to reduce NO_x and improve surface air quality. As reported by the EPA's National Emissions Inventory (NEI), NO_x emissions decreased by 46% between 2002 and 2011 over the US east of the Mississippi River (available at https://www.epa.gov/sites/production/files/2018-04/national_tier1_caps.xlsx).

Estimates of the decreases in surface nitrogen dioxide (NO_2) from both satellite and in situ observations are 20-55% over US cities from 2005 to 2015 (Lamsal et al., 2013, 2015; Tong et al., 2015; Duncan et al., 2016), coinciding with a ~15% decrease in ozone as estimated from surface data from EPA's Air Quality System (AQS). Over the Eastern US, satellite observations of NO_2 tropospheric columns from the Ozone Monitoring Instrument (OMI), which serve as a proxy for surface concentrations in polluted areas (Leue et al., 2001; Velders et al., 2001), show widespread reductions between 2005 and 2011 (Figure 1), including decreases up to 30-50% over parts of the Ohio River Valley and major cities throughout the Mid-Atlantic and Northeast regions (Duncan et al., 2016; Krotkov et al., 2017). One of the largest NO_2 reductions between 2005 and 2012 occurred over Washington, D.C., with a reduction of 47% and 48% as inferred from data from OMI and AQS, respectively (Tong et al., 2015). The majority of the decreases in OMI tropospheric NO_2 column over the Eastern US occurred from 2005-2008, with little or no decreases after 2009 (Russell et al., 2012; Lamsal et al., 2013, 2015; Krotkov et al., 2016; Duncan et al., 2016; Tong et al., 2015; Jiang et al., 2018). These decreases are also reflected in the AQS observations (Duncan et al., 2013; Tong et al., 2015; Krotkov et al., 2016). However the AQS data show additional decreases after 2009 (Silvern et al., 2019), consistent with the NEI.

Hot temperatures and stagnant air, which are expected to become more common in the future in the US (e.g. Mickley et al., 2004), are conducive to the formation of high levels of surface ozone and, thus, have the potential to confound the effectiveness of NO_x reduction efforts. A correlation of high temperature and events with high ozone concentrations has been confirmed with air quality model simulations (Weaver et al., 2009; Jacob and Winner, 2009) and observations (Bloomer et al., 2009, 2010).

In this analysis, we show that the benefits from historical anthropogenic air pollution emissions (i.e., NO_x, volatile organic compounds (VOCs), and particulate matter (PM)) reductions between 2002 and 2011 are substantial for surface ozone and human health for July 2011. This month was chosen because of its extreme heat. July 2011 was the fourth hottest month on record in the Eastern US and the hottest month on record for the Baltimore-Washington metropolitan area. Therefore, the extreme heat experienced during this month represents the maximum benefit of emission reductions that may be expected. We chose the lower bound, 2002, of our study period to investigate the impact of emissions changes due to regulations discussed above, which occurred after 2002, and because of the availability of the US EPA's 2002 National Emissions Inventory (NEI). In addition, we were able to leverage modeling results from the NASA DISCOVER-AQ (Deriving Information on Surface Conditions from Column and Vertically Resolved Observations Relevant to Air Quality) field campaign, which took place in July 2011, for this study (Loughner et al., 2014).

In the next section, we describe our sensitivity simulation experiment which isolates the benefit of historical anthropogenic emissions changes by removing uncertainty in the role of meteorological variability on ozone. We also describe the tool used to calculate the health and economic benefits of these emissions reductions, the US EPA Environmental Benefits Mapping and Analysis Program – Community Edition v1.1 (BenMAP-CE; Sacks et al. 2018). After the Methods Section, a discussion of the results of our sensitivity simulation experiment and health and economic benefit analysis is presented. Concluding remarks follow the Results Section.

METHODS

Observations of temperature and ozone

We analyze observed temperature and ozone data to understand their relationships to anthropogenic NO_x emissions as reported by EPA and how their relationships have changed over our study period. We use historical temperature data observed at the Baltimore Washington International Thurgood Marshall Airport (BWI) and surface ozone reported to AQS for Maryland monitoring stations (Figure S1).

CMAQ model simulations

We performed two sensitivity simulations using the Community Multi-scale Air Quality (CMAQ; Byun and Schere, 2006) model to remove the impact of meteorological variability on ozone, such that a comparison between two model simulations reveals only the benefit of the historical anthropogenic emissions changes on ozone. (Therefore, we refer to this sensitivity analysis as "ozone avoided" sensitivity analysis hereafter). For the first simulation (hereafter referred to as CMAQ11), we simulated July 2011 using anthropogenic emissions (NO_x, VOCs, and PM) appropriate for 2011. For the second (hereafter referred to as CMAQ02), we used emissions appropriate for 2002 to reflect the conditions that would have been experienced in the absence of emission reductions. Both simulations used the same meteorology generated by a simulation of the Weather Research and Forecasting (WRF) model (Skamarock et al., 2008) and biogenic and lightning emissions for July 2011. WRF and CMAQ are standalone models with WRF meteorological model output fed into the CMAQ model, so feedback from the chemistry to the meteorology is ignored. Biogenic emissions were calculated with the Biogenic Emissions Inventory System (BEIS; Vukovich and Pierce, 2002) and lightning emissions were calculated within CMAQ from lightning flash count data from the National Lightning Detection Network and convective precipitation output from WRF (Allen et al., 2012). Results presented here are from a 12 km domain covering the Northeast and parts of the South and Midwest US that is nested inside a 36 km domain covering the Continental US (Figure 2). Details on the implementation, model inputs, evaluation, and uncertainties of the CMAQ simulation using anthropogenic emissions appropriate for 2011 are found in Loughner et al. (2014), Goldberg et al. (2014), He et al. (2014), Flynn et al. (2014), and Anderson et al. (2014). We used the best available emissions for 2002 and 2011 at the time of performing the CMAQ model simulations. CMAQ anthropogenic emissions inputs appropriate for 2002 were created by processing the EPA 2002 NEI with the Sparse Matrix Operator Kernel Emissions (SMOKE) modeling system (Houyoux and Vukovich, 1999).

Our analysis may underestimate the total benefits from air pollution reductions on ozone as our simulation methodology does not account for an increase in electricity demand during unusually hot weather, which plays a major role in increasing power plant emissions and air pollution levels (He et al., 2013; Farkas et al., 2015; Abel et al., 2017). He et al. (2013) found that one third of an increase in ozone concentrations during unusually hot summertime weather is due to an increase in power plant emissions from an increase in energy demand. Farkas et al. (2015) found that temporally misallocating emissions from the electricity sector can underestimate emissions.

BenMAP-CE

We used our CMAQ model output as input into the BenMAP-CE software to estimate the health and economic benefits of the historical air pollution emissions reductions during July 2011. To calculate the health impact (e.g., change in the incidence of a particular health outcome) BenMAP-CE uses population data, air quality data or model output (CMAQ simulated ozone for this study), and Health Impact Functions (HIF) based on epidemiological studies, which relate an increased risk of a negative health outcome to a unit change in exposure to a particular air pollutant. Using these inputs, BenMAP-CE calculates a delta that represents the change in the air pollutant concentration of interest. Health impacts are estimated using HIFs that relate a change in the concentration of a particular air pollutant to a change in the incidence, or occurrence, of a given health outcome, such as mortality or hospital admissions due to pneumonia. The HIF used in this study has the following functional form:

$$\Delta Y = Y_0\left(1 - e^{-\beta \Delta \mathrm{AQ}}\right) * Population \qquad (1)$$

where ΔY is the change in incidence, Y_0 is the baseline incidence, ΔAQ is the change in the air quality concentration of interest, and population is the exposed population (Sacks et al., 2018). For this analysis, population data for 2011 were used. The β coefficient represents a percent change in incidence per unit change in air pollution and is derived from epidemiological studies. Economic valuations for particular health impacts are calculated based on an estimated value of a statistical life (VSL), or using willingness-to-pay (WTP) or cost-of-illness (COI) estimates. For detailed information about the BenMAP-CE software, see Sacks et al. (2018).

RESULTS

Observed relationship between ozone, temperature, and precursor emissions

The decreasing trend in the number of ozone exceedance days per year in Maryland since 1980 correlates well with decreases in anthropogenic NO_x and VOC emissions with a correlation coefficient of 0.86 for NO_x and 0.81 for VOC emissions (Figure 3). However, this decrease is not reflected in the number of days per year when temperature observed 2 m above ground level (2 m temperature) exceeded 90°F at BWI (Figure 3). Note that while July 2011 was the hottest month on record at BWI airport, the summer of 2011 was not the hottest summer on record; therefore, 2011 does not show the highest number of days above 90°F. During the 1980s and 1990s, there were typically more than twice the number of ozone exceedance days than days ≥ 90°F. This trend diminished in the 2000s as air pollution regulations on NO_x and VOC emissions were implemented. In 2002, there were 48 days with temperatures ≥ 90°F and 67 exceedance days, while in 2011, there were 59 days with temperatures ≥ 90°F and 43 exceedance days. The first year there were consistently more days with temperatures ≥ 90°F than ozone exceedance days was 2009.

