Quantification of the Health Impacts Associated with Fine Particulate Matter due to Wildfires

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Abstract

Wildfires can be devastating to property and the ecological landscape; they also have a substantial impact on human health and welfare. Wildfires emit a variety of air pollutants such as fine particulate matter ($PM_{2.5}$), coarse particulate matter (PM_{10}), volatile organic compounds, as well as nitrogen and sulfur oxides. Fine particles ($PM_{2.5}$) have been linked to many cardiovascular and respiratory problems such as premature death, heart attacks, asthma exacerbation, and acute bronchitis. This project focuses on quantifying the incidence and monetary value of adverse human health impacts resulting from wildfire emissions of $PM_{2.5}$ in the Pacific Northwest during the summer of 2007. Using a combination of tools, including geospatial analysis and a benefits assessment tool developed by U.S. EPA (BenMAP), this project investigates the changes in incidence of certain health outcomes resulting from the change in air quality attributable to wildfire. The changes in incidence can then be given a dollar value using valuation functions to highlight the magnitude of the health effects caused by $PM_{2.5}$ wildfire emissions. In light of current climate change predictions, $PM_{2.5}$ wildfire emissions may be expected to increase in the future.

Table of Contents

Introduction

Fire has played an integral role in the health and vitality of ecosystems for centuries. Fire affects important ecological factors such as nutrient loss, genetic adaptation of plants, biomass accumulation and wildlife population dynamics (Barnes et al., 1998). Fire has also been important to humans as a tool for clearing land for crops, to ease the burden of travel, and for general land management (Daniel et al., 2007). Wildfire is defined as an unplanned, unwanted wildland fire (National Wildfire Coordinating Group, 2007). While wildlands are places of little or no development, in the past two centuries, wildland area has continued to shrink while humans proliferate and expand into previously unsettled, undeveloped areas, creating a wildland-urban interface (National Wildfire Coordinating Group, 2007). Many of such areas are at substantial risk for wildfire. When fire does occur in such locations, it may be costly and dangerous to both lives and property. Therefore, it has been the policy of the United States Forest Service to suppress any and all fire for the better portion of its tenure (Forest History Society, 2007). This complete suppression of fire led to fuel buildup which in turn increased the chances of catastrophic wildfire, which is a fire that brings physical or financial ruin (USDA Forest Service, 2002, 101; Ryan, 2000). In fact, there is a growing trend of larger, more significant wildfire events in the last decade (National Interagency Fire Center, 2007). Most recently, there have been significant wildfires in the Western and Southeastern United States (USDA Forest Service, 2002,

102). There was the Biscuit Fire of 2002 in Southwestern Oregon and Northern California which burned a total of 7.2 million acres with suppression costs of 153 million dollars (The Wilderness Society, 2002). There was also the Big Turnaround Complex fire in Georgia in 2007 which burned approximately 386,000 acres and cost over 26 million to suppress (Inciweb, 2007). Also in 2007, the Murphy Complex fire burned over 650,000 acres in Idaho (Inciweb, 2007).

Wildfires emit various air pollutants including fine particulate matter ($PM_{2.5}$), particles with aerodynamic diameters of less than 2.5 micrometers; coarse particulate matter (PM_{10}) of less than 10 micrometers in diameter: volatile organic compounds (VOCs); and nitrogen and sulfur oxides (U.S. EPA, 1998, 3). $PM_{2.5}$ will serve as the primary focus of this analysis due to its detrimental effects on human health. There has been extensive investigation into the effects of fine particulate matter on human health. Scientists have found both long-term and short-term effects associated with exposure to fine particulate matter pollution in both adults and children (U.S. EPA, 2005). $PM_{2.5}$ has been positively associated with health endpoints including total mortality, cardiovascular mortality, respiratory mortality, lung cancer, increased hospital admissions for respiratory and cardiovascular diseases, increased incidence of respiratory disease, decreased lung function, and onset of myocardial infarction (U.S. EPA, 2005).

Most research on wildfires has centered on the ecological impacts of fire (USDA, 2003, 181). Potential ecological impacts from wildfires include landslides, erosion, flooding, and water-quality impairment (Fried et al., 2004, 185). Wildfires also have the potential to substantially modify the existing species composition and distribution in an ecosystem (Malanson and Westman, 1991). Some economic valuation of ecological impacts has been conducted. Mills and Flowers (1986) examined the effects of wildfire on the net-present value of timber across various management objectives, while Dale (2006) details the avoided costs of treatment and suppression as well as ecological benefits associated with controlled wildland fire.

The impact of wildfire on humans has been limited primarily to calculations of property losses and fire suppression costs (Fried et al., 2004; Nitschke and Innes, 2008). There has been very little research conducted which examines the effect of wildfires on human health endpoints. One notable exception is the Butry et al. (2001) study of the Florida 1998 wildfire season in which the authors used actual cost of illness data to quantify the health impacts of 6 weeks of wildfire in Northeastern Florida. However, this study had data limitations as it included the confounding effects of several pollutants including particulates, carbon monoxide, volatile organic compounds and nitrogen oxides. Furthermore, the authors only examined respiratory health effects including emergency room visits, hospital admissions, and doctor visits for acute bronchitis and asthma exacerbation over a six-week period. Therefore, it did not

include the full suite of health endpoints that may be expected from wildfire pollution exposure (Butry et al., 2001, 15).

Another study by Frankenburg et al. (2005) uses longitudinal health survey data and subjective visibility measurements of regional haze as a proxy for coarse particulate matter (PM_{10}) to estimate health impacts from wildfires during the 1997 wildfire season in Indonesia. The authors examined impacts such as cough, difficulty in carrying a heavy load, sit-to-stand reaction time, and overall opinion of general health status. The study found a significant, negative effect on respiratory health among exposed respondents (Frankenburg et al., 2005).

Though these studies are valuable, we currently lack an adequate understanding of the human health costs associated with wildfire, either in specific locations or on a broad scale. The current literature lacks direct emissions data, a long-term view, and full accounting of health outcomes. Butry et al. lacks direct emissions data and only touches upon the very short-term health impacts, thereby underestimating the value of health outcomes. Furthermore, using a cost of illness approach to quantifying the value of health impacts due to wildfires does not capture the full potential disbenefits. Frankenburg et al. also lacks direct emissions data, objective measurements of respondent health, and fails to place a number on these health impacts. Thus, there are no complete health outcome characterizations with valuation based on both long- and short-term effects reflecting the peer-reviewed epidemiology literature.

