Climate Change and Your Health: Rising Temperatures, Worsening Ozone Pollution

Technical Appendix

I. Ground-level Ozone Formation

a. Precursor (Ozone-Forming) Emissions

Ground-level ozone (O_3) is produced by a complex set of atmospheric chemical reactions that depend on precursor (ozone-forming) emissions from numerous sources, both natural and of human origin. Ground-level ozone formation is also dependent on weather and climatic factors. Temperature is an important factor in both weather and climate and is a focal point in this report. The primary chemical species involved in ozone formation are nitrogen oxides (NO_x), a combination of nitrogen monoxide (NO) and nitrogen dioxide (NO_2); methane (CH_4); volatile organic compounds (VOCs); and carbon monoxide (CO). Emissions of NO_x due to human activities arise primarily from fossil fuel combustion, with vehicle exhaust, maritime shipping, and power plant energy production contributing the largest amounts (Cofala et al. 2007). Natural global emissions of NO_x are uncertain, but may be of equal magnitude with emissions from human activities. Some of the important natural sources are lightning, emissions from soil bacteria, and forest fires (The Royal Society 2008). Natural VOC sources outweigh sources from human activities and are primarily emitted by terrestrial plants with isoprene being the dominant VOC species (Guenther et al. 2006). Human activity-based VOC sources include fossil fuel combustion sources similar to NO_x, as well as evaporation from solvents and coatings such as paint (Warneke et al. 2007). Methane emissions from human activities include landfills, agriculture activity, and raising livestock while natural sources include emissions from wetlands (NOAA 2010). Methane concentrations in the atmosphere have been roughly constant between 1999 and 2007 (NOAA 2010), but there is a slight indication of an increase through 2010 and there are many uncertainties in how the rate of release from natural sources, such as permafrost, may be affected by climate change (Archer 2007). Carbon monoxide sources include man-made and natural combustion, and emissions from terrestrial vegetation and the oceans. Atmospheric CO concentrations are decreasing overall in the United States (EPA 2011a).

Both NO_x and VOCs are important in the process of forming ground-level ozone and are regulated as precursor pollutants.[1] VOCs, CH_4, and CO react with hydroxyl radical (OH) to form species that then interact with NO_x, ultimately producing ground-level ozone. However, the chemistry that forms ozone from these ingredients is complex enough that in certain conditions an excess of NO_x can lead to a net loss of ozone. This involves NO_2 reacting directly with OH, shutting down the pathways to ozone formation. The reaction is $OH+NO_2+M \rightarrow HNO_3+M$, in which HNO_3 (nitric acid) effectively sequesters NO_2 from further ozone-forming reactions and M is a third body to carry away excess energy from the reaction. These conditions tend to be

found in areas (such as urban or industrial centers) where NO_x emissions are high. However, if CH_4, CO, or VOC levels are also elevated then the normal ozone-forming pathways become important again and lead to overall increases in surface ozone. Also, in areas of high VOC emissions and lower NO_x levels, such as the southeastern United States, ozone formation can be suppressed due to the formation of isoprene nitrates. Isoprene is the primary VOC species and the formation of isoprene nitrates sequesters NO_x, not allowing it to participate in ozone formation. Key aspects of the formation and lifetime of the isoprene nitrates are still uncertain, leading to disagreement between models on overall formation of ozone in these high VOC/low NO_x areas (Wu et al. 2008). Controlling precursor emissions, primarily NO_x and VOC, remains one of the most essential tools in limiting ground-level ozone formation. Largely as a result of measures to meet acid rain reduction goals and National Ambient Air Quality Standards for ground-level ozone, both mandated by the Clean Air Act, the Environmental Protection Agency (EPA) reports a 48 percent decrease in estimated NO_x emissions between 1980 and 2009 (EPA 2011b). These estimated emissions are reported by various state and local air agencies and industry sources with no uncertainty range presented. Over the same time period, VOC emissions decreased by 57 percent. These reductions are for anthropogenic sources and do not account for natural emissions. The decreases in precursor emissions coincide with a 30 percent decrease in eight-hour average ozone concentration between 1980 and 2009 (EPA 2011b). These three percentages are national averages and do not reflect the important regional and local differences in air quality. This may lead to regional ozone trends that can be different from the overall national trends.

b. Weather and Climate Factors

Weather and climatic conditions are also involved in ground-level ozone formation (Jacob and Winner 2009). Some steps in the formation of ozone involve absorption of sunlight by the reacting chemicals. Thus, sunny, cloudless days are more conducive to ozone formation assuming the precursor species are present in sufficient amounts. Wind can also be important in controlling ozone levels as precursor species are dispersed and diluted, typically reducing the ozone-forming reactions. Wind can also disperse ground-level ozone that has already formed, reducing the amounts of, and exposure time to, elevated ozone levels in local generation areas; it can also add to another downwind region that may or may not already have local ozone levels approaching critical health risk levels. Dry deposition to vegetation and soil surfaces is one of the primary removal mechanisms for ground-level ozone and this is dependent on wind speed, among other factors. Precipitation is generally not important for removing ground-level ozone as it is not water soluble.

Temperature was chosen as the focus in determining how weather and especially longer-term climate changes influence ozone formation. Temperature can serve as a proxy for other meteorological conditions, such as stagnation events, conducive for formation of elevated levels of ozone. For instance, stagnation events often occur during periods of elevated temperatures. Temperature also determines rates of chemical reactions important for ozone formation. For example, peroxyacetyl nitrate (PAN) is an important reservoir for odd hydrogen species (OH and HO_2 [hydroperoxyl radical]) that are needed for ground-level ozone production. At higher temperatures, PAN is less stable and frees up these odd hydrogen species to participate in ozone-forming reactions (Sillman, Logan, and Wofsy 1990). Increasing temperatures generally lead to increased natural VOC emissions, which is important in low NO_x areas, such as the southeast US, possibly resulting in lower ozone levels overall.