The relationship between high temperatures and surface ozone concentrations changed over time because of the reductions in NO_x concentrations (Bloomer et al., 2009) (Figure 4). Temporal differences begin to emerge in the relationship between temperature versus ozone concentrations at about >70°F and maximize above 90°F. For example, for a daily maximum temperature of 90°F, the average daily 1 hour ozone maximum was 115 ppbv between 1983 and 1987 and 75 ppbv between 2013 and 2017, a difference of 40 ppbv.

"Ozone avoided" CMAQ sensitivity analysis

Our "ozone avoided" sensitivity analysis (CMAQ02 – CMAQ11) indicates that, in the absence of emissions reductions, the Eastern US would have experienced substantially higher maximum daily 8 hour average ozone > 75 ppbv (hereafter, referred to as "MDA8 ozone") throughout the modeling domain (Figure 5). At the 95th percentile (i.e., high ozone day; Figure 5, top row), the CMAQ02 simulation shows MDA8 ozone exceeding 90 ppbv in the Ohio River Valley, the Washington, DC – Boston, MA urban corridor, and urban areas in the South, 15-20 ppbv higher than seen in CMAQ11. At the 50th percentile (i.e., median ozone concentrations for the month of July, middle row), CMAQ02 shows simulated MDA8 ozone of 70-80 ppbv in the Ohio River Valley and Mid-Atlantic, an increase up to 15 ppbv over CMAQ11 in the Ohio River Valley and up to 10 ppbv in the Mid-Atlantic. The sensitivity analysis also shows improvements in ozone concentrations on relatively clean days (5th percentile, bottom row), by about 5 ppbv in the Ohio River Valley.

The CMAQ02 simulation has higher MDA8 ozone and more ozone exceedance days than the CMAQ11 simulation (Figure 6). In fact, in parts of the Ohio River Valley and Mid-Atlantic, more ozone exceedance days than clean days occur in the CMAQ02 simulation. The results of our sensitivity experiment indicate that emission reductions prevented ~10-15 ozone exceedance days in the Ohio River Valley and ~5-10 ozone exceedance days in the Washington, DC – Baltimore, MD metropolitan area.

The observed OMI tropospheric NO_2 column decreases (i.e., 20-55% over parts of the Ohio River Valley and major cities throughout the Mid-Atlantic and Northeast) between 2005 and 2011 (Figure 1) are similar to the simulated differences between the two model runs (Figure 7). The simulated decreases in the tropospheric NO_2 column are 50% less in the Ohio River Valley, where many power plants are located, and 20-40% less in the Washington, DC to Boston, MA urban corridor.

Ozone concentrations are biased high in the CMAQ11 simulation, which may lead to an overestimation of the health and economic benefits described in the next section. An in-depth evaluation of the CMAQ11 simulation is examined in Loughner et al. (2014), Goldberg et al. (2014), He et al. (2014), Flynn et al. (2014), and Anderson et al. (2014). The mean bias, normalized mean bias, root mean square error, and normalized mean error as defined by Eder and Yu (2006) for the CMAQ11 simulation as compared with maximum 8 hour average ozone concentrations at AQS monitoring locations is 5.9 ppbv, 11.6%, 14.5 ppbv, and 22.0%, respectively. Uncertainties in chemistry and emissions partially explain the bias. As described in Goldberg et al. (2014) and Loughner et al. (2014), a high bias in NO_y is present in the CMAQ11 simulation due to uncertainties in the conversion rates of NO_z to NO_2 within the CB05 chemical mechanism used in CMAQ. These uncertainties in the conversion rates also may have caused the sensitivity simulation with 2002 emissions (CMAQ02) to simulate too many ozone exceedance days. In addition, NO_x concentrations were biased high due to a high bias in the mobile emissions inventory (Anderson et al., 2014). These biases result in CMAQ simulating more ozone exceedance days with respect to high temperatures as observed. The ratio of the number of simulated and observed ozone exceedance days in Maryland to days when the temperature reached 90°F at BWI are 0.986 and 0.71, respectively.

Quantification of health and economic benefits of ozone concentration reductions from anthropogenic emissions reductions

MDA8 ozone for July 2011 from the CMAQ02 and CMAQ11 model simulations were used as inputs to the BenMAP-CE software to estimate the health and economic impacts of reductions in ozone between 2002 and 2011. The average MDA8 delta for the 12 km resolution domain for July 2011 was 5.68 ppb (min: 2.14, max: 9.41) (Figure 5). Table 1 lists all mortality related health outcomes and associated epidemiological studies used in this analysis. Table 2 lists the same information for all other health outcomes. The health impact functions used in this analysis were chosen based on their use in the 2008 EPA Ozone NAAQS Regulatory Impact Analysis (US EPA 2008). Incidences for each HIF were calculated using the Monte Carlo method within BenMAP-CE and pooled where appropriate. For this option, BenMAP-CE generates percentiles of the incidence distribution that reflect the variability in the analysis inputs.

We estimate that between 160 and 800 deaths (95% Confidence Interval (CI): 70-1,010) were avoided because of the lower ozone values that resulted from the reduction of anthropogenic emissions (Table 3). These numbers represent a total over all 22 states and

Washington, D.C. that lie completely within the simulation domain. Figure 8 shows the spatial distribution of these results for the mortality estimates using the HIF from Bell et al. (2004). Pennsylvania and Ohio show the greatest benefits at 60 (CI: 30-90) and 60 (CI: 30-90), respectively. The distribution of incidence will not necessarily reflect the distribution of the change in air quality (Figure 6) because it is also a function of population density and baseline incidence (i.e., the presence of a health endpoint that is typically found in the population) (Fann et al., 2011). Breakdowns of the estimates for individual states are presented in Tables S1 and S2. Estimated avoided mortalities associated specifically with cardiopulmonary disease were 230 (CI: 110-350). These avoided deaths equated to a savings of an estimated \$1.3 – \$6.5 billion (CI: \$173 million – \$15.5 billion). Mortality valuations are based on the VSL and represent the amount of money that the population is willing to pay to avoid one death (US EPA, 2018). All valuation methods chosen for this analysis are methods used by the US EPA for regulatory impact analyses and are listed in Table S3.

Our results are higher than previous reported estimates from Fann and Risley (2013), who calculated nationwide avoided mortality estimates from changes in ozone and $PM_{2.5}$ using interpolated air quality monitoring data from 2000-2007. Using two different epidemiological studies, they reported maximum year-to-year estimates of 560 (Bell et al., 2004, CI: 430-680) and 2,600 (Levy et al., 2005, CI: 2,300-2,800) over the five-month ozone season of May 1 to September 30. If we crudely assume an equal number of nationwide avoided mortalities per month, these equal ~112 and ~520, respectively, which are lower than our estimates of 160 and 800 for our simulation domain (Figure 2). We expect our estimates to be higher however, as they reflect extremes with respect to both temperature and ozone levels.

With respect to non-mortality health endpoints (Table 4), we estimated that emissions reductions resulted in 950 less hospital admissions for respiratory symptoms (370 for pneumonia specifically), 570 less Emergency Room (ER) visits for asthma symptoms, 922,020 less minor restricted activity days (MRADs), and 430,240 less symptoms of asthma exacerbation. Minor restricted activity days are days during which individuals reduce or limit their activities, but do not miss work or school (Hubbell et al., 2005). Similar to Figure 8, Figure 9 shows the spatial distribution of estimated avoided incidences of exacerbation of asthma symptoms. Ohio shows the maximum estimate, at 43,420. Reductions in incidence of these health endpoints were valued from \$0.25 million (ER visits) to ~\$60 million (MRADs) (Table 4). Valuation methods for non-mortality health endpoints vary, but are typically calculated using willingness-to-pay (WTP) or cost-of-illness (COI) estimates. WTP represents the amount of money that the population is willing to pay to reduce risk, whereas COI reflects actual medical costs or loss of earnings (US EPA, 2018). Specific valuations methods used in this analysis are listed in Table S3.

These results are not meant to reflect an actual estimation of the number of avoided health outcomes, such as those based on observations of pollutants. They are meant to illustrate the benefit of emissions reductions using our 'ozone avoided' sensitivity analysis for an extreme month (July 2011). Changes in incidence are highly sensitive to air quality inputs, and should be taken as a hypothetical estimate.

CONCLUSIONS

In this study, we estimate the air quality, health, and economic benefits of historic anthropogenic pollution emission reductions from 2002 to 2011 over the Eastern US, showing that they are substantial during a historically hot month, July 2011. Despite these substantial reductions, AQS observations show that Maryland exceeded the 2011 ozone standard on 14 days in July 2011. Using CMAQ-generated sensitivity simulations, we estimate that the impact of anthropogenic emissions reductions on surface ozone was extensive over the Eastern US with 10-15 fewer ozone exceedance days (MDA8 ozone) during July 2011 throughout the Ohio River Valley and 5-10 fewer ozone exceedance days in the Washington, DC – Baltimore, MD metropolitan area. In Maryland, an analysis of surface ozone and temperature data indicates that the decreasing trend in surface ozone from 1980 to 2017 is not significantly associated with changes in surface temperature as the number of days with high (≥ 90ºF) temperatures has not significantly changed over our study period, though there are significant variations from year to year.