The purpose of this project is to analyze quantitatively the human health impacts and associated health costs of wildfire emissions. Through a quantitative analysis focusing on a case study of fire-related $PM_{2.5}$ emissions in the U.S. Pacific Northwest, this project will provide important, specific information about human health costs associated with wildfire and the potential monetized health benefits that can be realized by preventing wildfire. By examining wildfires from this detailed perspective, the project will provide a greater understanding of the specific consequences of untamed wildfire. This is especially relevant in light of the Intergovernmental Panel on Climate Change's newly released fourth assessment report which predicts lower precipitation levels for much of the western United States (IPCC, 2007, 890). The fourth assessment report also predicts annual and summer mean temperature increases for much of the United States. The combination of reduced precipitation and temperature during the months of June, July and August can be expected to increase both the frequency and intensity of wildfire in the Pacific Northwest (IPCC, 2007, 890). Furthermore, a case study by the Pew Center on Global Climate Change predicts up to a 57% increase in area burned by fire over the coming century. Nitschke and Innes (2004) similarly modeled climate change scenarios which resulted in predictions of increases in area burned, fire season length and fire severity for British Columbia, Canada. By quantifying the monetary value of health impacts associated with wildfire,

policymakers could gain valuable knowledge affecting a multitude of policies from prescribed burning to climate change mitigation strategies.

Objective

This project is designed to assess the extent and the magnitude of human health impacts associated with increased levels of fine particulate matter due to wildfires. Using existing fire emissions data and information relating ambient $PM_{2.5}$ concentrations to a variety of morbidity and mortality endpoints, the project quantifies the human health costs associated with wildfire emissions of fine particles in the tri-state area of Washington, Oregon and Idaho, and interprets those results in light of emerging scientific predictions about wildfire rates under future climate change scenarios.

Data and Methods

In order to conduct a quantitative analysis of health impacts, it is necessary to select a specific location affected by wildfire where there is sufficient data to estimate both the air quality impacts of the wildfire and the resulting health impacts on the surrounding human population. This project therefore focuses on analyzing the impacts of wildfire in a specific case study location which has a) been affected by wildfire in the last five years; b) adequate emissions data to allow for quantitative estimates of changes in air quality before and after wildfire; and c) a potentially exposed population of greater than 100,000 people in order to ensure a sufficient exposure.

There is a significant prevalence of wildfire in the western United States. These states have historically been subject to a multitude of fires ranging in frequency and intensity (Agee, 1989, 11). Fire is a natural part of these climate and ecosystem regimes. In 2006, there was the Tripod Complex fire in Washington State and the South End Complex fire in Oregon which each burned over 100,000 acres (NIFC, 2007). In 2007, there were the Murphy Complex, Cascade Complex, and East Zone Complex fires in Idaho which burned over 1.2 million acres (NIFC, 2007).

I received emissions data from Dr. Shawn Urbanski, a research chemist who works for the Fire Science Laboratory in Missoula, Montana for the United States Forest Service. The air emissions, in kilograms per square kilometer per hour, geographically covered the entire Northwestern United States for the months of June through September of 2007. Most importantly, the emissions were the product of only wildfires and therefore did not include other sources of $PM_{2.5}$. Because the spatial distribution of the data only covered portions of California, Montana, Wyoming, Utah and Nevada, I selected the tri-state area of Oregon, Washington, and Idaho as my case study location whose boundaries were completely enclosed within the spatial distribution of the data.

These three states have adequate population numbers in order to ensure a sufficient exposure. To clarify, traditional risk management principles identify risk as a product of hazard and exposure. The wildfire emissions data clearly identified the hazard; to be able to conduct a risk analysis case study to demonstrate the resulting risk

of wildfire emissions, I also needed to choose a location where exposure to $PM_{2.5}$ pollution is likely—specifically, an area where people actually live. By choosing such an area and then analyzing the overlap between population centers and wildfire it was possible to reasonably assume an actual exposure to the $PM_{2.5}$ pollution. Idaho is the thirteenth largest state in the United States with cities such as Boise and Idaho Falls. Oregon includes such cities as Salem, Eugene and Portland, while Washington is home to the thriving metropolis of Seattle. Together, these three states have a combined 2007 population estimate of 11,715,281 (U.S. Census Bureau, 2007).

After obtaining the emissions data, I had to perform a complex, multi-step analysis in order to estimate the change in human health outcomes and their associated values. First, I needed to transform the collected $PM_{2.5}$ emissions data into air quality concentrations of the pollutant. The spatially-oriented data went from units of $kg/km^2/hr$ to units of $\mu g/36km^2/year$. The end result was ambient air quality concentrations of $PM_{2.5}$ attributable to wildfire emissions over time and space.