Longer term climatic changes generally include projected increasing temperatures overall, which are important for ozone formation as discussed above. There are other climatic changes besides temperature that play roles, also. Atmospheric humidity, which is projected to increase overall in a warmer world, leads to increased ozone in high NO_x areas and decreased ozone in low NO_x areas. Ozone is broken apart by sunlight into an oxygen molecule and an excited oxygen atom. The oxygen atom reacts with water molecules in humid air to produce OH. In low NO_x areas this leads to overall ozone destruction. However, in high NO_x regions OH forms hydroperoxyl radical (OOH), which converts NO to NO_2, leading to overall increased ozone production (Murazaki and Hess 2006). Some climatic changes are independent of NO_x levels and are projected to lead to overall increased ozone levels; examples include increased drought, shifts in weather patterns that lead to stagnant air events and heat waves, and possible increases in long-range transport of ozone and ozone precursors. Although ground-level ozone is fairly short-lived (roughly two to three days) and tends to be higher near precursor emissions sources, there have been studies that observe increases in ozone in the western United States due to long-range transport of both precursor species and already-formed ozone from Asia (Cooper et al. 2010).

c. Projected Future Temperatures for the United States

The projections of future climate change used in the report are drawn from a 2009 interagency report, *Global Climate Change Impacts in the United States* (USGCRP 2009). These projections show possible temperature changes for the United States around 2020), mid-twenty-first century, and at the end of the century for two different emissions scenarios. The projections are based on a well-established ensemble of 16 climate models from the Coupled Model Intercomparison Project Three (CMIP n.d.). The “higher” emissions scenario is the IPCC Special

Report on Emissions Scenarios (SRES) A2 scenario[2] that includes increasing heat-trapping gas emissions in the future and general population growth, but regionally oriented economic growth. This leads to per capita economic and technological growth that is slower than some other SRES scenarios. The "lower" emissions scenario is the SRES B1 scenario, which has lower future emissions than A2 and a population that peaks and begins declining in midcentury. It also includes the introduction of cleaner technologies. Carbon dioxide emissions, the primary driver of climate change, for the B1 scenario increase through roughly midcentury and then decline until 2100, while emissions for A2 increase throughout the entire century. For the U.S. temperature projections in the near term (around 2020), the two emissions scenarios (A2 and B1) are not very different, thus the temperature increases associated with each are similar. When presenting the temperature projections for the near term the USGCRP reports a combined average temperature increase of the two scenarios, which is approximately 2°F to 3°F above the baseline temperature average over the years 1961–1979. Further into the century the scenarios diverge in their projected emissions, and the projected temperature profiles differ as well. The higher scenario has a likely warming range of 4°F to 6.5°F by midcentury and 7°F to 11°F by the end of the century. The lower scenario has a likely warming range of 4°F to 6.5°F by the end of the century. The projected temperatures also cover multi-decade time periods with the reported ranges centered on the approximate midpoint of each range. For instance, the midcentury projections are averages of the time period 2040–2059 with the most likely range reported as that for around 2050. We adopt a similar method of reporting in that we use the same midpoint temperature ranges to represent temperature increases in the years 2020 and 2050.

d. Climate Penalty on Ozone Pollution

This report seeks to investigate the relationship between climate change and harmful ground-level ozone. As described above, the pathway to formation of ground-level ozone depends greatly on emissions of precursor gases and various weather and climate factors in a complex relationship. One of the most common metrics for characterizing future climate change is in terms of projected temperature change at both global and regional levels. Temperature is known to affect ozone formation and a survey of modeling studies finds an overall consistently positive dependence of ozone air quality on temperature (Jacob and Winner 2009). These studies hold ozone-precursor emissions constant, but allow climate to vary in an attempt to isolate the changes in ozone due to climate change alone. These studies include other climatic factors in addition to temperature, such as wind speed and humidity. There have also been studies comparing ozone levels and temperature based on measured data. One published study examined 21 years of measured ozone and temperature data for four regions of the eastern

United States and observed an overall linear correlation between the two quantities over a temperature range of 68°F to 95°F (Bloomer et al. 2009). The study used rural data from the Clean Air Status and Trends Network (CASTNET) resulting in more than 3 million valid simultaneous measurements of temperature and ozone. The overall result was that for roughly every degree of warming (°F) in the observed data, there was a corresponding increase of 1.2 parts per billion (ppb) in ozone pollution. This study termed 1.2 ppb/°F the "climate penalty factor." Future climate change will include changes to variables important for ozone levels besides temperature, but Bloomer et al. 2009 suggest that this climate penalty factor can be combined with future temperature projections to quantify possible changes in future air quality. We use this simplifying assumption in this report—that temperature serves as a proxy for overall climate change in approximating changes in future ozone levels.

A survey of some of the published studies that investigate the connections between climate change and ozone levels provided rough estimates of climate penalty factors. The majority surveyed were air quality modeling studies using various models, climate scenarios, time horizons, and regions of focus. Most of the modeling studies surveyed for this report are found in an air quality and climate review article (Jacob and Winner 2009) and of those, we restricted the studies to those involving the United States. There was one study that focused on air quality modeling in the Los Angeles basin and Sacramento Valley (Taha 2001). More such studies are needed before comprehensive regional analysis could be combined into a more realistic national look at the climate penalty on ozone. In addition to the modeling studies we found two studies that derived relationships between temperature and ozone levels based on measured data. Generally, the modeling studies do not report climate penalty factors (ppb/°F) directly, but simply report changes (often in graphical form only) in ozone levels under future climate changes with respect to current or recent levels. We combined the reported changes in ozone (ppb) for the various regions with projected temperature changes (often not included in the individual air quality studies) to determine a rough range of climate penalty factors based on these studies. The studies based on observed data did directly report climate penalty factors, so no combination of ozone changes and temperature projections were needed. We also only looked at the modeling study results that isolated the contribution to ozone levels from climate alone by holding ozone precursor emissions constant. The studies based on measurements inherently accounted for all of the climatic and meteorological factors as well as changing precursor emissions. One study observed a decrease in the climate penalty factor due to reductions in precursor emissions, highlighting the central importance of controlling precursor emissions for reducing ozone (Bloomer et al. 2009).

In order to carry out as broad a comparison as possible between the various published studies we had to account for the differing time horizons and climate scenarios for each of the studies.

To do this we used published future temperature projections for the United States (USGCRP 2009) to estimate the temperature changes for the reported regions in the year for which the air quality studies were carried out. We then combined our estimated temperature changes with the changes in ozone reported for the regions in the various years to arrive at a rough climate penalty factor for that region. In this way, the varying time horizons (i.e., 2020 vs. 2050 vs. 2100) and varying climate scenarios (i.e., well-established scenarios from the Intergovernmental Panel on Climate Change such as A2, B1, or A1B; see IPCC 2000) are accounted for and the estimated climate penalty factors should be roughly time- and climate scenario-independent, leading to a more reasonable comparison between all of the studies. These derived climate penalty factors from the modeling studies should be treated only as estimates, as no error analysis was performed on the data or in the mapping of temperature projections onto reported regional ozone changes. Some of the values were arrived at based on visual inspection, since quantitative regional data were not provided in all cases. We were only attempting to obtain an overall approximation of what range of climate penalty factors could be derived from the literature, realizing the numerous sources of uncertainty involved with our derivation. We also found two studies that reported climate penalty factors based on observed data. They report the slopes of measured ozone data versus measured temperature data, which are the climate penalty factors.