Using the EPA BenMAP-CE tool for July 2011, we estimate that the reduction in ozone attributed to the anthropogenic emissions reductions resulted in 160-800 avoided mortalities (95% CI: 70-1010 lives) in 22 states and the District of Columbia during July 2011. These avoided mortalities were valued at a savings of \$1.3 – \$6.5 billion (CI: \$173 million – \$15.5 billion). Further, we estimate that the implementation of emissions reductions avoided 950 hospital admissions for respiratory symptoms (370 for pneumonia specifically), 570 ER visits for asthma symptoms, 922,020 MRADs, and 430,240 symptoms of asthma exacerbation. These non-mortality health endpoints were valued to be a savings of $0.25 million (ER visits) to ~$60 million (MRADs). As stated above, these estimates are meant to represent a hypothetical "ozone avoided" sensitivity analysis, and do not represent a calculation of health and economic benefits due to an observed change in air quality.

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DISCLOSURE STATEMENT

The scientific results and conclusions, as well as any views or opinions expressed herein, are those of the authors and do not necessarily reflect the views of NOAA or the Department of Commerce.

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REFERENCES

Abel, D., T. Holloway, R.M. Kladar, P.Meier, D. Ahl, M. Harkey, and J. Patz. 2017. Response of power plant emissions to ambient temperature in the eastern United States. *Environ. Sci. Technol.* 51: 5838-5846.

Allen, D.J., K.E. Pickering, R.W. Pinder, B.H. Henderson, K.W. Appel, and A. Prados. 2012. Impact of lightning-NO on eastern United States photochemistry during the summer of 2006 as determined using the CMAQ model. *Atmos. Chem. Phys.* 12: 1737-1758.

Anderson, D.C., C.P. Loughner, A. Weinheimer, T.P. Canty, R.J. Salawitch, H. Worden, A. Fried, T. Mikoviny, A. Wisthaler, and R.R. Dickerson. 2014. Measured and modeled CO and NOy in DISCOVER-AQ: An evaluation of emissions and chemistry over the eastern US, *Atmos. Environ.* 96:78-87.

Anenberg, S.C., D.K. Henze, V. Tinney, P.L. Kinney, W. Raich, N. Fann, C.S. Malley, H. Roman, L. Lamsal, B. Duncan, R.V. Martin, A. van Donkelaar, M. Brauer, R. Doherty, J.E. Jonson, Y. Davila, K. Sudo, and J.C.I. Kuylenstierna. 2018. Estimates of the Global Burden of Ambient $PM_{2.5}$, Ozone, and NO_2 on Asthma Incidence and Emergency Room Visits. *Environ. Health Persp.*. 126(10): 107004. doi:10.1289/EHP3766.

Bell, M.L., A. McDermott, S.L. Zeger, J.M. Samet, and F. Dominici. 2004. Ozone and short-term mortality in 95 US urban communities, 1987-2000. *JAMA.* 292: 2372-8.

Bell, M.L., F. Dominici and J.M. Samet. 2005. A meta-analysis of time-series studies of ozone and mortality with comparison to the national morbidity, mortality, and air pollution study. *Epidemiology.* 16: 436-45.

Bloomer, B.J., J.W. Stehr, C.A. Piety, R.J. Salawitch., and R.R. Dickerson. 2009. Observed relationships of ozone air pollution with temperature and emissions, *J. Geophys. Res.* 36: L09803, doi:10.1029/2009GL037308.

Bloomer, B.J., R.R. Dickerson, and K. Vinnikov. 2010. A chemical climatology and trend analysis of ozone and temperature over the eastern US. *Atmos. Environ.* 44: 2543-2551.Burnett R.T., J.R. Brook, W.T. Yung, R.E. Dales, and D. Krewski. 1997. Association between Ozone and Hospitalization for Respiratory Diseases in 16 Canadian Cities. *Environ. Res.* 72:24-31.

Burnett R.T., J.R. Brook, W.T. Yung, R.E. Dales, and D. Krewski. 1997. Association between Ozone and Hospitalization for Respiratory Diseases in 16 Canadian Cities. *Environ. Res.* 72:24-31.

Byun, D. and K.L. Schere. 2006. Review of the governing equations, computational algorithms, and other components of the Models-3 Community Multiscale Air Quality (CMAQ) modeling system. *Appl. Mech. Rev.* 59:51-77.

Duncan, B.N., L.N. Lamsal, A.M. Thompson, Y. Yoshida, Z. Lu, D.G.Streets, M.M. Hurwitz, and K.E. Pickering. 2016. A space based, high-resolution view of notable changes in urban NOx pollution around the world (2005-2014). *J. Geophys. Res. Atmos.* 121:976-996, doi:10.1002/2015JD024121.

Duncan, B. N., Y. Yoshida, B. de Foy, L. N. Lamsal, D. G. Streets, Z. Lu, K. E. Pickering, and N. A. Krotkov. 2013. The observed response of Ozone Monitoring Instrument (OMI) NO_2 columns to NO_x emission controls on power plants in the United States: 2005-2011. *Atmos. Environ.* 81:102-111. doi:10.1016/j.atmosenv.2013.08.068.

Eder, B. and S. Yu. 2006. A performance evaluation of the 2004 release of Models-3 CMAQ. *Atmos. Environ.* 40:4811-24. doi:10.1016/j.atmosenv.2005.08.045.

Fann, N., A.D. Lamson, S.C. Anenberg, K. Wesson, D. Risley, and B.J. Hubbell. 2011. Estimating the national public health burden associated with exposure to ambient PM2.5 and ozone. *Risk Anal.* 32:81-95. doi: 10.1111/j.1539-6924.2011.01630.x

Fann, N and D. Risley. 2013. The public health context for $PM_{2.5}$ and ozone air quality trends. *Air Qual. Atmos. Health.* 6: 1-11. doi:10.1007/s11869-010-0125-0.

Farkas, C.M., M.D. Moeller, F.A. Felder, K.R. Baker, M. Rodgers, and A.G. Carlton. 2015. Temporalization of peak electric generation particulate matter emissions during high energy demand days. *Environ. Sci. Technol.* 49: 4696-4704.

Flynn, C., K.E. Pickering, J.H. Crawford, L. Lamsal, N. Krotkov, J. Herman, A. Weinheimer, G. Chen, X. Liu, J. Szykman, S.-C. Tsay, C.P. Loughner, J. Hains, P. Lee, R.R. Dickerson, J.W. Stehr, and L. Brent. 2014. Relationship between column-density and surface mixing ratio: Statistical analysis of O_3 and NO_2 data from the July 2011 Maryland DISCOVER-AQ mission. *Atmos. Environ.* 92:429-441.

Goldberg, D.L., C.P. Loughner, M. Tzortziou, J.W. Stehr, K.E. Pickering, L.T. Marufu, and R.R. Dickerson. 2014. Higher surface ozone concentrations over the Chesapeake Bay than over the adjacent land: Observations and models from DISCOVER-AQ and CBODAQ campaigns. *Atmos. Environ.* 84:9-19.

He, H., L. Hembeck, K.M. Hosley, T.P. Canty, R.J. Salawitch, and R.R. Dickeron. 2013. High ozone concentrations on hot days: The role of electric power demand and NO_x emissions, *Geophys. Res. Lett.* 40: 5291-5294.

He, H., C.P. Loughner, J. Stehr, H. Arkinson, L. Brent, M. Follette-Cook, M.A. Tzortziou, K.E. Pickering, A. Thompson, D.K. Martins, G. Diskin, B. Anderson, J.H. Crawford, A. Weinheimer, P. Lee, J. Hains, and R.R. Dickersion. 2014. An elevated reservoir of air pollutants over the Mid-Atlantic States during the 2011 DISCOVER-AQ campaign: Airborne measurements and numerical simulations. *Atmos. Environ.* 85:18-30.

Houyoux, M.R. and J.M. Vukovich. Updates to the Sparse Matrix Operator Kernel Emissions (SMOKE) Modeling System and Integration with Models-3. Paper presented at The Emission Inventory: Regional Strategies for the Future, Air and Waste Management Association, Raleigh, NC, October, 26-28, 1999.

Huang, Y., F. Dominici and M.L. Bell. 2005. Bayesian hierarchical distributed lag models for summer ozone exposure and cardio-respiratory mortality. Environmetrics. 16: 547-562.

Hubbell B.J., A. Hallberg, D.R. McCubbin, and E. Post. 2005. Health-related benefits of attaining the 8-hr ozone standard. *Environ. Health Perspect*. 113:73-82.