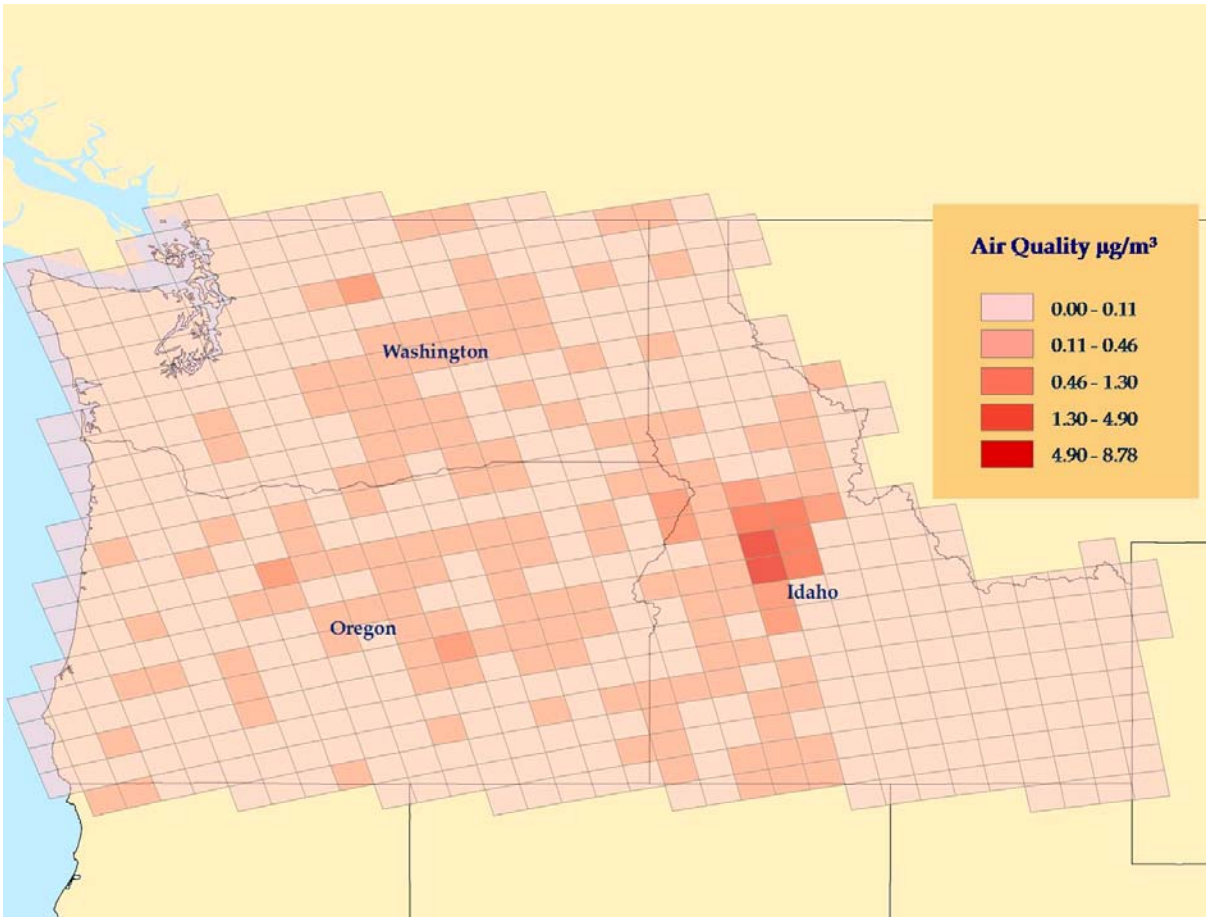

Figure 1. Air Quality Impacts Map in units of µg/m^3

The lightest pink areas represent an air quality impact from 2007 wildfire emissions of $PM_{2.5}$ ranging from 0.00 to 0.11µg/m^3 while the next pink represents areas with a 0.11 to 0.46 µg/m^3. The medium pink areas are areas with an impact ranging from 0.46 to 1.30 µg/m^3. The lighter red areas represent impacts of 1.30 to 4.90 µg/m^3 while the darkest red areas have impacts of 4.90 to 8.78 µg/m^3.

As the map in Figure 1 indicates, the air quality impacts were largest in Idaho, while Washington and Oregon saw only moderate air quality impacts from wildfires. Overall, these air quality impacts are quite modest when compared to other sources of

$PM_{2.5}$ emissions, such as utilities and other major industrial sources, which can impact air quality quite significantly. While forest fire emissions were only measured over the wildfire season which included the months of June, July, August, and September, the emissions were annualized to reflect the annual contribution of $PM_{2.5}$ emissions in order to fit the specifications of the benefits model used to calculate and value the health outcomes. Thus, the seasonal impacts were much higher (in terms of shorter-term ambient concentrations of $PM_{2.5}$ in $\mu g/m^3$) than the low annual contributions shown by the map.

Traditionally, the calculation of air quality impacts associated with an emissions event of this magnitude would be done using a multi-scale air quality dispersion model; however, this type of modeling was infeasible given the time and resources available to conduct this analysis. Therefore, to translate emissions into air quality, I utilized an impact ratio instead. An impact ratio takes tons of emissions of a particular pollutant and translates these emissions into air quality in micrograms per cubic meter ($\mu g/m^3$) for a given geographical location. This air quality data was ready for input into BenMAP, a benefits mapping and analysis tool. For further detail about the steps of this data transformation, please see Appendix A.

Next, I sought to estimate the changes in human health outcomes using BenMAP. BenMAP is a peer-reviewed benefits mapping and analysis tool developed for use in the regulatory impact analyses conducted by the U.S. EPA (U.S. EPA, 2007). BenMAP

has been used to conduct benefits analyses for several major national and regional air quality regulations, including the Clean Air Interstate Rule in 2005 and the Particulate Matter National Ambient Air Quality Standard in 2006.

BenMAP is designed to estimate change in a variety of health outcomes associated with changes in ambient levels of certain air pollutants, in this case, $PM_{2.5}$. To estimate changes in health outcomes, BenMAP relates changes in air quality to changes in human health outcomes using health impact functions derived from a variety of concentration-response (C-R) relationships detailed in the peer-reviewed literature. The tool then places a dollar value on the change in incidence of these health outcomes by multiplying the estimated change in incidence by a program-supplied valuation amount. These valuation amounts come from the peer-reviewed literature and include both cost-of-illness values and willingness-to-pay values. Cost-of-illness values are conservative estimates in that they only account for the medical expenses necessary for care and ignore payments for pain and suffering (U.S. EPA, 2007). Willingness-to-pay values include measures of lost wages and avoided pain and suffering; these values provide a more complete value of avoiding a health outcome (U.S. EPA, 2007).

To begin, the user must specify air quality data; in this case, these were taken from the air quality file created through data transformation as well as a baseline file, detailed in Appendix A, which reflects background presence of $PM_{2.5}$. Then the user

selects the relevant population of interest, in this case: Oregon, Idaho, and Washington. These steps are used to simulate exposure of the population to the pollutant.

Next, the user selects the health endpoints of interest; BenMAP contains a variety of health outcomes to choose from. I included the following health endpoints in this analysis:

- adult and infant premature mortality
- non-fatal heart attacks
- respiratory hospital admissions
- respiratory emergency room visits
- cardiovascular hospital admissions
- chronic bronchitis
- acute bronchitis
- acute respiratory symptoms
- upper respiratory symptoms
- asthma exacerbation
- lower respiratory symptoms
- lost work days

These endpoints reflect the suite of endpoints included in the Particulate Matter National Ambient Air Quality Standard Regulatory Impact Analysis conducted in 2006. Replicating the U.S. EPA's choice of endpoints allowed for easy comparison across studies. All of the above endpoints except adult premature mortality and infant premature mortality are considered morbidity health endpoints, indicating they do not result in death.

Of the included health endpoints, the mortality endpoints and underlying concentration-response functions are the most controversial. While scientists agree that fine particulate matter is significantly associated with premature death, there is

disagreement and uncertainty over the exact incidence rate of premature death for a given change in air quality. Therefore, I have included estimates calculated from the concentration-response functions presented by several authors. First, I included the American Cancer Society study conducted by Pope et al. (2004). This study which began following 1.2 million American adults in 1982 has found consistent relationships over time between fine particles and premature mortality across the U.S. This study has been used historically by U.S. EPA as a basis for mortality incidence calculation (Pope et al., 2004). Next, is the Harvard Six-City Study conducted by Laden et al. (2006). This study found total mortality to be positively associated with ambient $PM_{2.5}$ concentrations (Laden et al., 2006). This study is now used as a central estimate by U.S. EPA. Then, the U.S. Environmental Protection Agency conducted an expert elicitation panel following the publication of the Clean Air Interstate Rule in 2005. The intent of the panel was to better characterize the uncertainty surrounding the incidence rate of premature death for a given change in ambient $PM_{2.5}$ levels. There were 12 experts and they were given labels to protect their anonymity. Expert E represents the upper bound with the greatest change in premature mortality incidence for a given change in $PM_{2.5}$ air quality, while Expert K represents the lower bound of estimates.

The change in incidence of health outcomes was then computed using the health-impact functions supplied by BenMAP for the geographic area of Washington, Oregon and Idaho. Thus, changes in incidence of mortality estimates represent premature

deaths attributable to air quality changes from wildfires during the 2007 season in Washington, Oregon and Idaho, while changes in incidence of morbidity estimates represent new cases attributable to air quality changes resulting from wildfires during the 2007 season in these three states.