There is a great deal of regional variation in the climate penalty factors derived from the literature and this reflects the regional dependence on future climate (in our case, future temperature projections) as well as the highly regional character to air quality. The climate penalty factors should be treated as rough approximations only. The climate penalty factor estimates are summarized in Table 1.

Most of the studies did not have the resolution to provide information on high population, urban areas and were averaged over larger regions. This likely masks higher climate penalties associated with the more polluted urban areas that reside at scales finer than the model resolutions. The Steiner et al. 2006 study did focus on individual cities and determined climate penalty factors that were at the higher end of the range of values in the table. The modeled values in these cases were lower, however. A limited study confined to the areas in and around Los Angeles modeled a value as high as 15 ppb/°F (Taha 2001). Also, some of the broader region studies do show grid cells with locally elevated ozone levels above the surrounding regions and these generally correspond to larger cities (for example, see Hogrefe et al. 2004). However, our highly averaged analysis likely does not capture these localized “hot spots” with higher populations. The Pacific Northwest region is not included in the table as the studies surveyed here generally do not show a climate penalty. The southeastern United States shows a very small climate penalty factor in one study reported in the table below. Most modeling studies,

Table 1: Climate Penalty Factor Estimates

Locality/Region	Derived Climate Penalty Factor (ppb/°F)	Data Source
Los Angeles Basin	2—4.5	Taha 2001
Sacramento Valley	1.4—2	Taha 2001
Bay Area	1.8*,2.2	Steiner et al. 2006
Sacramento	1.8*,0.6	Steiner et al. 2006
Fresno	2.5*,0.6	Steiner et al. 2006
Midwest urban areas	>3	Hogrefe et al. 2004
Southern California	0.6–1.1	Wu et al. 2008
Texas	0.33–2.7 (combined range of the cited studies)	Wu et al. 2008; Nolte et al. 2008 Bloomer et al. 2009*
Iowa	1.5	Wu et al. 2008
Interior West (portions)	0.6	Wu et al. 2008
Midwest	0.9–1.1 1.7	Wu et al. 2008 Hogrefe et al. 2004
New England	1.7 0.4–0.9 negligible	Hogrefe et al. 2004 Liao et al. 2006 Wu et al. 2008
Mid-Atlantic	0.4–0.9 1 1.3 1.7 0.6 (NY,PA,VA,WV)	Liao et al. 2006 Avise et al. 2009 Wu et al. 2008 Hogrefe et al. 2004 Wu et al. 2008
Eastern United States	0.25–1.67 (lower end of the range represents southeastern United States) 1.2*	Murazaki and Hess 2006 Bloomer et al. 2009

*Denotes climate penalty factor based on measured data. All others are based on modeling results.

however, do not see a climate penalty for this region, although there is some disagreement between models. A large part of the modeling discrepancy seems to reside in how the individual models treat isoprene chemistry (Wu et al. 2008). The Southeast generally has high VOC emissions (with isoprene the most important) and relatively low NO_x emissions, which leads to suppressed ozone formation in future climates, but it depends on how the chemistry is treated. Florida also typically shows no climate penalty factor as its peninsular geographic leads to ventilation processes that do not favor elevated levels of ozone to build. We do not analyze Alabama, Louisiana, Mississippi, Georgia, South Carolina, or Florida, because the climate penalty is either absent, models are inconclusive, or the penalty may in fact be a benefit rather than a penalty. But again it should be noted that certain localized areas in these states,

especially populated urban areas, could see climate penalties (though on smaller scales not captured by the models).

Our report looks at ozone changes from 2007 levels. Temperature changes beginning around 2007 are what need to be used to determine the concurrent changes to ozone. The USGCRP report also presents present-day average U.S. warming (i.e., average over the years 1993–2008) above the 1961–1979 baseline of approximately 1°F. This report scales all of the future projected temperatures down by 1°F to more accurately represent the warming with respect to the year 2007. The year 2007 is chosen as a point of reference, because the increases in ozone in the BenMAP model are with respect to measured ozone levels in 2007. This leads to approximate warming of 1°F to 2°F around 2020 and 3°F to 5.5°F by midcentury for the high scenario and 2°F to 4°F by midcentury for the lower scenario. If the climate penalty metric above is used with these projections this may lead to increases in surface ozone of anywhere from 1.2 ppb to 2.4 ppb by 2020 and 2.4 ppb to 6.6 ppb by around 2050 depending on the scenario. Uncertainties reside within these numbers depending on the many factors outlined above and should only be treated as approximations.

This report used the summary of the climate penalty factors from other published studies described above in conjunction with future temperature projections to determine values of 1 ppb and 2 ppb increases in ground-level ozone above 2007 levels as a plausible scenario for 2020 under future climate change in the United States and 2 ppb and 7 ppb increases for 2050. The two numbers represent the entire range of the low end of the lower scenario up the to the high end of the higher scenario for each year. The values chosen for our modeling runs were rounded to the nearest whole number, as uncertainties in the projected warming did not warrant a decimal place of precision for potential ozone changes. These increases in ground-level ozone are also broadly consistent with the summary range of 1 ppb to 10 ppb from numerous climate-air quality modeling studies, although some of those numbers are reported for the end of the century (Jacob and Winner 2009). Some studies have reported increases of upwards of 30 ppb in the Northern Hemisphere by the end of the century (Prather et al. 2003). Some basic numbers to put the above ozone concentrations into context are that current background ozone levels in the Northern Hemisphere are roughly 35 ppb to 40 ppb. The current EPA standard for permissible levels of ozone is 75 ppb averaged over an eight-hour period. Also, during extreme pollution episodes ozone levels have been as high as 200 ppb (e.g., the 2003 French heat wave) and during high smog episodes in California during the 1960s ozone levels approached 400 ppb (The Royal Society 2008). The increases of 1 ppb, 2 ppb, and 7 ppb in this report appear modest by comparison, but speak to the potentially devastating health costs during extreme events. And while we did not investigate variation in the climate penalty factor directly (we simply used one value), some information on the variation can be obtained by

comparing, say, 1 ppb and 2 ppb for 2020. Although these numbers represent changes in ozone due to range of temperatures they can also be treated as demonstrating the differences between lower and higher climate penalty factors. For a given temperature increase 1 ppb could represent a lower climate penalty factor and 2 ppb could represent a higher climate penalty factor. This sort of comparison can begin to address the regional dependence of the climate penalty on ozone. The 2 ppb results may be more representative of portions of California or the Midwest, which may see climate penalty factors higher than the one we chose for our analysis.