Ito, K., S.F. De Leon, and M. Lippmann. 2005. Associations between ozone and daily mortality: analysis and meta-analysis. *Epidemiology.* 16: 446-57.

Jacob, D.J., D.A. Winer. 2009. Effect of climate change on air quality. *Atmos. Environ.* 43: 51-63.

Jiang, Z., B.C. McDonald, H. Worden, J.R. Worden, K. Miyazaki, Z. Qu, D.K. Henze, D.B.A. Jones, A.F. Arellano, E.V. Fischer, L. Zhu, K.F. Boersma. 2018. Unexpected slowdown of US pollutant emission reduction in the past decade. *P. Natl. Acad. Sci.* 115:5099-5105. doi:10.1073/pnas.1801191115.

Krotkov, N.A., L.N. Lamsal, E.A. Celarier, W.H. Swartz, S.V. Marchenko, E.J. Bucsela, K.L. Chan, and M. Wenig. 2017. The version 3 OMI NO_2 standard product. *Atmos. Meas. Tech.* 10:3133-3149. doi:10.5194/amt-2017-44.

Krotkov, N.A., C.A. McLinden, C. Li, L.N. Lamsal, E.A. Celarier, S.V. Marchenko, W.H. Swartz, E.J. Bucsela, J. Joiner, B.N. Duncan, K.F. Boersma, J.P. Veefkind, P.F. Levelt, V.E. Fioletoz, R.R. Dickerson, H. He, Z. Lu, and D.G. Streets. 2016. Aura OMI observations of regional SO_2 and NO_2 pollution changes from 2005 to 2015. *Atmos. Chem. Phys.* 16:4605-4629. doi:10.5194/acp-16-4605-2016.

Lamsal, L.N., B.N. Duncan, Y. Yoshida, N.A. Krotkov, K.E. Pickering, D.G. Streets, and Z. Lu. 2015. U.S. NO_2 trends (2005-2013): EPA Air Quality System (AQS) data versus improved observations from the Ozone Monitoring Instrument (OMI), *Atmos. Environ.* 110:130-143. doi:10.1016/j.atmosenv.2015.03.055.

Lamsal, L.N., R.V. Martin, D.D. Parrish, and N.A. Krotkov. 2013. Scaling relationship for NO_2 pollution and urban population size: A satellite perspective. *Environ. Sci. Technol.* 47:7855-7861. doi:10.1021/es400744g.

Leue, C., M. Wenig, T. Wagner, O. Klimm, U. Platt, and B. Jähne. 2001. Quantitative analysis of NO_x emissions from Global Ozone Monitoring Experiment satellite image sequences. *J. Geophys. Res.* 106:5493-5505.

Levy, J.I., S.M. Chemerynski, and J.A. Sarnat. 2005. Ozone exposure and mortality: an empiric bayes metaregression analysis. *Epidemiology.* 16:458-68.

Loughner, C.P., M. Tzortziou, M. Follette-Cook, K.E. Pickering, D. Goldberg, C. Satam, A. Weinheimer, J.H. Crawford, D.K. Knapp, D.D. Montzka, G.B. Diskin, L.T. Marufu, and R.R. Dickerson. 2014. Impact of bay breeze circulations on surface air quality and boundary layer export. *J. Appl. Meteor. Clim.* 53:1697-1713.

Medina-Ramón M, A. Zanobetti, J. Schwartz. 2006. The Effect of Ozone and PM10 on Hospital Admissions for Pneumonia and Chronic Obstructive Pulmonary Disease: a national multicity study. *Am. J. Epidemiol.* 163:579-588.

Mickley, L.J., D.J. Jacob, B.D. Field, and D. Rind. 2004. Effects of future climate change on regional air pollution episodes in the United States. *Geophys. Res. Lett.* 31:L24103. doi:10.1029/2004GL021216.

Moolgavkar, S.H., E.G. Luebeck, and E.L. Anderson. 1997. Air pollution and hospital admissions for respiratory causes in Minneapolis St. Paul and Birmingham. *Epidemiology.* 8:364-370.

Mortimer, K.M., L.M. Neas, D.W. Dockery, S. Redline, and I. B. Tager. 2002. The effect of air pollution on inner-city children with asthma. *Eur. Respir. J.* 19:699-705.

O'Connor, G.T., L. Neas, B. Vaughn, M. Kattan, H. Mitchell, E.F. Crain, R. Evans III, R. Gruchalla, W. Morgan, J. Stout, G.K. Adam, and M. Lippmann. 2008. Acute respiratory health effects of air pollution on children with asthma in US inner cities. *J. Allergy Clin. Immun.* 121:1133-1139.

Ostro, B.D. and S. Rothschild. 1989. Air Pollution and Acute Respiratory Morbidity - an Observational Study of Multiple Pollutants. *Environ. Res.* 50:238-247.

Peel, J.L., P.E. Tolbert, M. Klein, K.B. Metzger, W.D. Flanders, K. Todd, J.A. Mulholland, P.B. Ryan and H. Frumkin. 2005. Ambient air pollution and respiratory emergency department visits. *Epidemiology.* 16:164-74.

Russell, A.R., L.C. Valin, and R.C. Cohen. 2012. Trends in OMI NO_2 observations over the United States: Effects of emission control technology and the economic recession. *Atmos. Chem. Phys.* 12:12197-12209. doi:10.5194/acp-12-12197-2012.

Sacks, J.D., J.M. Lloyd, Y. Zhu, J. Anderton, C.J. Jang, B. Hubbell, and N. Fann. 2018. The Environmental Benefits Mapping and Analysis Program – Cummunity Edition (BenMAP-CE): A tool to estimate the health and economic benefits of reducing air pollution. *Environ. Modell. Softw.* 104:118-129.

Schildcrout, J.S., L. Sheppard, T. Lumley, J.C. Slaughter, J.Q. Koenig and G.G. Shapiro. 2006. Ambient air pollution and asthma exacerbations in children: an eight-city analysis. *Am. J. Epidemiol.* 164:505-517.

Schwartz, J. 1994a. Air Pollution and Hospital Admissions For the Elderly in Detroit, Michigan. *Am. J. Resp. Crit. Care.* 150:648-655.

Schwartz, J. 1994b. PM(10) Ozone, and Hospital Admissions For the Elderly in Minneapolis St Paul, Minnesota. *Arch. Environ. Health.* 49:366-374.

Schwartz, J. 1995. Short term fluctuations in air pollution and hospital admissions of the elderly for respiratory disease. *Thorax.*50:531-538.

Schwartz, J. 2005. How sensitive is the association between ozone and daily deaths to control for temperature? *Am. J. Respir. Crit. Care.* 171:627-31.

Silvern, R.F., D.J. Jacob, L.J. Mickley, M.P. Sulprizio, K.R. Travis, E.A. Marais, R.C. Cohen, J.L. Laughner, S. Choi, J. Joiner, and L.N. Lamsal. 2019. Using satellite observations of tropospheric NO_2 columns to infer long-term trends in US NO_x emissions: the importance of accounting for the free tropospheric NO_2 background. *Atmos. Chem. Phys. Discuss.* https://doi.org/10.5194/acp-2019-168, in review.

Skamarock, W.C., J.B. Klemp, J. Dudhia, D.O. Gill, D.L. Barker, M.G. Duda, X.-Y. Huang, W. Wang, J.G. Powers. 2008. A description of the Advanced Research WRF Version 3. NCAR Technical Note. NCAR/TN-475+STR. NCAR, Boulder, CO.

Tong, D.Q., L. Lamsal, L. Pan, C. Ding, H. Kim, P. Lee, T. Chai, K.E. Pickering, and I. Stajner. 2015. Long-term NO_x trends over large cities in the United States during the great recession: Comparison of satellite retrievals, ground observations, and emission inventories. *Atmos. Environ.* 107:70-84.

US EPA (US Environmental Protection Agency). 2008. Final Ozone NAAQS Regulatory Impact Analysis. EPA-452/R-08-003. Research Triangle Park, NC:US EPA, Office of Air Quality Planning and Standards.

US EPA (US Environmental Protection Agency). 2009. The NOx Budget Trading Program: 2008 Highlights. EPA-430-R-09-026. https://www.epa.gov/sites/production/files/2015-09/documents/2008_highlights.pdf.

US EPA (US Environmental Protection Agency). 2018. Environmental Benefits Mapping and Analysis Program - Community Edition: User's Manual, Updated for BenMAP-CE Version1.4.8. Available: https://www.epa.gov/sites/production/files/2015-04/documents/benmap-ce_user_manual_march_2015.pdf

Velders, G.J.M., C. Granier, R.W. Portmann, K. Pfeilsticker, M. Wenig, T. Wagner, U. Platt, A. Richter, and J.P. Burrows. 2001. Global tropospheric NO_2 column distributions: Comparing 3-D model calculations with GOME measurements. *J. Geophys. Res.* 106:12, 643-12,660.