Then, these changes in incidence are valued using dollar amounts derived from the peer-reviewed literature. All values in this analysis were given in terms of 2006 dollars. In this analysis, one case of adult premature mortality was valued at $6.6 million dollars, while one case of chronic bronchitis was valued at $410,000. Non-fatal heart attacks were valued at an amount between $80,000 and $168,000. Hospital admissions for cardiovascular and respiratory symptoms were valued at amounts between $10,000 and $25,000. Emergency room visits had a dollar value of $384, while respiratory symptoms were attributed an amount between $19 and $30. Asthma exacerbations were valued at $50, and lost work days at $130. Finally, one case of acute bronchitis was valued at $429. The total valuation results will reflect the incidence of the health outcomes multiplied by the outcome values. Thus, if the incidence of health outcomes is relatively small, total valuation results can still be substantial due to the magnitude of the adult premature mortality value which drives the total valuation results. Further technical details of the BenMAP analysis can be found in Appendix B – Steps in BenMAP.

I discounted certain portions of the valuation results at a 3% and 7% discount rate which reflect recommendations for benefits analyses found in EPA guidance documents. The 3% and 7% rates are those recommended in EPA's *Guidelines for Preparing Economic Analyses* (U.S. EPA, 2000b) and OMB Circular A-4 (OMB, 2003). The 3% reflects the social discount rate for public and governmental expenditures; the 7% discount rate reflects an approximation to the private discount rate and is slightly less than the average return on private investments in the United States stock market. Mortality and non-fatal heart attacks were discounted to the year 2020 to reflect the lag period between the exposure and the outcome, while the other morbidity endpoints remain undiscounted because these outcomes are assumed to commence immediately following exposure. I aggregated the incidence and valuation results by state and across all three states. Once the discounting and aggregation was complete, I constructed the tables presented in the results section below.

Results

I constructed the following tables: Selected Total Mortality Incidence Results, Mortality Incidence Results by State, Total Morbidity Incidence Results, Morbidity Incidence Results by State, Mortality Valuation at a 3% discount rate, Mortality Valuation at a 7% discount rate, Morbidity Valuation Results, Total Valuation at a 3% discount rate and Total Valuation at a 7% discount rate. Thus, each of the mortality incidence and

valuation tables includes a full range of values from the peer-reviewed literature as well as the expert elicitation panel.

Table 1. Selected Total Mortality Incidence Results

ACS Study (Pope et al., 2004)	1.63
Harvard Six-City Study (Laden et al., 2006)	3.69
Expert E (upper bound)	4.93
Expert K (lower bound)	0.48

Table 2. Mortality Incidence Results by State, 2007

	Washington	Oregon	Idaho
Mortality Impact Functions Derived from Epidemiology Literature			
ACS Study (Pope et al., 2004)	0.563 (0.2199 - 0.9061)	0.7746 (0.3025 - 1.2468)	0.2917 (0.1137 - 0.4708)
Harvard Six-City Study (Laden et al., 2006)	1.274 (0.6922 - 1.856)	1.7532 (0.9532 - 2.5549)	0.6639 (0.3591 - 0.9717)
Woodruff et al. 1997 (infant mortality)	0.0019 (0.0009 - 0.0028)	0.001 (0.0005 - 0.0015)	0.0006 (0.0003 - 0.0008)
Mortality Impact Functions Derived from Expert Elicitation			
Expert A	1.3648 (0.2493 - 2.4811)	1.8784 (0.343 - 3.4161)	0.7118 (0.129 - 1.3055)
Expert B	1.0531 (2.2251 - 0.1672)	1.4493 (3.0634 - 0.2301)	0.5479 (1.1684 - 0.864)
Expert C	1.03 (0.1859 - 2.2258)	1.4172 (0.2557 - 3.0644)	0.5357 (0.0961 - 1.1688)
Expert D	0.7175 (1.1824 - 0.1506)	0.9872 (1.6271 - 0.2073)	0.3723 (0.6157 - 0.0779)
Expert E	1.6998 (2.579 - 0.8554)	2.3396 (3.5511 - 1.177)	0.8889 (1.3582 - 0.4444)
Expert F	0.939 (0.6618 - 1.3584)	1.2921 (0.9106 - 1.8694)	0.4881 (0.3433 - 0.7084)
Expert G	0.6008 (1.1125 - 0)	0.8267 (1.5309 - 0)	0.3121 (0.579 - 0)
Expert H	0.7655 (0.0028 - 1.7632)	1.0533 (0.0039 - 2.4269)	0.3974 (0.0014 - 0.9224)
Expert I	1.0192 (1.8216 - 0.1623)	1.4025 (2.5074 - 0.2233)	0.5303 (0.9534 - 0.0839)
Expert J	0.8239 (0.2451 - 1.8311)	1.1336 (0.337 - 2.5206)	0.4279 (0.1268 - 0.9585)
Expert K	0.1652 (0.7668 - 0)	0.2273 (1.055 -0)	0.0855 (0.398 - 0)
Expert L	0.7893 (1.372 - 0.1671)	1.0861 (1.8882 - 0.2299)	0.4099 (0.7155 - 0.0863)

Table 1 represents the total mortality incidence results for the concentration-response functions of selected authors. The values are in units of cases of premature death in Washington, Oregon and Idaho due to the 2007 wildfire season. The authors Pope et al. and Laden et al. represent incidences derived from the author's adult concentration-response mortality function presented in the peer-reviewed literature. The Experts E and K represent incidence calculated by concentration-response functions from EPA's expert elicitation panel. Pope et al. (2004) represents the historic estimate used by EPA in previous benefits assessments, while Laden et al. (2006) is now used as a central estimate when presenting a range of mortality incidence values. Expert E represents the upper bound with an estimate of 4.93 additional premature deaths, while Expert K represents the lower bound of the incidence change calculations with only 0.48 premature deaths.

In Table 2, the mortality incidence for each state is presented by author. Pope et al. (2004) and Laden et al. (2006) concentration-response functions are again represented. The value from Woodruff et al. (1997) corresponds to an infant mortality concentration-response function from that study. The estimates of Experts A-L come from the full expert elicitation panel conducted by EPA. For Washington, the low estimate is 0.1652 more incidences of premature deaths resulting from the air quality change, with a 90% confidence interval (90%CI) of 0.7668 to 0 additional deaths. The high estimate in Washington State is 1.6998 more incidences of premature death given the change in air

quality, with a 90%CI of 2.579 to 0.8554 additional deaths. For Oregon, the low estimate is a 0.2273 change in the incidence of premature death with a 90%CI of 1.055 to 0 additional premature deaths, while the high end estimate is a 2.3396 unit change in incidences of premature death with a 90%CI of 3.5511 to 1.177. Finally, in Idaho the low end estimate is represented by a calculation of 0.0855 more premature deaths with a 90%CI of 0.398 to 0 as a result of the air quality impacts. The high estimate, given by Expert E, is approximately 0.8889 additional premature deaths with a 90%CI of 1.3582 to 0.4444 due to the prediction of the underlying concentration-response function. Air quality impacts in Idaho were the most significant of the three states; however, it also had the lowest change in incidence of premature mortality which probably reflects the low population numbers surrounding the air quality impacts in Idaho. Contrastingly, Oregon had the highest change in total incidence of premature mortality which can be attributed to the large size of the population in proximity to the wildfire emissions.