e. Simplifying Assumptions

Due to resource and model constraints our analysis only used one climate penalty factor to model potential changes in ozone levels with future projected increases in temperature and associated health impacts for most of the continental United States. We removed states that the balance of the studies indicated would likely have no future climate penalty on ozone or those for which the models do not generally agree. The chosen factor was weighted toward the longest observational record over the largest number of rural stations, excluding likely higher values known to occur in urban settings.

The analysis holds precursor emissions constant to isolate the effect on ozone due to climate alone. However, these precursor emissions are likely to continue to decrease over time and holding them constant is a simplifying assumption in our analysis. This simplification is common in other published work. A more realistic study on the climate penalty on ozone would carry out air quality modeling based on future climates on a finer spatial scale and determine the climate penalties on ozone that are more specific to individual regions and states.

Based on our publications survey we chose a single climate penalty factor—1.2 ppb/°F—that we evaluated as being representative of large portions of the continental United States. This was the overall value reported by Bloomer et al. 2009 for the most recent time period of observations in that study. It is grounded in observations and does not rely solely on modeling output. The network of data, though not providing coverage across the entire United States, does cover a significant portion of it. It falls within the rough range of values derived from other published studies. We made attempts to reflect some of the regional variability by removing states from the analysis for which there was no clear climate penalty factor. Air quality studies focused on finer scale regions should be carried out for more accurate representations of the climate penalty on ozone for specific regions and states. Also, by imposing one average climate penalty factor we are likely not capturing those areas (i.e., large cities) that could face larger climate penalties.

II. The BenMAP Model

The Environmental Benefits Mapping and Analysis Program (BenMAP), Version 4.0, was used for the health impacts analysis in this report.[3] It was developed by the EPA and is supported by the Center for Environmental Modeling for Policy Development at the University of North Carolina at Chapel Hill to facilitate public support for the model. BenMAP is freely available to the public, but does not have open source code. BenMAP is updated periodically through the release of new versions of the model that, among other things, include newly available peer-reviewed epidemiological and valuation studies used in calculating the health impacts of various air quality scenarios. The BenMAP model was most recently updated in August 2010 (EPA 2010a).

Briefly, BenMAP estimates the health impacts (both in terms of mortality and morbidity incidence and monetary estimates) from changes in air quality. Impacts from reducing individual pollutant species such as ozone, particulate matter, sulfur dioxide, and lead can be determined at national, regional, state, and county levels. Some examples of reductions could be a percentage decrease in lead from current ambient levels in California or a rollback (tightening) of the ozone concentration standard in absolute terms for the entire nation.

The BenMAP analysis includes the following default steps. First, the changes in air quality are generated from baseline and control air quality grids. The baseline represents air pollution without any regulatory policies in place and can be derived from model output or from a database of monitoring sites. The control represents the air pollution with a user-defined change in emissions level—this could be a decrease due to a regulatory policy or an increase due to unchecked growth in emissions. The difference is the change in air pollution. The next step is the health effects incidence estimation due to the change in air pollution. BenMAP contains a population database to determine changes in population-level exposure to air pollution. This includes population projections through 2050 for the entire country, and by state. However, the composition of state populations may change greatly by 2050 as states lose or gain residents. Therefore this report does not report health incidence by state for 2050. Health effects incidence data are then combined with numerous peer-reviewed epidemiological studies containing various pollutant concentration-response (CR) functions to arrive at a change in health effects incidence. This would include premature mortality, days of acute respiratory symptoms, hospital visits, etc. The next step is the economic valuation of the change in health effects incidence. An example of this would be the value in avoided deaths. The avoided deaths in the previous step are multiplied by a valuation function (i.e., value of a statistical life) to arrive at a dollar value for those avoided deaths. Similar valuation exercises are carried out for

hospital admissions, lost school days, emergency room visits, and respiratory symptoms with the appropriate valuation functions drawn from the economic literature, including willingness-to-pay (WTP) studies and cost-of illness studies. (For more information about the valuation functions, see EPA 2010a.) Finally, the incidence and valuation results from the changes in air quality can be summarized in tabular format or GIS-based maps.

While the health incidence projections for ozone are robust through 2050, an accurate monetization of these impacts in BenMAP is constrained to 2024 due to some model limitations. In particular, the economics literature shows that WTP estimates for risk reduction increase with increases in income. Thus an income adjustment factor has to be applied to the monetary estimates for future years to account for this income increase. Version 4.0 of BenMAP only contains this adjustment factor for years through 2024, thus we are only able to present these results for our 2020 run. For the 2050 run, we can calculate monetary benefits but then are unable to adjust them appropriately. Presenting the results without the adjustment would lead to a serious downward bias in health impact costs since U.S. income can be expected to grow quite substantially by 2050. Thus we only present health incidence results for 2050.

Previous peer-reviewed studies have used BenMAP to investigate various aspects of public health and its relation to air quality and climate. Tagaris et al. 2009 projected the health effects of changes in ozone and particulate matter less than 2.5 microns in diameter ($PM_{2.5}$) in 2050 due to future meteorological conditions at the national, state, and county levels. Another study used air quality observations for changes in ozone and $PM_{2.5}$ over the period 2000–2007 to estimate the number of avoided premature mortalities due to changes in air quality (Fann and Risley 2011). BenMAP has been used to determine health impacts of future projected heat-related stress (Voorhees et al. 2011). In addition to these studies and others, BenMAP is also used in the EPA rule-making process for determining National Ambient Air Quality Standards. The ways in which BenMAP is used in these cases is consistent with how it is used in this report.