Vukovich, J.M. and T. Pierce. 2002. The implementation of BEIS-3 within the SMOKE modeling framework. *Proc. 11th Int. Emission Inventory Conf.*, Atlanta, Ga, EPA, 7 pp. [Available online at https://www.epa.gov/ttn/chief/conference/ei11/modeling/vukovich.pdf]

Weaver, C.P., and Coauthors. 2009. A preliminary synthesis of modeled climate change impacts on U.S. regional ozone concentrations. *Bull. Am. Meteorol. Soc.* 90: 1843-1863.

Wilson, A.M., C.P. Wake, T. Kelly and J.C. Salloway. 2005. Air pollution, weather, and respiratory emergency room visits in two northern New England cities: an ecological time-series study. *Environ Res.* 97:312-21.

FIGURES

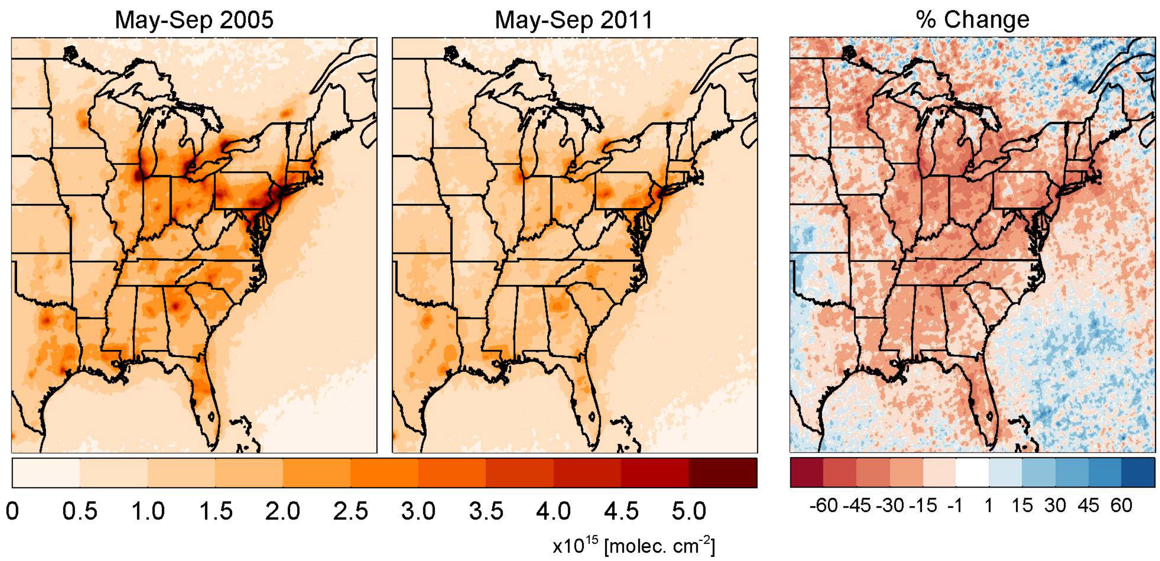

Figure 1. OMI tropospheric NO_2 vertical column density averaged for the ozone season of May through September for the Eastern US in 2005 (left) and 2011 (middle). The percent change from 2005-2011 is shown on the right.

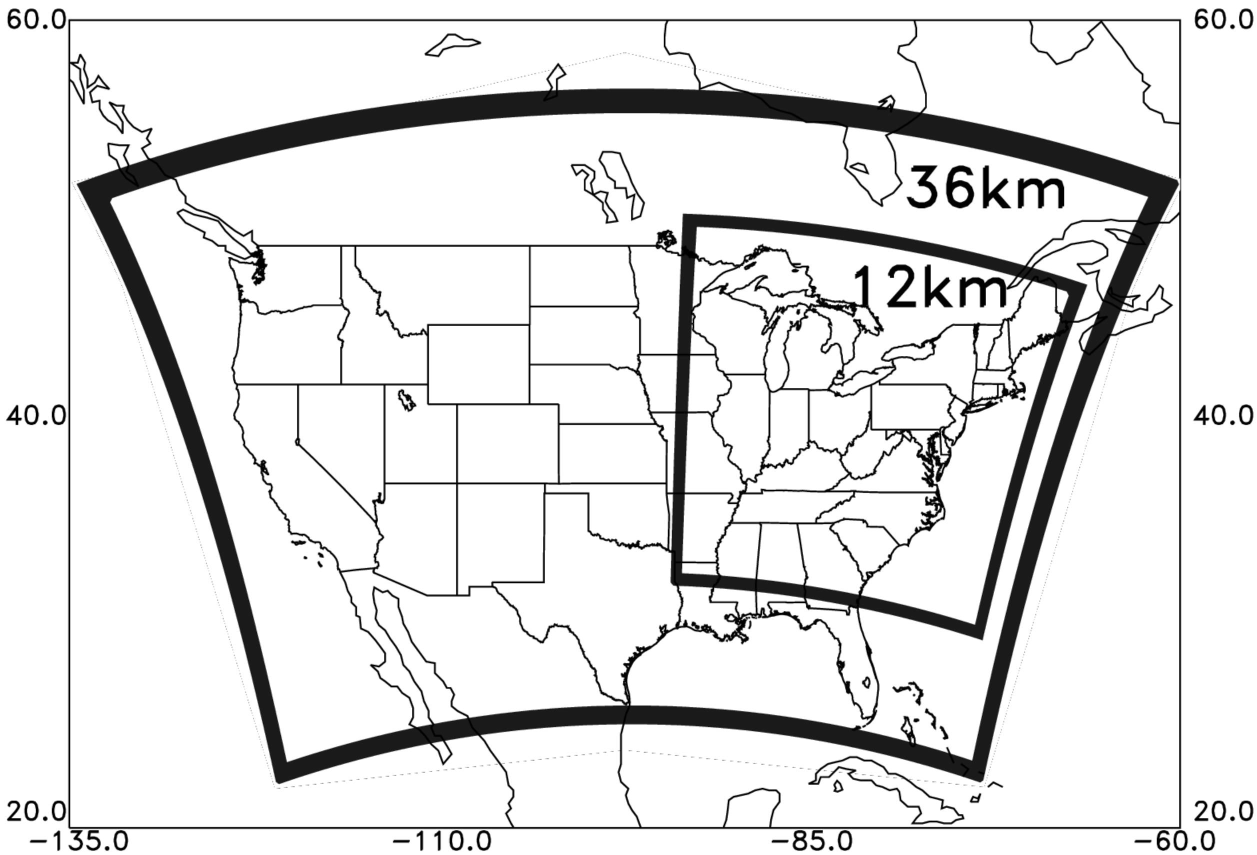

Figure 2. Diagram showing the 36 and 12 km modeling domains. Model results presented in this study are from the 12 km modeling domain.

