Air Quality Impacts and Benefits under U.S. Policy for Air Pollution, Climate Change, and Clean Energy

by

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Submitted to the Engineering Systems Division in
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Abstract

Policies that reduce greenhouse gas emissions can also reduce outdoor levels of air pollutants that harm human health by targeting the same emissions sources. However, the design and scale of these policies can affect the distribution and size of air quality impacts, i.e. who gains from pollution reductions and by how much. Traditional air quality impact analysis seeks to address these questions by estimating pollution changes with regional chemical transport models, then applying economic valuations directly to estimates of reduced health risks. In this dissertation, I incorporate and build on this approach by representing the effect of pollution reductions across regions and income groups within a model of the energy system and economy. This new modeling framework represents how climate change and clean energy policy affect pollutant emissions throughout the economy, and how these emissions then affect human health and economic welfare. This methodology allows this thesis to explore the effect of policy design on the distribution of air quality impacts across regions and income groups in three studies. The first study compares air pollutant emissions under state-level carbon emission limits with regional or national implementation, as proposed in the U.S. EPA Clean Power Plan. It finds that the flexible regional and national implementations lower the costs of compliance more than they adversely affect pollutant emissions. The second study compares the costs and air quality co-benefits of two types of national carbon policy: an energy sector policy, and an economy-wide cap-and-trade program. It finds that air quality impacts can completely offset the costs of a cost-effective carbon policy, primarily through gains in the eastern United States. The final study extends the modeling framework to be able to examine the impacts of ozone policy with household income. It finds that inequality in exposure makes ozone reductions relatively more valuable for low income households. As a whole, this work contributes to literature connecting actions to impacts, and identifies an ongoing need to improve our understanding of the connection between economic activity, policy actions, and pollutant emissions.

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Acknowledgments

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Introduction

This dissertation comprises three studies on the topic of air quality impacts and co-benefits under policy for clean air, clean energy, and climate change. Its aim is to develop a methodological framework that can explore how the design of policy affects the distribution of air quality impacts and benefits. Its contributions are thus methodological and policy-relevant.

This dissertation makes a methodological contribution to the strand of the broader sustainability science literature that studies the links between the grand sustainability challenges of air pollution, clean energy, and climate change. The sustainability science literature listed its core questions in Kates et al. (2001), including the need to illuminate the dynamics and incentives of sustainable systems. Scholars of air quality impacts have pointed to interactions with climate change and climate policy as a significant link in assessing sustainable solutions, while citing epistemic divides as a reason that air quality co-benefits have lacked policy traction. Nemet et al. (2010) and Thompson et al. (2014) describe the need to link methods from climate policy analysis to air quality impacts in order to capture the full policy-to-impacts pathway and illuminate the dynamics and incentives that influence the air quality co-benefits of climate policy.

To contribute to addressing this methodological need, this dissertation develops an integrated modeling framework that combines advanced models used for energy and climate policy analysis with an air quality impacts modeling system used for U.S. regulatory analysis. The air quality modeling system includes a pollutant emissions model with detailed emissions sources used to

produce hourly, gridded emissions, the Sparse Matrix Operating Kernel Emissions model, or SMOKE (CMAS 2010). It converts emissions into ambient pollutant concentrations with a regional chemical transport model, the Comprehensive Air Quality Model with extensions, CAMx (Environ International Corporation 2013). It estimates human health impacts related to two criteria air contaminants, ozone and fine particulate matter, using the Benefits Mapping and Analysis Program, BenMAP (Abt Associates, Inc. 2012). This dissertation's innovation is to add an energy and climate policy analysis model, the U.S. Regional Energy Policy Model (Rausch et al. 2010), to the start and end of this process. Completing this loop allows us to capture feedbacks in the economy that affect economy-wide pollutant emissions under policy, and to capture the nonlinear chemistry and concentration-response functions that transform emissions into human impacts.

Representing these linkages and feedbacks within a common framework allows us to complement existing studies of the air quality impacts of policy, while also answering new questions about how policy design affects the distribution of these impacts. This dissertation comprises three studies that each develops a new aspect of this modeling framework in order to answer a different policy-relevant question.

First Essay

Air Quality under State-Level Limits or Regional Trading to Regulate CO_2 from Existing Power Plants

The methodological development of the first essay is to link the economic output of a Computable General Equilibrium (CGE) economic model, USREP, with state-level detail, to a regulatory emissions modeling system, SMOKE. Typically, in air quality co-benefits analysis, a model like SMOKE would be driven by emissions changes from engineering estimates or electricity modeling results that are limited to the electricity sector. Our approach complements this technique by using an economy-wide model to capture emissions changes outside of the electricity sector, e.g., from emissions due to extracting, processing, and transporting fossil fuels. The technique follows Thompson et al. (2014), matching sectoral input and output in USREP to detailed source classification codes in SMOKE to develop a future emissions inventory under energy or climate policy.