III. Description of BenMAP Economic Valuation Functions

Assigning dollar values to health costs is a complex endeavor and necessarily requires an indirect monetization of these costs since there is no “market” or “market price” for these health outcomes. One way economists try to “value” health outcomes is by directly asking people how much they would be willing to pay for a small change in a health risk (often called willingness-to-pay or contingent valuation studies). Another frequently used proxy is the direct costs of a hospital stay or emergency room visit. There are, of course, drawbacks to each of these methods. For example, willingness-to-pay studies assume that people have a choice in

making risk trade-offs, but in the case of air pollution the risk is imposed upon them. Also, just using hospital costs does not account for the pain and suffering caused by illness.

The valuation methods and assumptions used in our analysis are all built into the BenMAP model, and are described in more detail below. The values listed below are all in 2000$. The model then coverts them to 2008$ (our chosen base year) and applies an income growth adjustment factor to increase them out to 2020.

1. Risk of premature death: This risk is valued using society's willingness to pay for small reductions in the risk of premature death—or flipped around, in this case, it would be the amount of money that would compensate people for small increases in their risk of death from ozone pollution. The dollar value for this is usually expressed as the value of a statistical life (VSL).[4] VSL is an aggregation of WTP estimates, such that when summed over enough people the small probabilities of death of any one individual add up the expectation of one death in the large group. It should ***not*** be interpreted as the value of an individual person's life (Cameron 2010). The VSL estimate used was $6.3 million, which is a mean estimate calculated from 26 different studies.
2. Hospital stays: The costs include medical costs of a hospital stay caused by respiratory illness and the loss of income for a sick patient unable to work. This does not include any value for pain and suffering. The value used in this study was $15,647 per stay.
3. Emergency room visits (respiratory): This is the cost to the emergency room (or hospital) for treating a patient for respiratory illness. The value used was $261 per visit. Again, it does not include the cost of pain and suffering.
4. Acute respiratory symptoms: Several health problems are associated with air pollution, including upper and lower respiratory symptoms and exacerbation of asthma. For our analysis, we assumed a cost of $98 per day of three or more symptoms caused by air pollution. This is actually a measure of an adult's willingness to pay to avoid such an outcome, derived from the literature.
5. School loss days: For computing the costs of school loss days, the values used are derived from the income lost by a parent who stays home with his or her sick child. This value is imputed to be $75 per day. It does not include any value for pain and suffering and therefore assigns no value to school loss days when no parent stays home.

IV. BenMAP Modeling Methods

This report conducted a custom analysis using the BenMAP Version 4.0 model. We followed the EPA's protocols and default assumptions throughout the analysis (EPA 2009). The following description details step by step how the BenMAP model was used for this work.

a. **Air quality grid creation.** We first created two air quality grids in order to compare the baseline data from 2007 monitors all over the country with a control data set in which we adjusted monitors for a change in 1 ppb, 2 ppb, or 7 ppb in ozone for each of our model runs. (The method for choosing these values is described in a previous section.)
    i. **Control air quality grid data.** We used the monitor rollback setting using the state rollback grid type for ozone pollution using the data set "EPA's standard monitors for O_3" in monitor year 2007. We then rolled all of the monitors back choosing an incremental rollback method for both 1 ppb and 10 ppb (the rollback method can be used to model an increase as well as a decrease because data are log-linear). We chose a background ozone level of 30 ppb (Wang et al. 2009) We chose the Voronoi Neighborhood Averaging method to interpolate data. This is necessary because all grid cells do not have the same level of monitor coverage. Some have multiple monitors and others none. The Voronoi method takes an inverse distance-weighted average of all the monitors surrounding a grid cell center to determine a representative monitor value for the center.
    ii. **Baseline air quality grid data.** The monitor rollback setting has an option of creating a baseline air quality grid data set and we used this setting choosing a state grid type for ozone pollution using the data set "EPA's standard monitors for O_3" in monitor year 2007, which is the most recent year available for this model version. Again, we used the Voronoi Neighborhood Averaging method in order to interpolate levels in between monitors.

b. **Incidence estimation.** In order to first estimate the incidences of particular health endpoints one needs to first create a new configuration. Configuration is defined as a record of choices you make in estimating the change in adverse health effects between a baseline and control scenario. The EPA Ozone Wizard (or default configuration) was used in order to load all CR functions and valuation studies that the EPA has used. Unless otherwise noted, epidemiological studies represent ages 0 to 99. These choices include the following:
    i. **The air quality grids for the baseline and control scenarios.** The air quality grids referenced above were used.
    ii. **The population data set.** U.S. census projections for population year 2020 and 2050 were used for the relevant runs. BenMAP includes census data from the year 2000.[5] Population projections are built into the BenMAP Model (EPA 2010b).
    iii. **The threshold.** For the analysis the threshold[6] chosen was zero, because air quality metrics below the threshold should be given a real value instead of being replaced with a static threshold value (Bell, Peng, and Domenici 2006).

iv. **Latin hypercube points.** Latin hypercube points were used instead of point mode because we wanted a range of results that mirror the variability in the inputs to the health impact functions. We choose 100 Latin hypercube points in order to obtain results in increments of 5 percentiles. This setting allows you to generate specific percentiles along the estimated incidence distribution.

v. **Health impact functions.** We used the EPA's Standard Health Functions and selected the following health functions per EPA methodology recommendations found in EPA 2009. When more than one study was used, a sum-independent summing methodology was implemented.

1. Mortality (non-accidental): Both Bell et al. 2004 and Schwartz 2005 were used to evaluate mortality based on eight-hour maximum warm season ozone levels. Values were pooled using a sum-independent pooling method. This was based on a 2020 and 2050 mortality incidence prediction.
2. Senior and infant hospital admissions (all respiratory): Both Burnett et al. 2001 and Schwartz 1995 were used to evaluate the associated hospital admissions from the daily eight-hour maximum warm season for ozone. Burnett et al. 2001 is a study focused on infants ages 0 to 1 and Schwartz 1995 is a study focused on seniors ages 65 and older.
3. Emergency room visits (asthma-related): Both Peel et al. 2005 and Wilson et al. 2005 were used to evaluate associated emergency room visits.
4. Acute respiratory symptoms: Ostro and Rothschild 1989 was used to evaluate acute respiratory symptoms. This study was based on eight-hour maximum ozone levels.
5. School loss days: Both Chen et al. 2000 and Gilliland et al. 2001 were used to evaluate school loss days. Both studies were based on eight-hour maximum ozone levels throughout the year.

c. **Aggregation, pooling, and valuation (APV).** In order to combine the incidence results and place an economic value on the combined results we created an APV configuration file in order to aggregate incidence at the state level, pool the incidences, and then assign economic values to those incidences.