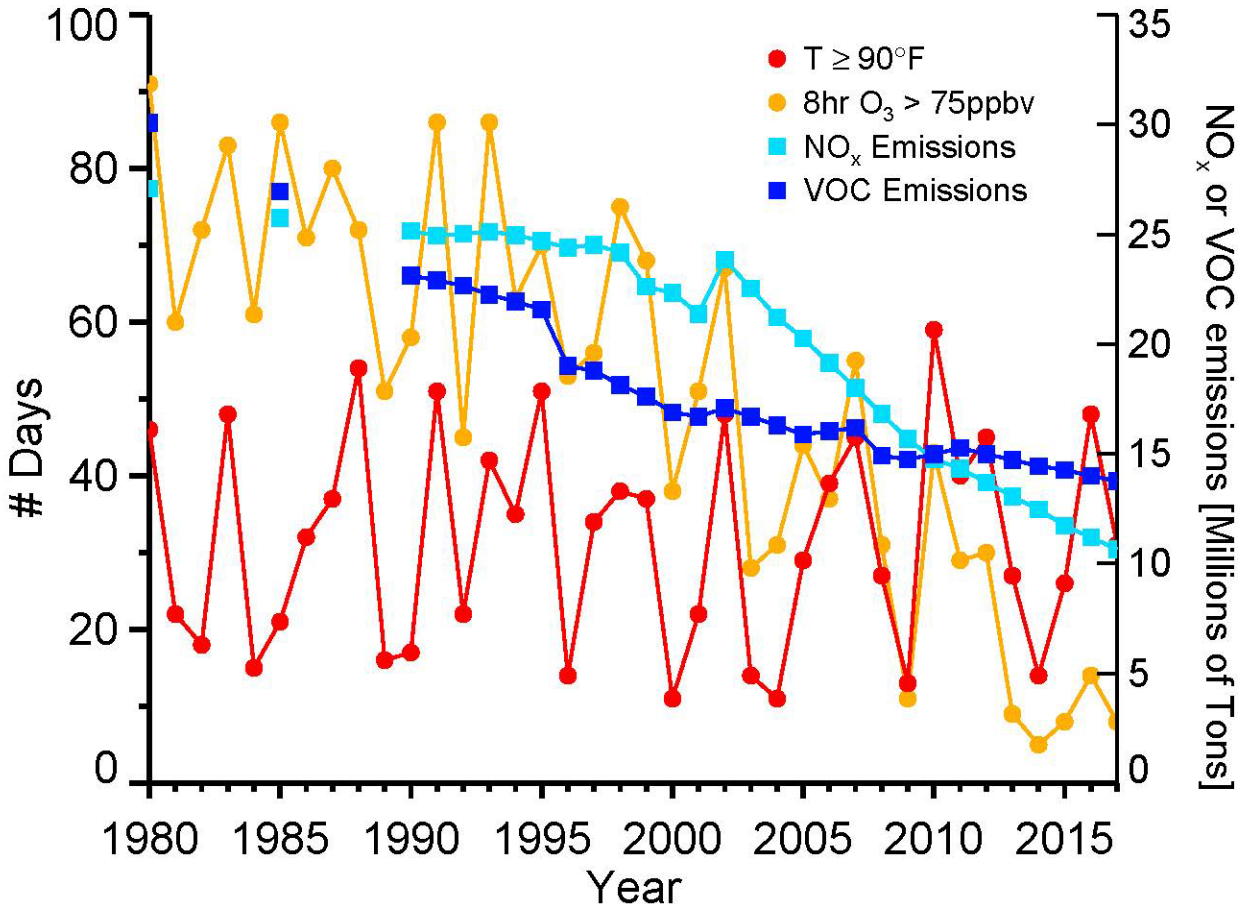

Figure 3. Number of days daily maximum temperature reached 90°F or above at BWI airport (red), number of days MDA8 ozone exceeded the current EPA ozone standard (MDA8 > 75ppbv) from all monitored data collected in Maryland (orange), and annual NO_x (light blue) and VOC (dark blue) emissions in the US as reported by the EPA.

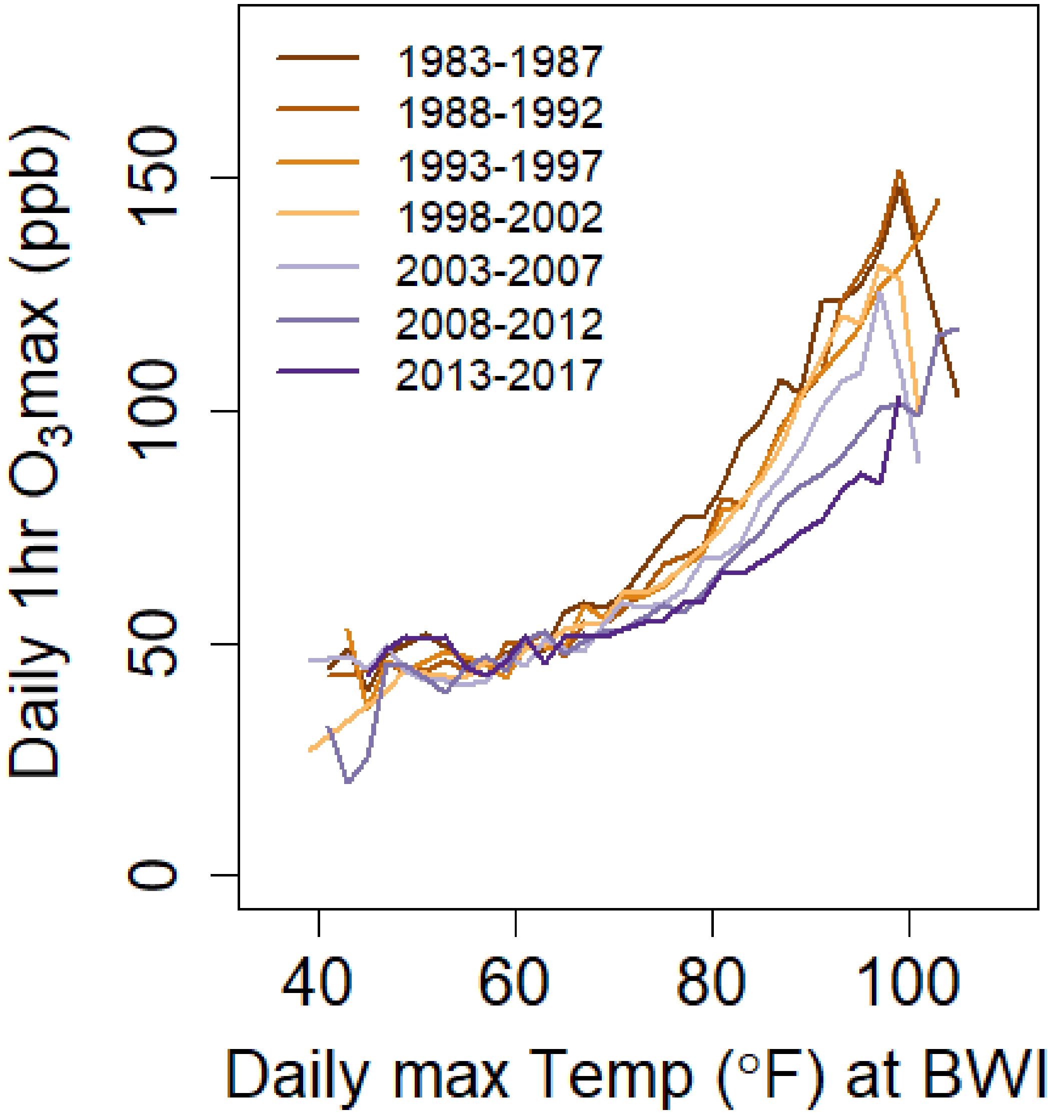

Figure 4. The relationship between daily 1 hour maximum ozone observed in Maryland and daily maximum temperature at BWI airport during the months of May through September. Data placed in 2 degree bins and each bin averaged in 5 year increments.

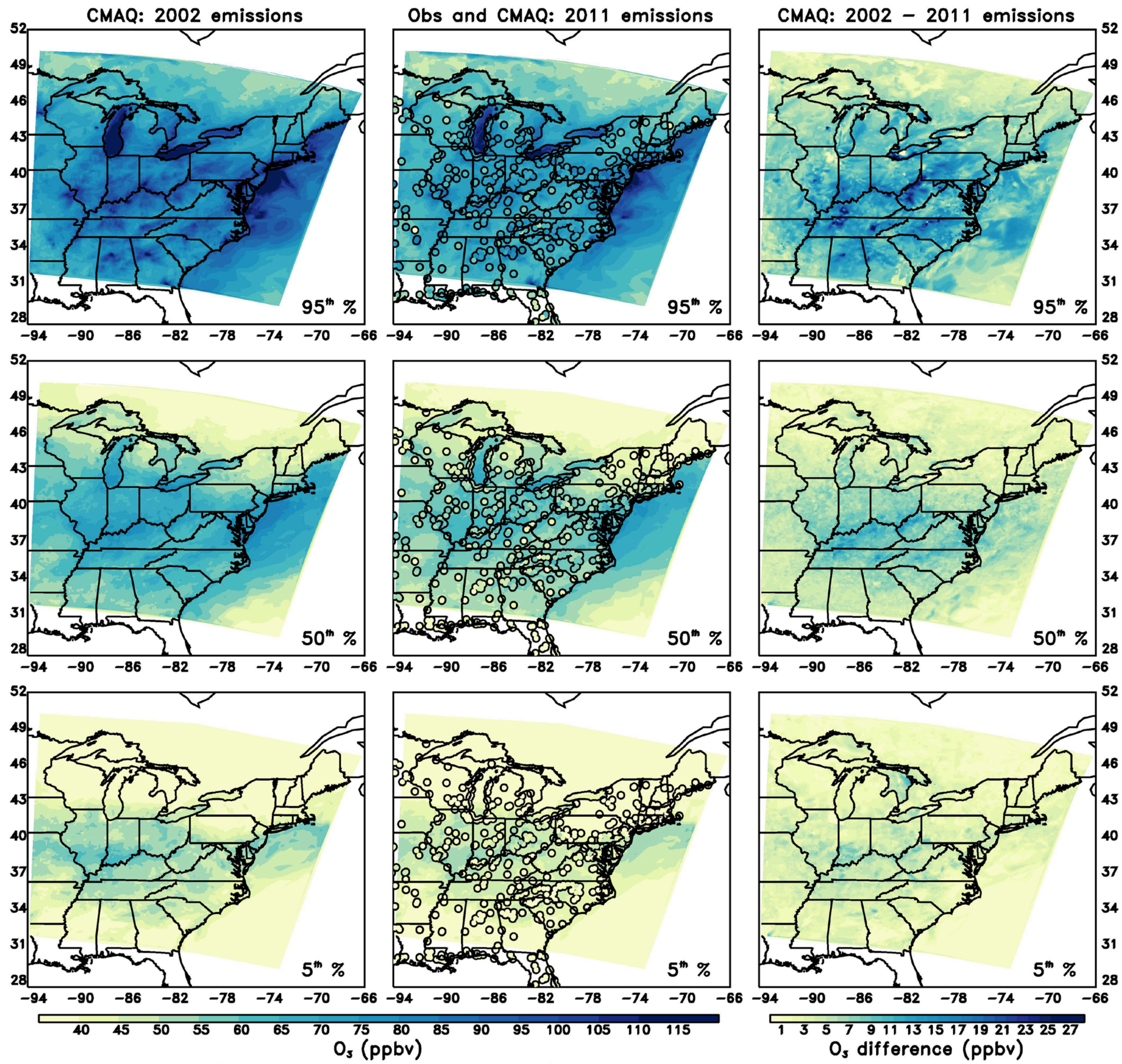

Figure 5. The 95th (top), 50th (middle), and 5th (bottom) percentile of maximum 8 hour average ozone concentrations for the month of July 2011 for CMAQ02 (left), CMAQ11 (middle), and CMAQ02-CMAQ11 (right). Results are from the 12 km CMAQ domain. Corresponding EPA AQS O_3 observations are shown on the CMAQ11 plots.