Table 3. Total Morbidity Incidence Results

Morbidity Endpoint	Total number of 2007 Cases
Hospital Admissions, Respiratory	0.31
Emergency Room Visits- Respiratory	0.58
Hospital Admissions, Cardiovascular	0.67
Chronic bronchitis	1.10
Acute Bronchitis	3.00
Non-fatal Heart Attacks	3.20
Upper Respiratory Symptoms	24.00
Asthma Exacerbation	30.00
Lower Respiratory Symptoms	33.00
Lost Work Days	206.00
Acute Respiratory Symptoms	1200.00

Table 3 presents the total morbidity incidence results summed across the three states. There are 1200 more predicted cases of acute respiratory symptoms and 206 lost work days because the worker was sick or had to care for a sick family member. The concentration-response functions also estimate 3.2 non-fatal heart attacks and 1.1 cases of chronic bronchitis. Fortunately, there tend to be fewer cases of more serious health outcomes such as hospital admissions and non-fatal heart attacks while there were many more cases of less severe outcomes.

Table 4 shows the morbidity incidence, categorized by state and the underlying concentration-response function utilized by BenMAP (by study author). These morbidity incidence estimates represent the change in frequency of morbidity endpoints per state given the air quality change. Thus, the units of each endpoint shown are new cases for 2007 resulting from the air quality change attributable to wildfires. As a highlight, for the State of Washington, there are estimated to be 79 work loss days attributable to wildfires as well as 473 cases of acute respiratory symptoms and 13 cases of asthma exacerbation. In Oregon, there are an estimated 89 work days lost due to health impacts attributable to wildfire with 11 cases of asthma exacerbation, 9 cases of upper respiratory symptoms, and 539 cases of acute respiratory symptoms. In Idaho, there are only 38 lost work days attributable to health impacts from wildfires, presumably due to the lower population density near areas of fire occurrence.

Table 4. Morbidity Incidence Results by State, 2007

Number of additional cases			
	Washington	Oregon	Idaho
Chronic bronchitis	0.4009 (0.0738 - 0.7283)	0.4787 (0.088 - 0.87)	0.2139 (0.0391 - 0.3919)
Acute myocardial infarction	1.1101 (0.5992 - 1.6214)	1.4715 (0.7939 - 2.1506)	0.6631 (0.3532 - 0.982)
Emergency Room Visits- respiratory	0.2617 (0.1537 - 0.3698)	0.221 (0.1298 - 0.3124)	0.0923 (0.0541 - 0.1308)
Acute Respiratory Symptoms	473.4448 (399.866 - 547.028)	538.9634 (455.187 - 622.7508)	229.4769 (193.679 - 265.3281)
Lower Respiratory Symptoms	14.4394 (6.9374 - 21.9455)	12.659 (6.0801 - 19.2455)	5.4016 (2.5865 - 8.2387)
Upper Respiratory Symptoms	10.539 (3.316 - 17.7623)	9.2762 (2.9186 - 15.6344)	3.9099 (1.2299 - 6.5912)
Work Loss Days	79.3929 (69.1717 - 89.6145)	89.0878 (77.6172 - 100.5592)	37.958 (33.06 - 42.8595)
Acute Bronchitis	1.3225 (0.045 - 2.692)	1.1565 (0.0394 - 2.3563)	0.4969 (0.0167 - 1.0226)
Asthma Exacerbation	13.3344 (38.1548 - 1.4569)	11.7524 (33.6319 - 1.284)	4.744 (13.5907 - 0.5179)
Hospital Admissions, Respiratory	0.1166 (0.0895 - 0.1408)	0.1335 (0.1098 - 0.1536)	0.0581 (0.048 - 0.0664)
Hospital Admissions, Cardiovascular	0.232 (0.2516 - 0.2118)	0.3067 (0.3366 - 0.2759)	0.1325 (0.1449 - 0.12)

The next two tables present health costs associated with the increase in health impact incidence of premature death discounted at both 3% and 7% discount rates. Table 5 presents the mortality costs in 2005 dollars at a 3% discount rate for each of the three states. In Washington, using the concentration-response function presented in Pope et al. (2004), BenMAP generated an estimate of mortality costs of approximately $3.4 million, while using Laden et al. (2006), BenMAP calculated a higher estimate of $7.7 million. The costs of infant mortality generated using the underlying concentration-response function by Woodruff et al. (1997) are approximately $11,000. Experts A-L which represent the expert elicitation panel conducted by U.S. EPA for adult mortality, contain a high estimate of $10.3 million dollars (Expert E) and a low estimate of $1 million dollars (Expert K), reflecting the value of $6.6 million dollars (2006 dollars) of one more premature death multiplied by the change in incidence predicted by the author's concentration-response function.

In Oregon, the value of the change in incidence calculated by Pope et al. (2004) is approximately $4.7 million dollars; correspondingly, the estimate generated by the C-R relationship presented by Laden et al. (2006) is a mortality cost of $10.6 million dollars. For infant mortality, the cost of the incidence estimate generated by Woodruff et al. (1997) is approximately $6000. The value of the change of incidence estimates in cases of premature mortality from the expert elicitation concentration-response functions range from $1.4 million dollars (Expert K) to $14.2 million dollars (Expert E). In Idaho,

the value of the change in incidence calculated by Pope et al. (2004) is approximately $1.8 million dollars, while the change in premature death incidence from Laden et al. (2006) corresponds to a cost of $4 million dollars.

Table 6 presents the mortality costs in 2005 dollars at a 7% discount rate for each of the three states. In Washington, the change in premature death incidence from the concentration-response function presented by Pope et al. (2004) is valued at a cost of approximately $3.1 million dollars, while the change in incidence estimated by the function from Laden et al. (2006), is valued at approximately $7 million dollars. The change in incidence of infant mortality is valued at approximately $10,400. The values of premature mortality incidence estimation by Experts A-L, contain a high estimate of $9.4 million dollars (Expert E) and a low estimate of $900,000 (Expert K).

In Oregon, the 0.7746 additional deaths resulting from the change in air quality calculated by Pope et al. (2004) correspond to a cost of $4.3 million dollars; meanwhile, the change in premature death calculated by the function presented by Laden et al. (2006) is valued at $9.7 million dollars. For infant mortality, the value of the change in incidence of premature death estimated by Woodruff et al. (1997) is approximately $5700. From the expert elicitation panel the value of the incidence estimates range from $1.2 million dollars (Expert K) to almost $13 million dollars (Expert E).

In Idaho, the value of the change estimated by the underlying concentration-response function by Pope et al. (2004) is approximately $1.6 million dollars, while the

value of the change in incidence estimate by Laden et al. (2006) is $3.7 million dollars. Woodruff et al. (1997) calculates a change of 0.0006 cases of infant mortality which are valued at a health cost of around $3000. In the expert elicitation panel values, the low estimate is given as approximately $470,000 (Expert K) for the author's estimated change in occurrence of premature mortality, while the high estimate is $4.9 million dollars (Expert E). In both Table 5 and Table 6, the magnitude of the premature death values is driven by the premature death dollar value of $6.6 million dollars (2006 dollars).