Developing this approach allows us to compare economy-wide policy costs and emissions changes under U.S. energy policy. Specifically, we study the type of state-level carbon intensity limits proposed in the EPA Clean Power Plan (CPP), and compare the effect of state, regional, and national compliance. This question is motivated by the fact that the CPP allows this flexibility in achieving compliance, and that this will likely reduce costs but can increase pollutant emissions, potentially shifting emissions to regions with elevated pollutant damages. In other words, the net effect on co-benefits and costs is unclear.

This study suggests that there are important emissions reductions from non-electricity sector sources, accounting for up to 15% of fine particulate matter reductions and up to one third to two thirds of reductions in carbon monoxide, volatile organic compounds, and ammonia. It also suggests that the more flexible implementations of the CPP, including national and regional compliance, might increase net co-benefits. However, to make a commensurate comparison, one would have to convert emissions changes into co-benefits. Complex nonlinear relationships link pollutant emissions to their resulting health and economic impacts. The Second Essay completes this next link in order to compare costs and air quality co-benefits for two types of carbon policy.

Second Essay

A Self-Consistent Method to Assess Air Quality Co-Benefits from U.S. Climate Policies
The methodological development of the second essay is the mirror of the first: it links the output of an air quality impact modeling system to a Computable General Equilibrium (CGE) economic model, USREP. Using the air quality modeling system captures nonlinear chemical transformation and concentration-response relationships, as is done in regulatory air quality impact analysis. Representing these impacts within a CGE framework complements this approach by estimating the economic welfare impacts of pollution-related health effects under the constraints of multiple interacting policies, limited resources, and price responses. This approach also addresses a gap, identified by Nemet et al. (2010), that models commonly used to assess climate policies, like CGE models (Paltsev and Capros 2013), do not include air quality co-benefits. Further, it extends existing comparisons of costs and co-benefits, such as Thompson et al. (2014), by using a common, more self-consistent, economy-wide approach to estimate costs and co-benefits.

Currently, the co-benefits literature spans many methods and many carbon policy types, making it difficult to draw comparisons (Nemet et al. 2010). The self-consistency offered by this new modeling framework allows us to directly compare costs and co-benefits for multiple policy types. The second essay employs this framework to ask whether co-benefits can exceed the costs of an energy sector or economy-wide carbon policy. The energy sector policy limits CO_2 emissions by specifying a percentage of electricity sales from low-carbon sources. The economy-wide policy limits the same amount of CO_2 emissions but allows reductions from anywhere in the economy via a cap-and-trade program. We find that the more cost-effective cap-and-trade instrument has median air quality co-benefits that exceed its policy costs. We find that including air quality co-benefits shifts the distributional effects of the policy to favor eastern states. Finally, we find that general equilibrium effects have a relatively minor effect on the distribution of air quality co-benefits, but can shift them slightly toward high productivity regions.

This study explores the distribution of air quality impacts by location. Another key dimension of a policy's distributional implications is how its effects vary with income. The third essay introduces the ability to distinguish air quality impacts not only by location but also by income group.

Third Essay

Human Health and Economic Impacts of Ozone Reductions by Income Group
The methodological development of the third essay is to represent the health-related economic impacts of ozone pollution with household income. It builds on USREP's ability to track economic welfare among representative households in nine income categories. Along with economy and environment, social equity is a key pillar of sustainability, meaning that representing economy-wide effects of environmental policy with income is needed, though empirical and theoretical challenges remain. This method allows us to explore the consequences of current approaches common in CGE modeling and regulatory analysis for the assessment of ozone policy with income.

Previous studies find inequality in ozone exposure and the effect of policy, which motivates analyzing U.S. ozone policy with income (Bell and Dominici 2008; Bento, Freedman, and Lang

2014). We use this approach to estimate the relative economic value of ozone reductions by household income category. We use a modeling scenario that compares 2005 ozone levels to a suite of reductions that were planned for 2014. We compare the effect of potential outcomes of policy – like unequal reductions with income, and delay – with different valuation techniques. We find that ozone reductions appear relatively more valuable to low income households, who are also relatively more affected by delay. The factor having the greatest effect on the relative value of reductions was the valuation technique, followed by delay and the effect of accounting for disproportionate ozone reductions among low-income households.

Conclusion

This dissertation develops a modeling framework that can be used to aid the analysis of climate, energy, and air quality policy in terms of the three pillars of sustainability: economy, environment, and equity. It illuminates the dynamics between climate policy and air quality impacts and their distributions under different types of economic incentives. Using this new technique serves to bolster some findings while highlighting remaining gaps. For example, the second essay reaffirms the prevalent claim that air quality co-benefits can exceed policy costs for a cost-effective instrument, this time with a more self-consistent comparison. Conversely, the first essay finds that energy-sector models may miss up to 15% of $PM_{2.5}$ reductions, and even more reductions of precursor contaminants. The third finds that, while ozone-related health risk inequality persists, there are perhaps greater opportunities to understand and address economic inequality.