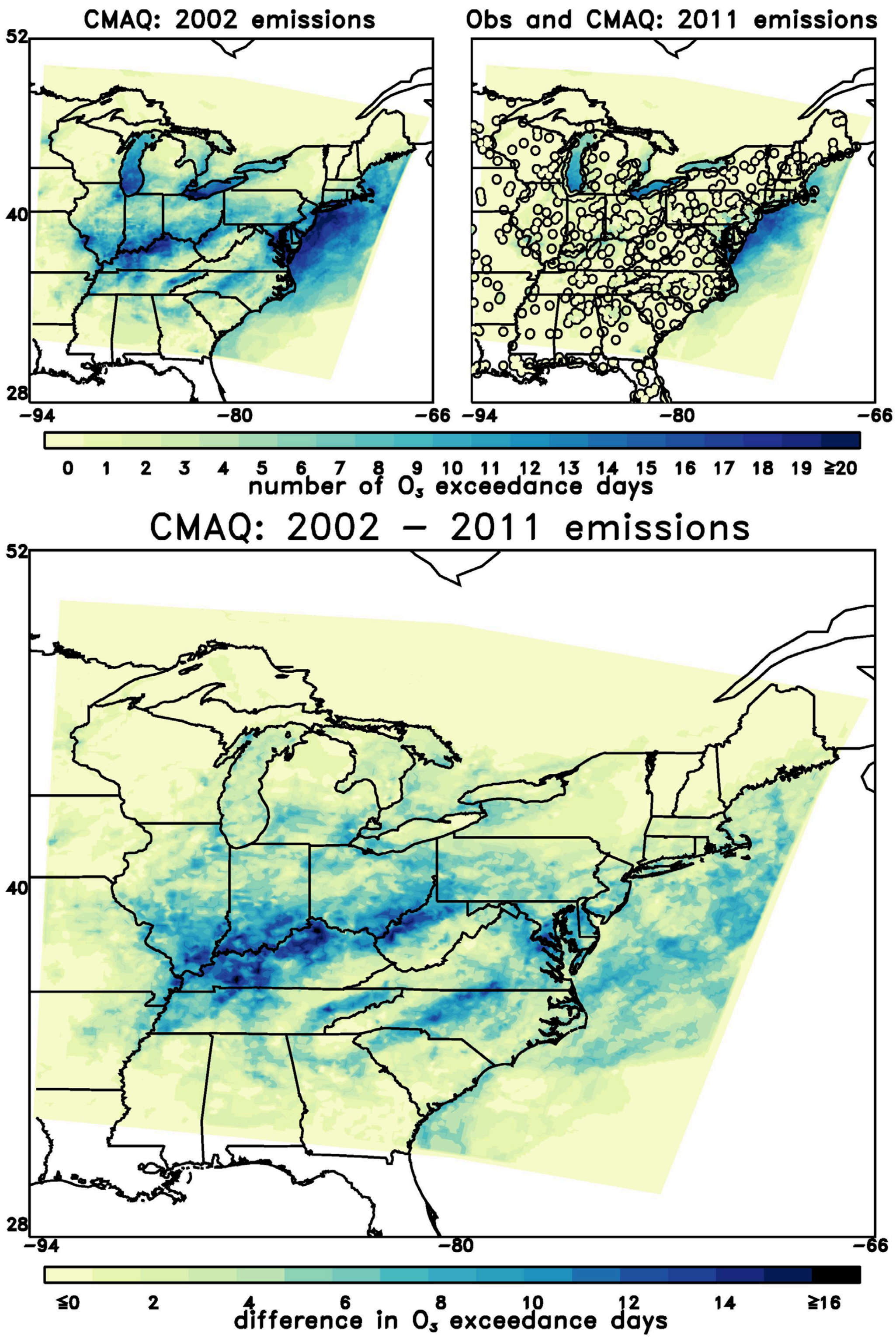

Figure 6. Number of ozone exceedance days (MDA8 ozone > 75 ppbv) for July 2011 from CMAQ02 (upper left), CMAQ11 with EPA AQS O_3 observations overlayed (upper right), and the CMAQ02-CMAQ11 difference in number of ozone exceedance days (bottom). Results are from the CMAQ 12 km domain.

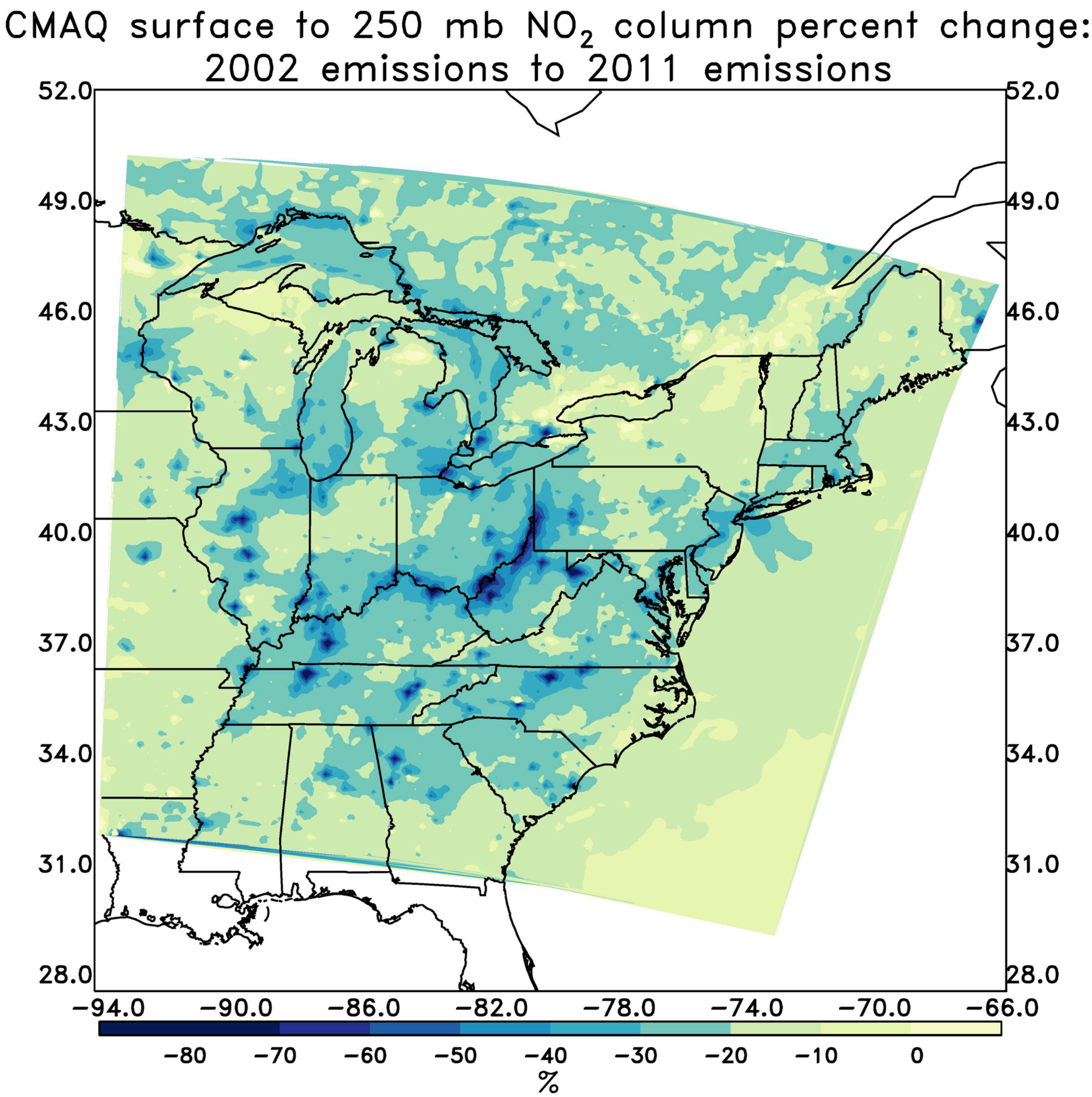

Figure 7. Percent change in July 2011 CMAQ simulated surface to 250 mb NO_2 column from CMAQ02 to CMAQ11.