Table 5. 2007 Mortality Valuation Results by State in 2005 dollars at a 3% Discount Rate

3% Disc rate in 2005 dollars	Washington	Oregon	Idaho
Mortality Impact Functions Derived from Epidemiology Literature			
ACS Study (Pope et al.)	$3,416,824 ($759,096 - $6,827,415)	$4,700,888 ($1,044,267 - $9,393,875)	$1,770,868 ($392,705 - $3,543,306)
Harvard Six-City Study (Laden et al.)	$7,732,245 ($2,010,327 - $14,527,102)	$10,641,110 ($2,766,211 - $19,996,008)	$4,029,438 ($1,044,729 - $7,590,719)
Woodruff et al. 1997 (infant mortality)	$11,402 ($2,809 - $21,896)	$6,263 ($1,543 - $12,026)	$3,341 ($823 - 6,417)
Mortality Impact Functions Derived from Expert Elicitation			
Expert A	$8,283,662 ($1,106,826 - $18,091,675)	$11,400,387 ($1,522,587 - $24,907,808)	$4,319,868 ($572,295 - $9,503,844)
Expert B	$6,391,710 ($714,129 - $17,024,957)	$8,795,562 ($982,374- $23,437,733)	$3,325,692 ($369,201 - $8,932,171)
Expert C	$6,251,180 ($825,141 - $15,796,206)	$8,602,026 ($1,135,063 - $21,745,240)	$3,251,449 ($426,445 - $8,280,627)
Expert D	$4,354,623 ($668,524 - $9,136,159)	$5,991,579 ($919,662 - $12,572,361)	$2,260,210 ($345,800 - $4,755,070)
Expert E	$10,316,613 ($2,605,946 - $19,923,293)	$14,200,249 ($3,585,929 - $27,429,215)	$5,395,054 ($1,355,298 - $10,464,028)
Expert F	$5,698,834 ($1,635,216 - $11,013,076)	$7,841,675 ($2,250,057 - $15,155,577)	$2,962,074 ($849,772 - $5,734,664)
Expert G	$3,646,330 ($0 - $8,709,894)	$5,017,238 ($0 - $11,984,907)	$1,894,072 ($0 - $4,526,949)
Expert H	$4,646,334 ($14,436 - $12,214,860)	$6,392,976 ($19,857 - $16,812,207)	$2,411,809 ($7,451 - $6,381,185)
Expert I	$6,186,145 ($757,387 - $13,528,996)	$8,512,679 ($1,041,918 - $18,621,666)	$3,218,691 ($391,834 - $7,073,088)
Expert J	$5,000,629 ($960,769 - $12,893,854)	$6,880,618 ($1,321,714 - $17,747,167)	$2,596,875 ($497,112 - $6,737,728)
Expert K	$1,002,633 ($0 - $4,685,176)	$1,379,398 ($0 - $6,446,035)	$519,426 ($0 - $2,429,228)
Expert L	$4,790,720 ($709,449 - $10,850,634)	$6,591,723 ($976,002 - $14,932,929)	$2,487,360 ($367,261 - $5,656,632)

Table 6. 2007 Mortality Valuation Results in 2005 dollars at a 7% Discount Rate

7% Disc rate in 2005 dollars	Washington	Oregon	Idaho
Mortality Impact Functions Derived from Epidemiology Literature			
ACS Study (Pope et al.)	$3,116,444 ($692,362 - $6,227,203)	$4,287,623 ($952,463 - $8,568,040)	$1,615,187 ($358,181 - $3,231,807)
Harvard Six-City Study (Laden et al.)	$7,052,487 ($1,833,595 - $13,249,994)	$9,705,628 ($2,523,028 - $18,238,117)	$3,675,202 ($952,884 - $6,923,403)
Woodruff et al. 1997 (infant mortality)	$10,400 ($2,562 - $19,971)	$5,712 ($1,407 - $10,969)	$3,048 ($751 - $5,852)
Mortality Impact Functions Derived from Expert Elicitation			
Expert A	$7,555,428 ($1,009,523 - $16,501,199)	$10,398,155 ($1,388,734 - $2,2718,111)	$3,940,099 ($521,983 - $8,668,342)
Expert B	$5,829,802 ($651,349 - $15,528,258)	$8,022,325 ($896,011 - $21,377,273)	$3,033,324 ($336,744 - $8,146,925)
Expert C	$5,701,626 ($752,601 - $14,407,528)	$7,845,804 ($1,035,277 - $19,833,570)	$2,965,607 ($388,956 - $7,552,660)
Expert D	$3,971,799 ($609,752 - $8,332,980)	$5,464,847 ($838,813 - $11,467,098)	$2,061,510 ($315,400 - $4,337,042)
Expert E	$9,409,658 ($2,376,852 - $18,171,794)	$12,951,875 ($3,270,682 - $25,017,856)	$4,920,763 ($1,236,151 - $9,544,113)
Expert F	$5,197,837 ($1,491,461 - $10,044,893)	$7,152,297 ($2,052,250 - $13,823,219)	$2,701,672 ($775,066 - $5,230,517)
Expert G	$3,325,774 ($0 - $7,944,189)	$4,576,163 ($0 - $10,931,289)	$1,727,561 ($0 - $4,128,976)
Expert H	$4,237,866 ($13,167 - $11,141,026)	$5,830,956 ($18,112 - $15,334,211)	$2,199,782 ($6,796 - $5,820,202)
Expert I	$5,642,308 ($690,803 - $12,339,634)	$7,764,312 ($950,321 - $16,984,596)	$2,935,729 ($357,387 - $6,451,278)
Expert J	$4,561,014 ($876,306 - $11,760,328)	$6,275,728 ($1,205,519 - $16,186,977)	$2,368,578 ($453,410 - $6,145,401)
Expert K	$914,490 ($0 - $4,273,293)	$1,258,132 ($0 - $5,879,351)	$473,762 ($0 - $2,215,670)
Expert L	$4,369,558 ($647,079 - $9,896,732)	$6,012,230 ($890,200 - $13,620,144)	$2,268,691 ($334,974 - $5,159,346)