Some of the co-benefits literature motivates its work as a means to reduce the apparent costs of climate mitigation, to inform multi-pollutant management, or to produce assessments of sustainable development and risk inequality (Ravishankara, Dawson, and Winner 2012; Nemet, Holloway, and Meier 2010; Fann et al. 2011). This dissertation provides insights that bear on the relative significance of air quality co-benefits to climate mitigation costs, the interplay of CO_2 reductions and pollutant emissions, and the intersection of environmental policy and income inequality. In so doing, it serves to quantify and compare the relative importance of several ways in which air pollution is no longer an isolated public health issue, but one component of the challenge to develop informed policy within sustainability systems.

First Essay

1. Air Quality under State-Level Limits or Regional Trading to Regulate CO_2 from Existing Power Plants

INTRODUCTION

The energy sector is a significant source of greenhouse gas emissions that cause climate change as well as air pollutants that harm human health (Burtraw et al. 2003). The EPA recently proposed the Clean Power Plan (CPP) to limit emissions of CO_2 from existing power plants (U.S. Environmental Protection Agency 2014a). The CPP proposes state-specific carbon intensity limits, but allows regional compliance. For example, several states could work together to meet a combined limit. The CPP is also expected to yield significant benefits for ozone and fine particulate matter, in excess of policy costs and direct domestic benefits from reducing CO_2 (U.S. Environmental Protection Agency 2014a). If regional or national compliance results from the CPP, it has the potential to reduce both costs and air quality impacts, likely at different rates. Previous analyses of the CPP have focused on energy-sector emissions (U.S. Environmental Protection Agency 2014a; Driscoll et al. 2014), but changes to emissions in non-energy sectors can also be significant (Rausch, Metcalf, and Reilly 2011; Thompson et al. 2014), including upstream effects in coal producing states (Larsen 2014). Here, we build on previous analyses by capturing economy-wide emissions changes and directly exploring the effect of state-level, regional, or national implementation on air pollutants. We use this analysis to examine the effect of the geographic scale of trading to meet state-level CO_2 limits on the size and distribution of air pollutant emissions.

The proposed EPA CPP designates state-level CO_2 intensity limits, but permits states to achieve "regional compliance" through emissions trading. The ability to trade will likely have different effects on the costs and air quality co-benefits of the rule. Trading will likely lower the policy costs by increasing access to low-cost abatement opportunities. Trading is also likely to affect air quality co-benefits. The response of air quality impacts is complicated by that fact that the marginal damages of pollution vary by source, time, and location (Tietenberg 1995). Some studies suggest accounting for this can increase the benefits and economic efficiency of markets for clean air such as the NO_X Budget Program or Acid Rain Trading program (Mesbah, Hakami, and Schott 2014; Spencer Banzhaf, Burtraw, and Palmer 2004; Mauzerall et al. 2005; Tong et al. 2006; Graff Zivin, Kotchen, and Mansur 2014; Muller and Mendelsohn 2009). Conversely, accounting for the varying damages of co-emitted pollutants in greenhouse gas policy has a more

complicated effect, and may not be socially beneficial (Muller 2012). Allowing trading is likely to affect costs more than co-benefits, based on previous studies comparing energy sector and economy-wide policies (Thompson et al. 2014; Saari et al. 2015). Thus, the spatial extent of trading for a carbon policy will likely yield cost savings and may reduce air quality co-benefits. Therefore we hypothesize that allowing trading will decrease costs but may increase pollutant emissions.

In particular, trading may reduce air quality co-benefits in already polluted or populated areas where the damages from pollution are high. Trading will tend to shift emissions to areas with low marginal abatement costs for CO_2, but some of these areas might also have high marginal damages from pollution. The potential for emissions trading to lead to pollution "hotspots", or areas of high concentrations, has been studied for toxic pollutants (Adelman 2013) and criteria pollutants emitted from the energy sector as in the acid rain trading program (Burtraw and Mansur 1999; Shadbegian, Gray, and Morgan 2007). A recent study showed that SO_2 trading shifted emissions from low to high damage areas (Henry, Muller, and Mendelsohn 2011). Others have explored whether the differing incentives of state-level or regional-level compliance might lead to 'environmental browning' (Bellas and Lange 2008). The air quality co-benefits of the CPP provide a complex case for several reasons. First, the trading will focus on CO_2 while the air quality impacts will result from multiple co-emitted pollutants. Second, states are subject to non-uniform targets that depend on existing conditions, opportunities, and commitments. The interaction of the differing limits, costs, and damages will determine how the extent of trading affects high-damage areas. Since air quality impacts comprise a large share of the benefits, the potential outcome of trading is important to explore because it affects the distributional implications of this policy.

Here, we explore how potential trading to meet state-level CO_2 limits will affect the size and distribution of air pollutant emissions. We implement these policies using a recently developed U.S. energy and economic model with state-level detail (the U.S. Regional Energy Policy Model, USREP) to assess economy-wide costs and emissions changes. We connect this state-level model to an advanced emissions modeling system following Thomson et al. (2014). For each policy scenario, we compare the resulting changes in emissions of criteria pollutants. We

examine trade-offs in policy cost and air quality improvement at various scales. We draw conclusions from this work to inform strategies for policy implementation.