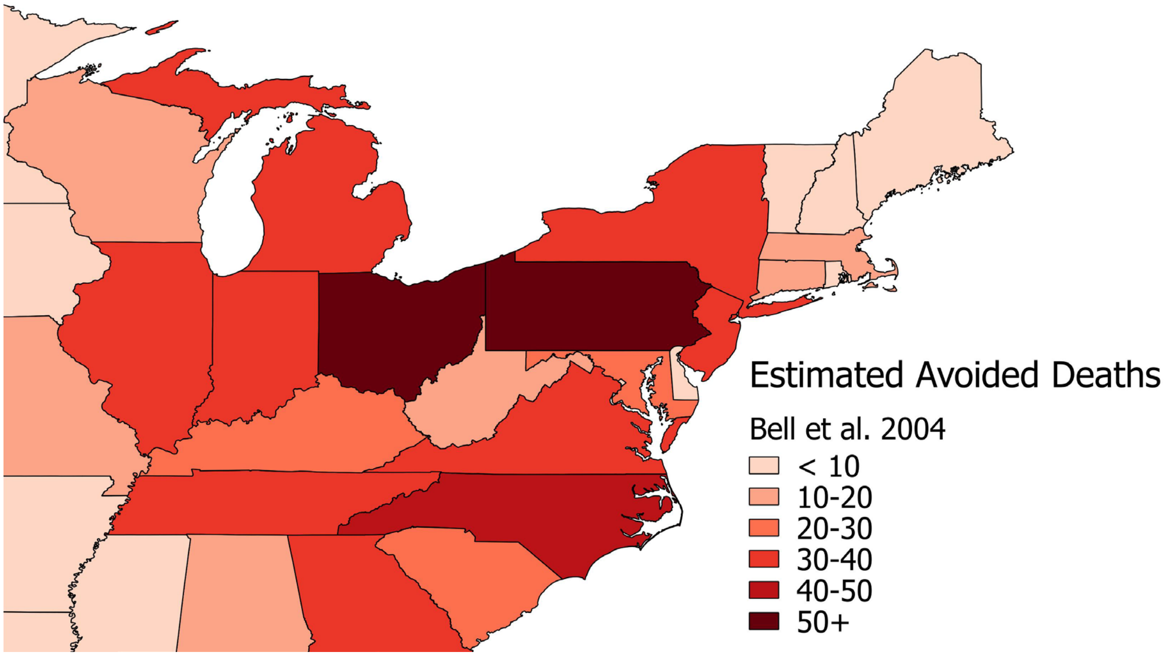

Figure 8. The number of estimated avoided mortalities during July 2011 due to anthropogenic emissions reductions between 2002 and 2011. The estimate shown here was generated using the C-R function from Bell et al. (2004).

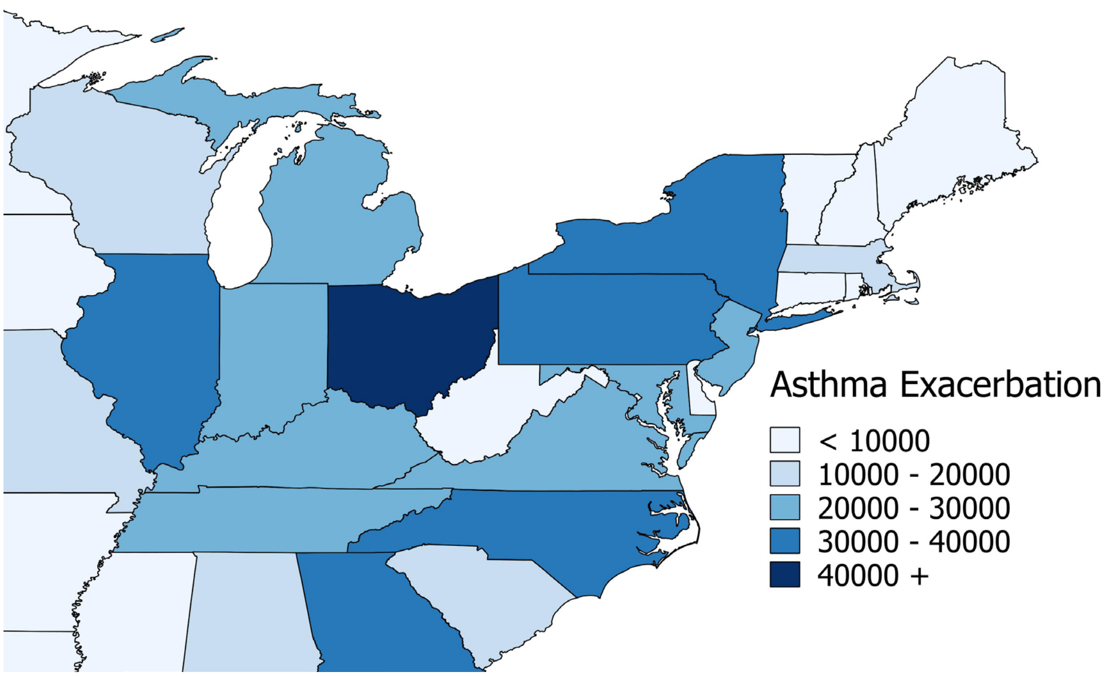

Figure 9. The number of estimated avoided asthma exacerbation incidents during July 2011 due to anthropogenic emissions reductions between 2002 and 2011.

Health Endpoint - Mortality	*C-R Function*	*Age range*	*Beta (Std Err)*
Mortality – All Cause	Levy et al. 2005	0 - 99	0.001119 (0.000179)
	Bell et al. 2004	0 - 99	0.000795 (0.000212)
Mortality – Non-Accidental	Bell et al. 2004	0 - 99	0.000261 (0.000089)
	Ito et al. 2005	0 - 99	0.001173 (0.000239)
	Schwartz et al. 2005	0 - 99	0.000426 (0.000150)
Mortality - Cardiopulmonary	Huang et al. 2005	0 - 99	0.000813 (0.000259)

Table 1. Mortality health studies used in this analysis and their associated references.

Health Endpoint	*C-R Function*	*Age range*	*Beta (Std Err)*
ER Visits – Asthma	*Pooled estimate*		
	Peel et al. 2005	0 - 99	.000870 (.000529)
	Wilson et al. 2005	0 - 99	-0.001000 (0.002000)
Hospital Admissions (HA) – Respiratory	*Pooled estimate*		
	Schwartz et al. 1995	65 - 99	0.001777* (0.000936) 0.004931 (0.001770)
HA – Pneumonia	Moolgavkar et al. 1997	65 - 99	0.00266 (.000762)
	Schwartz et al. 1994a	65 - 99	0.003230 (0.000806)
	Schwartz et al. 1994b	65 - 99	0.002784 (0.001305)
Minor Restricted Activity Days	*Pooled estimate*		
	Ostro and Rothschild 1989	18 - 64	0.002200 (0.000658)
Asthma Exacerbation	*Pooled estimate*		
	Schildcrout et al. (2006)	6 - 18	0.002220 (0.002822)
	O'Connor et al. (2008)	6 - 18	0.000966 (0.002991)
	Mortimer et al. (2002)	6 - 18	0.009288 (0.003872)

Table 2. Health studies and associated references used in this analysis. All pooling was done using the random/fixed effects model (US EPA, 2018).

	Mortality All Cause		*Mortality Non-Accidental*			*Mortality - Cardiopulmonary*
Reference	Bell et al.	Levy et al.	Bell et al.	Ito et al.	Schwartz	Huang et al.
Incidence	569 (319-817)	801 (590-1,011)	159 (69-248)	712 (473-949)	242 (101-381)	232 (110-353)
Valuation (in millions)	4678 (665.56-11,461.7)	6,595.9 (989.70-15,497.2)	1,308.1 (173.94-3,306.21)	5,858.3 (869.39-13,896.3)	1,987.94 (259.81-5,054.94)	1,909.11 (260.23-4,782.17)

Table 3. Total estimated incidence and valuation results for July 2011 for mortality health endpoints for all 22 states included in the simulation domain and Washington D.C. 95% Confidence intervals shown in parentheses.

	HA – All Respiratory	*HA – Pneumonia*	*Emergency Room Visits– Asthma*	*Minor Restricted Activity Days*	*Asthma Exacerbation*
Incidence	950 (91-2,346)	372 (180-579)	573 (0-1,645)	922,018 (469,963-1,370,054)	430,244 (-280,350-963,186)
Valuation (in millions)	31.06 (2.97-76.69)	10.43 (5.05-16.22)	0.25 (-0.00-0.66)	60.83 (28.01-102.27)	24.27 (-15.81-65.22)

Table 4. Same as Table 3 for non-mortality health endpoints.