Table 7. 2007 Morbidity Valuation Results in 2005 dollars

2005 dollars	Washington	Oregon	Idaho
Morbidity Impact Functions Derived from Epidemiology Literature			
Chronic bronchitis	$167,151 ($12,909 - $526,918)	$199,596 ($15,408 - $629,179)	$89,213 ($6,836 - $282,039)
Nonfatal myocardial infarction 3% Discount Rate	$118,772 ($32,015 - $229,953)	$157,327 ($42,351 - $304,789)	$71,206 ($18,996 - $138,918)
Nonfatal myocardial infarction 7% Discount Rate	$114,922 ($29,488 - $226,058)	$152,236 ($39,010 - $299,639)	$68,878 ($17,494 - $136,531)
Emergency Room Visits- respiratory	$92 ($50 - $135)	$78 ($42 - $114)	$32 ($18 - $48)
Acute Respiratory Symptoms	$12,814 ($1,153 - $24,596)	$14,587 ($1,313 - $28,000)	$6,211 ($559 - $11,925)
Lower Respiratory Symptoms	$259 ($99 - $475)	$227 ($86 - $416)	$97 ($37- $178)
Upper Respiratory Symptoms	$299 ($82 - $649)	$263 ($72 - $571)	$111 ($31 - $241)
Work Loss Days	$12,422 ($10,823 - $14,021)	$12,643 ($11,016 - $14,272)	$4,791 ($4,173 - $5,409)
Acute Bronchitis	$576 ($21 - $1,391)	$504 ($18 - $1,217)	$217 (-$8 - $526)
Asthma Exacerbation	$658 ($66 - $1,780)	$580 ($58 - $1,569)	$234 ($23 - $634)
Hospital Admissions, Respiratory	$2,376 ($1,162 - $3,524)	$2,736 ($1,342 - $4,043)	$1,183 ($580 - $1,752)
Hospital Admissions, Cardiovascular	$6,223 ($3,879 - $8,546)	$8,195 ($5,115 - $11,249)	$3,533 ($2,205 - $4,857)

Table 7 summarizes the costs, in 2005 dollars, associated with the incidence of non-fatal illnesses and diseases associated with the air quality changes by state. These costs are not discounted, with the exception of nonfatal myocardial infarction, since they are expected to begin occurring immediately. In Washington State, the cost of 1.3225 additional cases of acute bronchitis is estimated to be $167,000 while the cost of the estimated 1.1101 additional myocardial infarctions at a 3% discount rate is approximately $119,000. The approximately 79 days of work lost are valued at $12,400; 473 additional cases of acute respiratory symptoms will cost $12,800.

In Oregon, the cost of new cases of acute bronchitis will approach $200,000. The cost associated with additional cases of acute respiratory symptoms will be almost $14,600, and the value of work days lost is expected to be $12,600. For Idaho, the costs associated with new cases of chronic bronchitis will be $89,000 while the cost of additional cases of acute respiratory systems will total $6,200. The value of work days lost will approach $5000.

Table 8. Total Valuation at a 3% Discount Rate in 2005 dollars

Total Valuation 3% Disc Rate in 2005 dollars	
ACS Study (Pope et al.)	$10,804,794
Harvard Six-City Study (Laden et al.)	$23,319,007
Expert E upper bound	$30,828,129
Expert K lower bound	$3,817,671

The total valuation results in Table 8 and Table 9 represent the full suite of mortality and morbidity costs for the population of the tri-state area; the estimates vary between authors based on the underlying mortality C-R functions utilized. The total valuation results at a 3% discount rate were created using the following formula: Total Valuation at a 3% discount rate = 3% Discounted adult mortality costs + 3% discounted infant mortality costs + 3% discounted acute myocardial infarction costs + all other undiscounted morbidity costs

The results presented in Table 8 represent the total valuation estimates from the change in incidence due to the concentration-response relationships for premature mortality presented by four authors, Pope et al., Laden et al., Expert E, and Expert K. The Laden et al. incidence change estimate multiplied by the premature death value of $6.6 million dollars provides a central estimate of costs while the value of the change in incidence calculated by Expert E and Expert K represent the upper and lower bound of such costs, respectively. The costs presented in Table 8 are higher than those presented below in Table 9 because of the difference in the rate of time preference for money or the discount rate. The change in incidence from Pope et al. (2004) translates to a total annual health cost of almost $11 million dollars; the change in incidence from the concentration-response function presented by Laden et al. (2006) translates a total health cost of approximately $23 million dollars. Expert E's calculated change in incidence corresponds to a total cost of almost $31 million dollars while Expert K' s

estimates of changes in health outcomes holds a total health cost value of about $4 million dollars.

Table 9. Total Valuation in 2005 dollars at a 7% Discount Rate

Total Valuation 7% Disc Rate in 2005 dollars	
ACS Study (Pope et al., 2004)	$9,922,351
Harvard Six-City Study (Laden et al., 2006)	$21,336,414
Expert E upper bound	$28,185,394
Expert K lower bound	$3,549,481

The total valuation results represent the full suite of mortality and morbidity costs for a population; the estimates vary between authors based on the underlying C-R functions utilized. The total valuation results at a 7% discount rate were created using the following formula:

Total Valuation at a 7% discount rate = 7% Discounted adult mortality costs + 7% discounted infant mortality costs + 7% discounted acute myocardial infarction costs + all other undiscounted morbidity costs

The results presented in Table 9 represent the total valuation estimates from the change in incidence due to the concentration-response relationships presented by four authors, Pope et al., Laden et al., Expert E, and Expert K. The value of the change in health outcomes utilizing the adult mortality function by Pope et al. (2004) is approximately $9.9 million dollars. The change in the incidence of health outcomes by

Laden et al. (2006) corresponds to a central estimate of health costs valued at a little over $21 million dollars, while the value attributed to Expert E's change in incidence calculation represents the upper bound of such costs at approximately $28 million dollars. The value of the change in incidence estimated by Expert K' s concentration-response function represents the lower bound of health costs at a value of approximately $3.5 million dollars.

Discussion

This case study of Washington, Oregon and Idaho found substantial human health impacts from $PM_{2.5}$ wildfire emissions. The particulate emissions were relatively small in terms of their impact on air quality, less than 2 $\mu g/m^3$ even at the most impacted locations. However, the estimated change in incidence of health outcomes was found to be quite large with up to 4.93 total adult premature deaths attributable to these emissions. Furthermore, the value of human health outcomes was estimated in the millions of dollars.

Since this very spatially and time limited case study found such large impacts, in light of future climate change scenarios which predict a potential increase in both the incidence of and area burned by wildfire, future impacts could be expected to be even larger.

Further analysis should concentrate on conducting sensitivity analyses to further illustrate the potential impacts of future climate change scenarios. While the

Intergovernmental Panel on Climate Change's predictions are purely qualitative, the Pew Center of Global Climate Change suggested quantitative increases in the number of square kilometers burned. Translating this estimate into impacts would further the science considerably and help inform future decisions by policy makers.

More analysis could focus on analyzing the values of these health outcomes along with the associated ecological costs of wildfire as well as the costs of wildfire suppression techniques in order to inform a full cost-benefit analysis of western wildfires. Additionally, a repetition of this analysis using a multi-scale air quality dispersion model such as the Community Multi-scale Air Quality dispersion model would better inform the full impact of wildfire particulate emissions. Better methods to measure particulate emissions from wildfires would also benefit science and policy decisions.