INTEGRATED ASSESSMENT METHODS

Integrated Modeling Framework

We model policies and resulting emissions using an integrated assessment framework similar to Thompson et al. (2014). This framework links the economic model USREP, used to estimate policy costs and economy-wide CO_2 emissions changes, to an advanced system for modeling air quality impacts.

The U.S. Regional Energy Policy Model

USREP is a computable general equilibrium (CGE) economic model designed to study the long-run dynamics of resource allocation and income distribution under energy and environmental policy. It is a recursive dynamic model that calculates the commodity prices that support equilibrium between supply and demand in all markets. USREP is suited to exploring the environmental impacts and distributional implications of U.S. national and sub-national energy policy because it includes rich detail in its energy sector, relates production to emissions of greenhouse gases, and represents multiple sectors, regions, and income groups. The version of USREP used in this study features the ability to simulate state-specific targets, updated 2012 energy and economic data, and international trade; it also solves in two-year periods (Caron, Rausch, and Winchester 2015; Caron, Metcalf, and Reilly 2015). This state-level version was developed recently and its results are considered to be preliminary. This means that we use it here to demonstrate the methodology, and to examine the general effects of the policy implementation scale, but not to predict specific state-level outcomes. Previous studies using USREP: describe its structure and inputs; present climate change and energy policy applications; test sensitivity to inputs, structure and assumptions; and compare its performance to other energy and economic models (Rausch, Metcalf, and Reilly 2011; Thompson et al. 2014; Saari et al. 2015; Caron, Rausch, and Winchester 2015; Rausch and Mowers 2014; Rausch et al. 2010; Rausch and Karplus 2014; Lanz and Rausch 2011).

Table 1-1: USREP Model Details: Household, and Sectoral Breakdown and Primary Input Factors; Regional Breakdown for the Regional-Level Implementation

Sectors	Regions	Primary production factors
Non-Energy	Pacific (PACIF)	Capital
Agriculture (AGR)	California (CA)	Labor
Services (SRV)	Alaska (AK)	Coal resources
Energy-intensive products (EIS)	Mountain (MOUNT)	Natural gas resources
Other industries products (OTH)	North Central (NCENT)	Crude oil resources
Commercial transportation (TRN)	Texas (TX)	Hydro resources
Passenger vehicle transportation (TRN)	South Central (SCENT)	Nuclear resources
Final demand sectors	North East (NEAS)	Land
Household demand	South East (SEAST)	Wind
Government demand	Florida (FL)	
Investment demand	New York (NY)	Household income classes
Energy	New England (NENGL)	($1,000 of annual income)
Coal (COL)		<10
Natural gas (GAS)		10-15
Crude oil (CRU)		15-25
Refined oil (OIL)		25-30
Electric: Fossil (ELE)		30-50
Electric: Nuclear (NUC)		50-75
Electric: Hydro (HYD)		75-100
Advanced Technologies		100-150
		>150

Linking USREP to Pollutant Emissions

Following Thompson et al. (2014), we link production in USREP to the relevant emissions sources in a national emissions inventory for 2005. This emissions inventory is temporally processed and speciated on a 36-km grid of the continental U.S. using the same year-long 2005 modeling episode described in Thompson et al. (2014), and documented and evaluated against ambient monitors in U.S. EPA (2011b). Here, we use the state-level output from USREP in 2020 to scale the corresponding detailed anthropogenic point and area sources in each state. We compare the changes in fuels used in the electricity sector, energy intensive industries, and other industry, as well as the outputs of the sectors of agriculture, electricity, fossil fuel production, service, transportation, and other manufacturing. These variables are mapped to hundreds of pollutant source classification codes depending on their fuel use and industrial classification, and used to scale the emissions profiles of the relevant sources. One advantage of this approach is that it allows a policy like the Clean Power Plan, which regulates a single sector, to affect

emissions across the economy. For example, reducing demand for coal in power generation affects the pollutant emissions associated with mining, processing, and transporting coal. A disadvantage is that this framework does not endogenously include pollution-specific post-process abatement options to reduce pollution under future air quality policies. Thus, our scenarios are scaled from emissions factors based on current pollution abatement levels and regulations. They should thus be interpreted as the effect of the carbon policies apart from any decisions regarding air pollution, though there may be important interactions.

We use these scaling factors in the SMOKE model to produce gridded, hourly emissions for each scenario in the year 2020. We estimate changes to criteria air contaminants and important precursors for the formation of fine particulate matter and ozone, including sulfur dioxide (SO_2), nitrogen oxides (NO_X), fine particulate matter ($PM_{2.5}$), carbon monoxide (CO), volatile organic compounds (VOC), and ammonia (NH_3).