i. Pooling incidence results. When more than one epidemiological study was used, results were pooled to report one single set of results. The studies used for each impact listed above were pooled using a random/fixed effects method. In this pooling method BenMAP first tests if random weights should be used. If not, BenMAP uses fixed effects weights. If yes, the weights take into account both the

variance within each set of results and the variance between sets of results. BenMAP then reports all data in Latin hypercube mode so that one can report the fifth and ninety-fifth percentiles of data.

ii. **Valuation functions:** For each health impact function a particular valuation function was established and can be found below (EPA 2010a).[7]

1. Mortality, (non-accidental): Results were pooled using two estimates for the Value of a Statistical Life (VSL) study based on 26 value-of- life studies for ages 0 to 99.
2. Seniors and infants hospital admissions, (all respiratory): Two cost- of-illness studies were used showing medical expenses and, for seniors, lost wages due to hospital admission for ages 0 to 1 in the population for the (infants) and ages 65 and older (for the seniors).
3. Emergency room visits (asthma-related): Stanford study was used from 1999 for ages 0 to 99.
4. Acute respiratory symptoms: WTP study was used to estimate three symptoms, one day illness using contingent valuation for ages 18 to 65.
5. School loss days: A standard EPA estimate based on economic studies for ages 0 to 17. This study uses a unit value based on the probability that if a child is sick the parent will have to stay home from work and on the lost wages of a parent that has to stay home with the child.

iii. **Pooling Valuation Results:** The value of hospital admissions for both seniors and infants was calculated separately and then combined using a sum- (independent) method of pooling where in which results are summed assuming they are independent and then a Monte Carlo analysis simulation is used, which reports results in hypercube mode in order to present the fifth and ninety-fifth percentiles of data. There was only one valuation study or one value reported from a meta-analysis used for the rest of the health impacts.

V. Full Results of Modeling

It should be noted that most of the changes in incidence and associated costs scale directly with the amount of ozone increase. For instance, a doubling in ozone from 1 ppb to 2 ppb results in roughly a doubling in total health costs. This arises because most CR functions commonly used have a log-linear relationship between change in pollutant level and change in incidence of a particular health effect (Abt Associates 1999). Changes in incidents and associated costs can then be readily determined for various increases in ozone. The following tables show the total incidences of acute respiratory symptoms by state in 2020, and their costs, for the 2 ppb

climate penalty on ozone. We modeled the US-40, and not all 50 states, for two reasons. First, the BenMAP model does not include Alaska and Hawaii. Second, for the eight states of Florida, Georgia, South Carolina, Alabama, Mississippi, Louisiana, Oregon, and Washington, the climate penalty is either absent, inconclusive, or may in fact be a benefit rather than a penalty.

Table 2: Occurrences of Acute Respiratory Symptoms Associated with a 2 ppb Climate Penalty in 2020

State	Low	Central	High
ARIZONA	39,202	77,065	114,831
ARKANSAS	16,166	31,781	47,355
CALIFORNIA	225,208	442,724	659,684
COLORADO	31,071	61,081	91,014
CONNECTICUT	19,640	38,609	57,530
DELAWARE	5,033	9,895	14,743
DISTRICT OF COLUMBIA	2,967	5,833	8,692
IDAHO	9,003	17,698	26,370
ILLINOIS	73,111	143,724	214,157
INDIANA	35,421	69,632	103,756
IOWA	16,174	31,796	47,378
KANSAS	15,605	30,676	45,709
KENTUCKY	24,306	47,781	71,197
MAINE	7,418	14,583	21,730
MARYLAND	35,046	68,894	102,656
MASSACHUSETTS	36,380	71,518	106,566
MICHIGAN	56,472	111,016	165,420
MINNESOTA	31,292	61,516	91,662
MISSOURI	33,687	66,224	98,677
MONTANA	5,549	10,909	16,256
NEBRASKA	9,938	19,536	29,110
NEVADA	17,290	33,989	50,646
NEW HAMPSHIRE	8,233	16,184	24,115
NEW JERSEY	51,032	100,320	149,483
NEW MEXICO	12,307	24,194	36,051
NEW YORK	108,148	212,601	316,788
NORTH CAROLINA	52,354	102,920	153,357
NORTH DAKOTA	3,565	7,008	10,442
OHIO	62,526	122,916	183,152
OKLAHOMA	20,246	39,801	59,305
PENNSYLVANIA	67,660	133,009	198,191

RHODE ISLAND	6,236	12,259	18,267
SOUTH DAKOTA	4,226	8,307	12,378
TENNESSEE	36,006	70,782	105,470
TEXAS	147,137	289,249	430,998
UTAH	15,995	31,443	46,852
VERMONT	3,658	7,191	10,714
VIRGINIA	47,253	92,893	138,416
WEST VIRGINIA	9,635	18,941	28,223
WISCONSIN	32,370	63,634	94,819
WYOMING	2,911	5,722	8,526

Table 3: Total Costs for Health Impacts Associated with a 2 ppb Climate Penalty in 2020

State	Low	Central	High
ARIZONA	$24,541,372	$148,995,076	$376,437,184
ARKANSAS	$11,717,838	$74,094,358	$188,578,080
CALIFORNIA	$122,327,848	$729,189,387	$1,833,793,408
COLORADO	$15,265,466	$86,843,601	$216,459,312
CONNECTICUT	$12,840,146	$79,483,661	$201,898,656
DELAWARE	$3,424,737	$21,186,847	$53,758,652
DISTRICT OF COLUMBIA	$1,889,881	$11,623,383	$29,406,254
IDAHO	$5,688,906	$33,793,570	$85,273,352
ILLINOIS	$44,397,884	$272,348,971	$688,928,896
INDIANA	$22,758,446	$141,018,228	$357,285,824
IOWA	$10,665,522	$66,484,775	$168,976,512
KANSAS	$9,769,944	$59,627,388	$150,986,464
KENTUCKY	$16,369,968	$102,270,928	$259,529,184
MAINE	$5,514,758	$35,650,636	$91,232,656
MARYLAND	$21,734,078	$133,398,027	$337,702,272
MASSACHUSETTS	$22,872,328	$141,137,311	$358,262,624
MICHIGAN	$37,111,388	$230,322,576	$584,559,104
MINNESOTA	$17,785,162	$106,897,755	$269,533,280
MISSOURI	$21,930,558	$136,052,194	$345,086,144
MONTANA	$4,165,776	$26,306,163	$67,127,880
NEBRASKA	$5,937,540	$35,842,552	$90,504,992
NEVADA	$11,448,860	$71,475,286	$181,536,496
NEW HAMPSHIRE	$5,309,230	$33,345,843	$84,840,416
NEW JERSEY	$32,958,788	$203,089,678	$515,592,448
NEW MEXICO	$8,097,121	$49,572,902	$125,504,816