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Appendix- A

Data Transformation Procedures

To estimate human health impacts associated with wildfire emissions during this period, a number of steps were necessary. First, the initial data had to be transformed from hourly emissions into annual ambient air quality. The comma-delimited text file I received included the following variables: day, hour, ipt, jpt, latitude, longitude, kg/km^2/hr. Together, the ipt and jpt values referenced the 1 kilometer by 1 kilometer grid cell to which the emissions belonged. I transformed these emissions into a Microsoft Excel worksheet. In order to create a unique identifying number for each grid cell, I concatenated the ipt and jpt columns. The emissions were then transformed from kilograms of $PM_{2.5}$ to tons of $PM_{2.5}$ by multiplying by 0.001 to get tons/km^2/hr. Then, using a pivot table, I aggregated the hourly emissions into daily emissions. Next, I aggregated the total emissions for each individual grid cell. I then calculated annual ambient air quality for each grid cell by multiplying total tons/km^2/day by its geographically relevant impact-per-ton estimate and the days/days of the year for which that particular grid cell had emissions. The end result was air quality with units of μg/km^2/year.

The geographically relevant impact-per-ton estimate utilized in this analysis was designed by EPA in 1997 and reflects an estimate of the incremental change in $PM_{2.5}$ associated with incremental changes in tons of $PM_{2.5}$ precursors for the region

surrounding Missoula, Montana. This method of calculating air quality is less precise than the output from a multi-scale air quality model; however it provides a reasonable estimation given the time available and scope of this project. The use of an impact-per-ton estimate will result in conservative estimates of air quality because it does not allow for dispersion of $PM_{2.5}$ beyond the borders of its source cell.

I then took the air quality estimates ($\mu g/km^2/yr$) and imported them into ArcGIS. I also imported a 36km by 36 km Community Multiscale Air Quality (CMAQ) grid. I spatially joined the air quality estimates to the CMAQ grid cell. The air quality estimates which were on 1km by 1km scale were summed into the 36km by 36km scale during this spatial join. This enabled the air quality to be dispersed along a reasonable, yet conservative geographic scale to combat the lack of dispersion problem from the utilization of an impact ratio instead of air quality modeling. I then exported these aggregated air quality estimates back into a Microsoft Excel worksheet.

In the Microsoft excel worksheet I formatted the air quality estimates into the appropriate layout required by BenMAP, a benefits analysis and mapping tool. This layout included the variables: row, column, metric, seasonal metric, statistic, and values. The row and column variables reflected the appropriate referencing for unique identification of the 36km by 36km CMAQ grid cells. The metric was filled in with "D24HourMean", while seasonal metric was filled with "QuarterlyMean". The statistic was filled by "Mean" and values were filled in with the aggregated air quality estimates

for the appropriate grid cell. This file was called the control file. I also created a baseline file with all of the same variables. The values for the first five variables are identical to the previous, control, file while the values column was filled with 12 μg/36km^2/year which represents background levels of $PM_{2.5}$. The data is now in a form that can be put into BenMAP, the next tool utilized in the analysis.

Appendix-B

Steps in BenMAP

BenMAP allows for custom analyses or simplified analyses. I selected a custom analysis. In the custom analysis, the program allows users to input both baseline air quality estimates and scenario specific, called the control scenario, air quality estimates using the air quality grid creation button (U.S. EPA, 2007). This step also calculates the air quality change between the baseline air quality grid and the control air quality grid. I created a baseline air quality grid using the Model Direct option. I specified the Community Multiscale Air Quality 36km grid type as well as $PM_{2.5}$ for my pollutant. For my model database, I chose the baseline Excel file I previously created. After the model created the air quality grid, I saved this as my baseline grid. I repeated these steps to create a control air quality grid (.aqg file) using the control Excel file.

I next selected the Configuration Creation Method step. This step allows users to specify a variety of pollutant specific peer-reviewed health endpoints (U.S. EPA, 2007). I chose the Open Existing Configuration Method and selected the Particulate Matter National Ambient Air Quality Standard Configuration supplied to me by Neal Fann of the U.S. Environmental Protection Agency. This configuration file was that used in the most recent Particulate Matter National Ambient Air Quality Standard Regulatory Impact Analysis. Also in this step of the analysis, I specified the baseline and control air quality grids I created previously. I chose to compute 10 Latin Hypercube Points which

yields results broken down into 10 percentiles starting at 5% and ending at 95%. I also specified the population year to be 2007 and used the U.S. Census as my reference population data set. I set the Mortality Incidence Data set to the year 2020; this means the change in premature mortality incidence will be calculated from 2007 through the year 2020 to provide a conservative estimate of premature death incidence. Next, I ran this step and saved the results in a configuration results file (.cfgr file).

The next step in the analysis is the Aggregation, Pooling and Valuation Configuration Creation step. This step creates incidence and valuation estimates of the health impacts for the specified population based on the pooling and aggregation methods selected in this step (U.S. EPA, 2007). I chose the Open Existing Configuration File for Aggregation, Pooling, and Valuation option. In this option selection, I specified the PM NAAQS RIA configuration.apv file also supplied by Neal Fann of the EPA. This was the .apv file used in the most recent Particulate Matter National Ambient Air Quality Standard Regulatory Impact Analysis. After specifying this .apv file I selected the .cfgr file created by the previous step. In the advanced option window, I aggregated both the incidence and valuation estimates by state. I selected the currency year as 2005 and specified the Income Growth Adjustment Data Set as Income Elasticity for 3-21-2007. I set the year to 2007 and selected all available endpoints. This step resulted in an Aggregation, Pooling and Valuation Results file (.apvr file).

In the final reports step, BenMAP creates an incidence report indicating estimates for occurrence of each specific health endpoint (U.S. EPA, 2007). Then the incidence report is translated into monetized human health costs using peer-reviewed valuation functions (U.S. EPA, 2007). The final output is a geographically specific incidence and valuation report of the human health costs (or benefits) of changes in ambient $PM_{2.5}$ levels. During this step, I chose to report the Incidence and Valuation Results based on the .apvr file I created in the previous step. I then selected the pooled incidence option and chose to view Endpoint, Endpoint Group, Author, Start Age and End Age. I then saved this report as an Excel File. I repeated these steps in order to create the pooled valuation report.

Then, I post-processed these report results. Since, I joined the air quality estimates to the CMAQ $36km^2$ grid cells by intersection, portions of the outermost grid cells fell in the surrounding states of California, Montana, Wyoming, Utah, and Nevada. To fix this problem, I added the results from California and Nevada to Oregon's results. I also added the results from Montana, Wyoming, and Utah to Idaho's results.

Next, I discounted both the adult (Laden et al., Pope et al., and Experts A-L) and infant mortality (Woodruff) valuation estimates at a 3% discount rate. However, since the mortality costs do not occur on a linear basis I used an approximation of 0.91. Additionally, I computed the discounted adult and infant mortality costs using a 7% discount rate which is approximated by 0.87.

BenMAP already outputs discounted costs for acute myocardial infarctions at both a 3% and 7% discount rate. The morbidity costs do not need to be discounted because they are assumed to commence immediately.