Analyzing State-Level, Regional, and National Policy Compliance

In this analysis, we represent the state-level limits for the electricity sector defined in the CPP. These are rate-based intensity limits which we convert to equivalent mass-based limits, as permitted by the rule. The EPA had computed targets based on four building blocks for reductions, including: 1. heat rate improvements; 2. re-dispatch to lower emission sources (e.g., switch from coal to natural gas); 3. expanded low-carbon generation (e.g., increase renewables penetration); and 4. demand-side measures. States are free to choose their means of compliance. By imposing state-level caps for CO_2 from electricity, our model allows states to endogenously choose between building blocks two through four to meet their targets.

We compare three policy scenarios: state-level compliance (State), regional trading (Regional), and national trading (National). In the Regional scenario, states can meet their combined limits within 12 regions which were determined to capture differences in electricity costs and are depicted in Figure 1-1 (Rausch et al. 2010). These regions comprise the states of California, Florida, New York, and Texas, as well as several multistate aggregations. In the National scenario, states can trade with any other state. We compare each policy scenario to the Business-as-Usual (BAU). The resulting changes in production are used to estimate future emissions for

each policy. We model emissions in the continental U.S. only, which excludes Alaska and Hawaii.

Figure 1-1: USREP Regions used in Regional Implementation.
They are the aggregation of the following states: NEW ENGLAND = Maine, New Hampshire, Vermont, Massachusetts, Connecticut, Rhode Island; SOUTH EAST = Virginia, Kentucky, North Carolina, Tennessee, South Carolina, Georgia, Alabama, Mississippi; NORTH EAST = West Virginia, Delaware, Maryland, Wisconsin, Illinois, Michigan, Indiana, Ohio, Pennsylvania, New Jersey, District of Columbia; SOUTH CENTRAL = Oklahoma, Arkansas, Louisiana; NORTH CENTRAL = Missouri, North Dakota, South Dakota, Nebraska, Kansas, Minnesota, Iowa; MOUNTAIN = Montana, Idaho, Wyoming, Nevada, Utah, Colorado, Arizona, New Mexico; PACIFIC = Oregon, Washington, Hawaii.

RESULTS OF COMPLIANCE SCALE ON PRODUCTION

Effects on Fossil Fuel Input to Electricity and Other Sectors

The scale of compliance, be it national, regional, or state, has different effects on fossil fuels used in electricity production, as shown in Figure 1-2. The largest effects are to coal, which decreases by about 13% in 2020 compared to Business-as-Usual. National and regional compliance both reduce about 1 percentage point more coal use in electricity production than state-level compliance. For both natural gas and oil, the regional and state level implementations have similar, larger decreases in fossil fuel use than the national implementation.

Fossil Input to Electricity vs. BAU in 2020 (%)
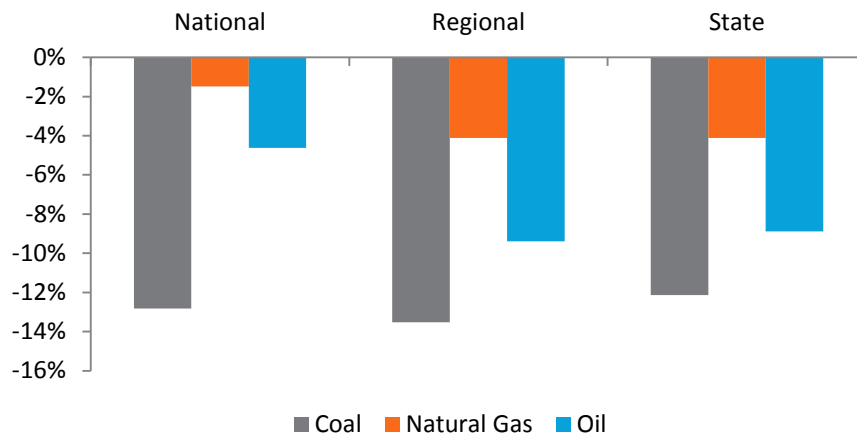
Figure 1-2: Change in Fossil Input to Electricity by Scenario Compared to Business-as-Usual in 2020.

As the use of fossil fuels decreases in electricity generation, it increases slightly in other sectors. Fossil fuel use increases in energy intensive industries and other activities between 0% and 1%. These increases do not compensate for the reduced fossil fuel use in electricity, as the majority of fossil fuels are used in the electricity sector, meaning that overall fossil fuel use in the economy decreases. For example, the decrease in coal use in electricity is over 100 times greater than the increase in coal used in electricity intensive industries.