NEW YORK	$64,435,576	$391,568,948	$989,410,432
NORTH CAROLINA	$33,827,116	$208,603,062	$528,660,192
NORTH DAKOTA	$2,470,045	$15,715,004	$40,077,660
OHIO	$43,131,708	$270,632,844	$688,944,832
OKLAHOMA	$13,737,882	$86,170,129	$219,039,216
PENNSYLVANIA	$51,854,224	$331,680,221	$849,044,416
RHODE ISLAND	$4,167,053	$26,110,417	$66,527,132
SOUTH DAKOTA	$2,845,233	$17,854,139	$45,373,184
TENNESSEE	$24,297,476	$152,182,818	$386,545,056
TEXAS	$79,533,664	$466,321,836	$1,168,692,992
UTAH	$7,800,370	$43,127,217	$107,172,240
VERMONT	$2,483,804	$15,789,236	$40,246,168
VIRGINIA	$29,436,950	$177,950,325	$449,390,848
WEST VIRGINIA	$7,703,738	$49,877,329	$127,637,192
WISCONSIN	$20,545,678	$127,007,301	$322,411,904
WYOMING	$2,011,663	$12,635,457	$32,126,238

REFERENCES

Abt Associates, Inc. 1999. *Out of breath: Health effects from ozone in the eastern United States*. Prepared for the Clean Air Task Force. Washington, DC: Clear the Air: National Campaign Against Dirty Power.

Archer D. 2007. Methane hydrate stability and anthropogenic climate change. *Biogeosciences* 4:521–544.

Avise, J., J. Chen, B. Lamb, C. Wiedinmyer, A. Guenther, E. Salathe, and C. Mass. 2009. Attribution of projected changes in summertime US ozone and $PM_{2.5}$ concentrations to global changes. *Atmospheric Chemistry and Physics* 9:1111–1124.

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. *Journal of the American Medical Association* 292(19):2372–2378.

Bell M.L., R.D. Peng, and F. Dominici. 2006. The exposure-response curve for ozone and risk of mortality and the adequacy of current ozone regulations. *Environmental Health Perspectives* 114(4):532–536.

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. *Geophysical Research Letters* 36(9):L09803.

Burnett, R.T., M. Smith-Doiron, D. Steib, M.E. Raizenne, J.R. Brook, R.E. Dales, J.A. Leech, S. Cakmak, and D. Krewski. 2001. Association between ozone and hospitalization for acute respiratory diseases in children less than 2 years of age. *American Journal of Epidemiology* 153(5):444–452.

Cameron, T.A. 2010. Euthanizing the value of a statistical life. *Review of Environmental Economics and Policy* 4(2):161–178.

Chen, L., B.L. Jennison, W. Yang, and S.T. Omaye. 2000. Elementary school absenteeism and air pollution. *Inhalation Toxicology* 12(11):997–1016.

Cofala, J., M. Amann, Z. Klimont, K. Kupiainen, and L. Hoglund-Isaksson. 2007. Scenarios of global anthropogenic emissions of air pollutants and methane until 2030. *Atmospheric Environment* 41(38):8486–8499.

Cooper, O.R., D.D. Parrish, A. Stohl, M. Trainer, P. Nedelec, V. Thouret, J.P. Cammas, S.J. Oltmans, B.J. Johnson, D. Tarasick, T. Leblanc, I.S. McDermid, D. Jaffe, R. Gao, J. Stith, T. Ryerson, K. Aikin, T. Campos, A. Weinheimer, and M.A. Avery. 2010. Increasing springtime ozone mixing ratios in the free troposphere over western North America. *Nature* 463(7279):344–348.

Coupled Model Intercomparison Project (CMIP). *CMIP - Coupled Model Intercomparison Project - Overview.* Online at *http://cmip-pcmdi.llnl.gov*, accessed May 13, 2011.

Environmental Protection Agency (EPA). 2011a. National trends in CO levels. Washington, DC. Online at *http://www.epa.gov/airtrends/carbon.html*, accessed on May 12, 2011.

Environmental Protection Agency (EPA). 2011b. Air quality trends and emission trends. Washington, DC. Online at *http://www.epa.gov/airtrends/aqtrends.html*, accessed on May 6, 2011.

Environmental Protection Agency (EPA). 2010a. Appendix J: Health Valuation Functions in U.S. Setup. In BenMAP: Environmental Benefits Mapping and Analysis Program, User's Manual Appendices. p. 365. Online at *http://www.epa.gov/air/benmap/models/BenMAPManualAppendicesAugust2010.pdf.*

Environmental Protection Agency (EPA). 2010b. Appendix K: Population & Other Data in U.S. Setup. In BenMAP: Environmental Benefits Mapping and Analysis Program, User's Manual Appendices. p. 380. Online at *http://www.epa.gov/air/benmap/models/BenMAPManualAppendicesAugust2010.pdf.*

Environmental Protection Agency (EPA). 2009. Summary of the Updated Regulatory Impact Analysis (RIA) for the Reconsideration of the 2008 Ozone National Ambient Air Quality Standard (NAAQS). Chapter S3-1. Online at *http://www.epa.gov/ttn/ecas/regdata/RIAs/s1-supplemental_analysis_full.pdf*.

Fann, N., and D. Risley. 2011. The public health context for PM2.5 and ozone air quality trends. *Air Quality, Atmosphere, and Health,* DOI:10.1007/s11869-010-0125-0.

Gilliland, F.D., K. Berhane, E.B. Rappaport, D.C. Thomas, E. Avol, W.J. Gauderman, S.J. London, H.G. Margolis, R. McConnell, K.T. Islam, and J.M. Peters. 2001. The effects of ambient air pollution on school absenteeism due to respiratory illnesses. *Epidemiology* 12(1):43–54.

Guenther, A., T. Karl, P. Harley, C. Wiedinmyer, P.I. Palmer, and C. Geron. 2006. Estimates of global terrestrial isoprene emissions using MEGAN (Model of Emissions of Gases and Aerosols from Nature). *Atmospheric Chemistry and Physics* 6:3181–3210.