Fossil Input to Other Sectors vs. BAU in 2020 (%)
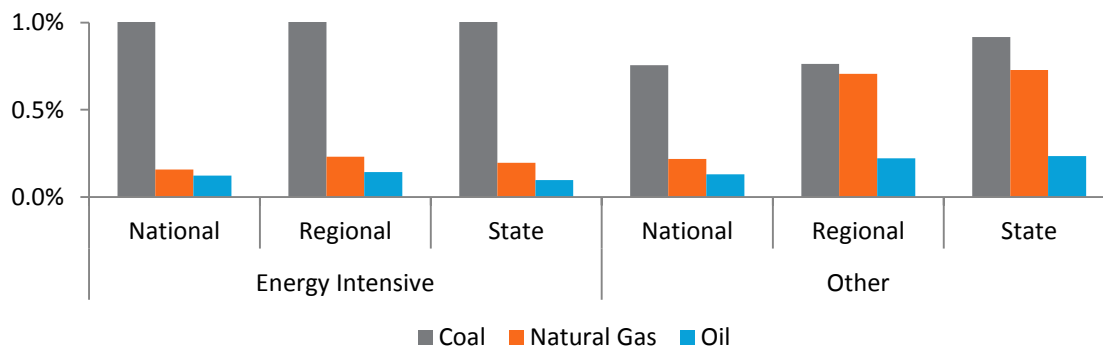
Figure 1-3: Change in Fossil Input to Other Sectors by Scenario Compared to Business-as-Usual in 2020.

Effects on Economic Output

For most sectors, the policy scenarios reduce economic output compared to Business-as-Usual. Output is changed by less than 0.2%, with the exception of fossil-powered electricity generation and fossil production, particularly coal and natural gas production. The majority of sectors have their output reduced by an amount that increases inversely with the extent of trading, from national, to regional, to state. In other words, the more flexible implementations have a smaller effect on economic output from fossil-based power generation. This relationship is not linear. In these initial findings, Regional and State are more similar to each other in their main effect on economic output, i.e., to fossil-fueled power generation, than they are to National. This trend also holds for most other sectors as well, with the national-level implementation having the smallest effect on all sectors, except that the regional implementation has slightly smaller effects on agriculture, refined oil, and service. Similarly, the regional-level implememtation has smaller effects on all sectors than the state-level, with the exception of agriculture and energy-intensive industries.

Effects on Economic Welfare Per Capita

The more flexible the implementation, the lower is the estimated consumption loss from the policy. Initial estimates of the consumption loss per capita in 2020 are inversely proportional to the extent of trading. Moving from state-level compliance to regional compliance reduces costs by about 20%, and allowing national compliance reduces costs by about 50%.

Reduction in Energy Sector Emissions by Pollutant

Emissions from the electricity sector are reduced for all pollutants in all implementation scenarios, as shown in Table 1-2. Compared to Business-as-Usual, by pollutant, reductions are highest for SO_2, followed by NO_X, CO, primary $PM_{2.5}$, VOC, and ammonia (NH_3). For most pollutants, reductions are similar across scenarios, except for SO_2 and CO; for both of these, National has a smaller effect than State or Regional, which echoes their effects on output from electricity production.

Table 1-2: Change in Pollutant Emissions from Electricity by Scenario in 2020 vs. BAU (thousand tons/year)

Scenario	SO_2	NO_X	$PM_{2.5}$	CO	VOC	NH_3
National	-932	-370	-47	-53	-3	-2
Regional	-998	-378	-49	-85	-4	-2
State	-993	-331	-48	-80	-3	-2
Percent Reductions from Fossil Fuel Based Electricity (%)						
National	-11%	-12%	-12%	-10%	-10%	-8%
Regional	-12%	-12%	-12%	-16%	-11%	-10%
State	-12%	-11%	-12%	-15%	-10%	-9%

The total emissions reductions in Table 1-2 correspond to a percentage reduction between 8% and 16% across these pollutants. For every pollutant, the Regional scenario reduces the most emissions. The more flexible National scenario always has higher emissions than the Regional scenario, but the results of the State scenario are more mixed. Overall, there is a relatively small percentage change between scenarios. The reductions are within 1 or 2 percentage points for all pollutants but CO. This result is consistent with the EPA's assessment of the Clean Power Plan, and Driscoll's assessment of three related policy scenarios (U.S. Environmental Protection Agency 2014a; Driscoll et al. 2014).

Total Emissions Reductions by Scenario and Pollutant

Considering emissions from all sources, total emissions of all pollutants are reduced under all scenarios compared to BAU, as shown in Table 1-3 and Figure 1-4. This means that the net effect of the policy is to reduce pollutant emissions overall, though emissions from some sources may increase; for example, emissions of SO_2 from area sources increases under the policy, which may be due to the increase in coal use outside the electricity sector. While the majority of emissions reductions come from the electricity sector, the policy scenarios indirectly affect emissions in energy intensive industries, transportation, and other sectors. Overall, the electricity sector accounts for nearly 100% of reductions in SO_2 and NO_X, 85%-92% in primary $PM_{2.5}$, and half to two thirds of reductions in CO, VOC, and ammonia.