Hogrefe, C., B. Lynn, K. Civerolo, J.-Y. Ku, J. Rosenthal, C. Rosenzweig, R. Goldberg, S. Gaffin, K. Knowlton, and P.L. Kinney. 2004. Simulating changes in regional air pollution over the eastern United States due to changes in global and regional climate and emissions. *Journal of Geophysical Research* 109:D22301.

Intergovernmental Panel on Climate Change (IPCC). 2000. *Emissions scenarios*, edited by N. Nakicenovic and R. Swart. Cambridge, UK: Cambridge University Press. Online at *http://www.ipcc.ch/ipccreports/sres/emission/index.php?idp=0*, accessed on May 11, 2011.

Jacob, D.J., and D.A. Winner. 2009. Effect of climate change on air quality. *Atmospheric Environment* 43(1):51–63.

Liao, H., W.-T. Chen, and J.H. Seinfeld. 2006. Role of climate change in global predictions of future tropospheric ozone and aerosols. *Journal of Geophysical Research* 111:D12304.

Murazaki, K., and P. Hess. 2006. How does climate change contribute to surface ozone change over the United States? *Journal of Geophysical Research* 111:D05301.

National Oceanic and Atmospheric Administration (NOAA). 2010. *The NOAA Annual Greenhouse Gas Index (AGGI).* Boulder, CO. Online at *http://www.esrl.noaa.gov/gmd/aggi*, accessed on May 13, 2011.

Nolte, C.G., A.B. Gilliland, C. Hogrefe, and L.J. Mickley. 2008. Linking global to regional models to assess future climate impacts on surface ozone levels in the United States. *Journal of Geophysical Research* 113:D14307.

Ostro, B.D., and S. Rothschild. 1989. Air Pollution and Acute Respiratory Morbidity: an Observational Study of Multiple Pollutants. *Environmental Research* 50(2):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(2):164–174.

Prather, M., M. Gauss, T. Berntsen, I. Isaksen, J. Sundet, I. Bey, G. Brasseur, F. Dentener, R. Derwent, D. Stevenson, L. Grenfell, D. Hauglustaine, L. Horowitz, D. Jacob, L. Mickley, M. Lawrence, R. von Kuhlmann, J.-F. Muller, G. Pitari, H. Rogers, M. Johnson, J. Pyle, K. Law, M. van Weele, and O. Wild. 2003. Fresh air in the 21st century? *Geophysical Research Letters* 30(2):1100.

The Royal Society. 2008. *Ground-level ozone in the 21st century: future trends, impacts and policy implications*. London.

Schwartz, J. 1995. Short term fluctuations in air pollution and hospital admissions of the elderly for respiratory disease. *Thorax* 50(5):531–538.

Schwartz, J. 2005. How sensitive is the association between ozone and daily deaths to control for temperature? *American Journal of Respiratory Critical Care Medicine* 171(6):627–631.

Sillman, S., J. Logan, and S. Wofsy. 1990. The Sensitivity of Ozone to Nitrogen Oxides and Hydrocarbons in Regional Ozone Episodes. *Journal of Geophysical Research* 95(D2):1837–1851.

Steiner, A.L., S. Tonse, R.C. Cohen, A.H. Goldstein, and R.A. Harley. 2006. Influence of future climate and emissions on regional air quality in California. *Journal of Geophysical Research* 111:D18303.

Tagaris, E., K.-J. Liao, A.J. DeLucia, L. Deck, P. Amar, and A.G. Russell. 2009. Potential Impact of Climate Change on Air Pollution-Related Human Health Effects. *Environmental Science and Technology* 43(13): 4979–4988.

Taha, H. 2001. *Potential impacts of climate change on tropospheric ozone in California: A preliminary episodic modeling assessment of the Los Angeles basin and the Sacramento valley*. Berkeley, CA: Lawrence Berkeley National Laboratory.

U.S. Global Change Research Program (USGCRP). 2009. *Global climate change impacts in the United States*, edited by T.R. Karl, J.M. Melillo, and T.C. Peterson. New York: Cambridge University Press. Online at *http://www.globalchange.gov/publications/reports/scientific-assessments/us-impacts/download-the-report*, accessed on May 11, 2011.

Voorhees, A.S., N. Fann, C. Fulcher, P. Dolwick, B. Hubbell, B. Bierwagen, and P. Morefield. 2011. Climate change-related temperature impacts on warm season heat mortality: A proof-of-concept methodology using BenMAP. *Environmental Science and Technology* 45(4):1450–1457.

Wang, H., D.J. Jacob, P. LeSager, D.G. Streets, R.J. Park, A.B. Gilliland, and A. van Donkelaar. 2009. Surface ozone background in the United States: Canadian and Mexican pollution influences. *Atmospheric Environment* 43(6):1310–1319.

Warneke, C., S.A. McKeen, J.A. de Gouw, P.D. Goldan, W.C. Kuster, J.S. Holloway, E.J. Williams, B.M. Lerner, D.D. Parrish, M. Trainer, F.C. Fehsenfeld, S. Kato, E.L. Atlas, A. Baker, and D.R. Blake. 2007. Determination of urban volatile organic compound emission ratios and comparison with an emissions database. *Journal of Geophysical Research* 112:D10S47.

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. *Environmental Research* 97(3):312–321.

Wu, S., L.J. Mickley, E.M. Leibensperger, D.J. Jacob, D. Rind, and D.G. Streets. 2008. Effects of 2000–2050 global change on ozone air quality in the United States. *Journal of Geophysical Research* 113:D06302.

ENDNOTES

[1] For a detailed overview of the chemistry involved, see The Royal Society 2008.

[2] Emissions scenarios are described in IPCC 2000.

[3] For more information on BenMAP, Version 4.0, visit *http://www.epa.gov/air/benmap.*

[4] The basic assumption underlying the VSL approach is that equal increments in fatality risks are valued equally. For similar reasons, the VSL approach is only appropriate for marginal changes in the risk of death and should not be used to value more significant changes. Because changes in individual fatality risks resulting from environmental regulation are typically very small, the VSL approach is usually acceptable for these types of benefit analyses (EPA 2010a).

[5] U.S. Census 2000 data and projects can be found at *http://www.census.gov/main/www/cen2000.html.*

[6] Threshold is defined as an air quality limit below which benefits are not calculated.