Table 1-3: Change in Total Pollutant Emissions by Scenario in 2020 vs. BAU (thousand tons/year)

	SO_2	NO_X	$PM_{2.5}$	CO	VOC	NH_3
Total Emissions Reductions						
National	-932	-375	-52	-81	-8	-2
Regional	-1,000	-387	-56	-125	-10	-3
State	-999	-341	-56	-123	-12	-3
Percent of Total Emissions Reductions from Electricity						
National	100%	99%	92%	65%	45%	66%
Regional	100%	98%	88%	68%	36%	59%
State	99%	97%	85%	65%	27%	48%

Overall, the percent of total reductions are small except for SO_2, followed by NO_X and primary $PM_{2.5}$, as shown in Figure 1-4. For all pollutants except NO_X, Regional and State have similar, lower emissions than National. Moving from State level compliance to Regional or National will typically increase emissions, i.e., lower the percent reductions. Just as costs drop with a more flexible policy, so does the size of emissions reductions. Table 1-4 compares the size of total emissions reductions under National, Regional, and State. For example, a National implementation will decrease SO_2 reductions from 999 to 932, or by 7%. National gains 10% more NO_X reductions, but erodes 8% of primary $PM_{2.5}$ reductions and about 30% of reductions in CO, VOC, and ammonia.

Table 1-4: Percent Change in Emissions Reductions Compared to State in 2020

	SO_2	NO_X	$PM_{2.5}$	CO	VOC	NH_3
National vs. State	7%	-10%	8%	34%	40%	31%
Regional vs. State	0%	-12%	0%	-2%	27%	3%

Total Emissions in 2020 vs. BAU (%)
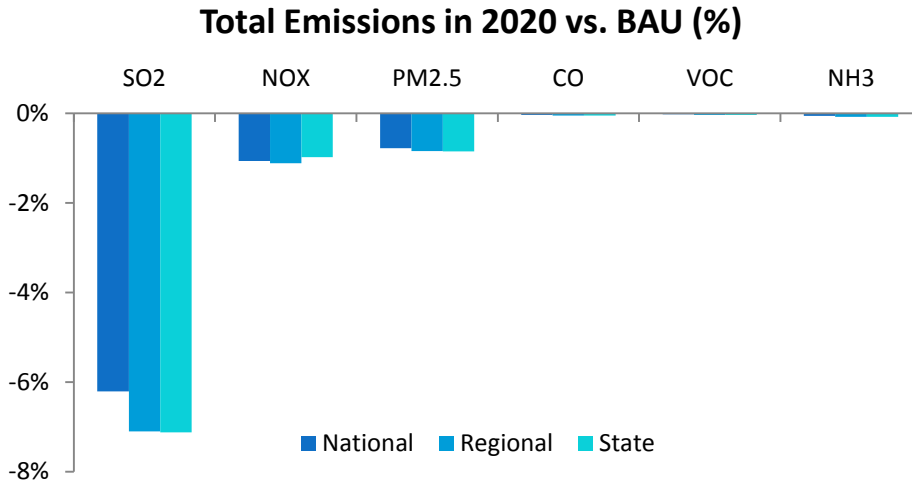
Figure 1-4: Change in Pollutant Emissions by Scenario Compared to Business-as-Usual in 2020.

Emissions Reductions by Location

Figure 1-5 shows the effect of National vs. State or Regional vs. State implementations on the change in total primary $PM_{2.5}$ emissions in 2020. Primary $PM_{2.5}$ emissions will not represent total ambient $PM_{2.5}$, much of which will be formed from precursor emissions, and these preliminary results are not meant to predict outcomes in a given state. Nonetheless, they can be used to hypothesize whether the air quality co-benefits may shift to high damage areas. The majority of benefits will be due to fine particulate matter. Fann et al. (2011) mapped the public health burden from $PM_{2.5}$ and ozone, showing the highest damages in the Eastern U.S. and California. Comparing National vs. State, the more flexible policy does qualitatively appear to increase emissions more in high-damage than low-damage areas. Regional vs. State implementation has a similar but less pronounced pattern.

DISCUSSION

Regardless of their implementation, the state-level carbon intensity limits explored here are estimated to decrease fossil fuel use in electricity production, decrease economic output from fossil-powered electricity, impose a policy cost, and decrease pollutant emissions compared to Business-as-Usual. We compare three different implementation scenarios, State, Regional, and National, representing three different scales of compliance. National, the most flexible approach, has the lowest policy cost, and the least effect on fossil fuel use, economic output, and total

emissions for each pollutant. The effect of a more flexible policy implementation is generally to lower costs, increase economic output, and increase pollutant emissions.

Annual Emissions of PM$_{2.5}$
National vs. State

(a)

Annual Emissions of PM$_{2.5}$
Regional vs. State

(b)
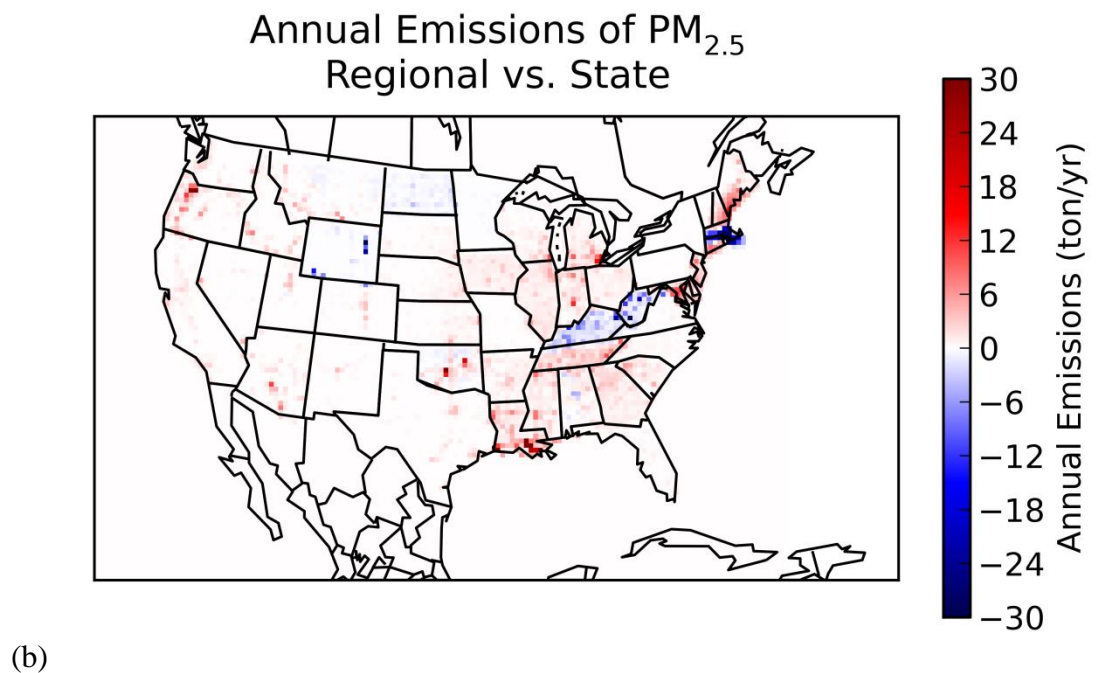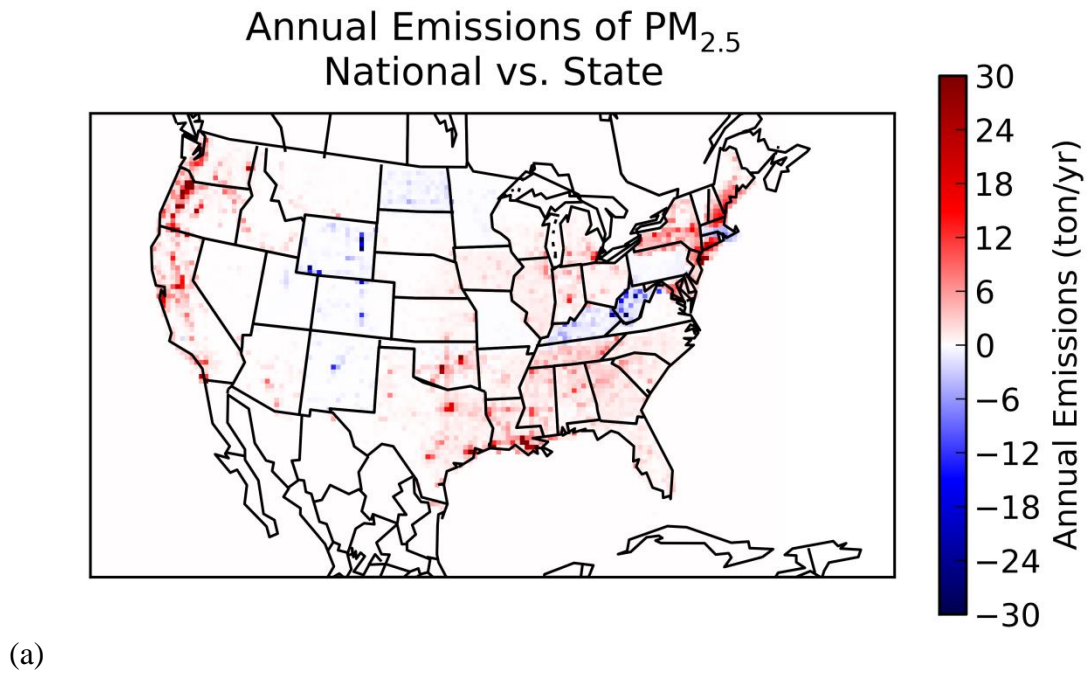
Figure 1-5: Change in Total Primary PM$_{2.5}$ Emissions Compared to State in 2020. (a) National vs. State (b) Regional vs. State.

All implementation scenarios reduce emissions of all pollutants compared to BAU. The largest reductions are to SO_2, NO_X, CO, and primary $PM_{2.5}$. The largest impact of each policy scenario is to electricity generated from fossil fuels. Coal use in this sector drops by 13%, its output drops, and its emissions drop between 8% and 16%.

Unlike previous analyses (Driscoll et al. 2014; U.S. Environmental Protection Agency 2014a), we capture non-electricity sector emissions changes. Reductions in fossil-based electricity account for 85%-100% of the total emissions reductions in SO_2, NO_X, and primary $PM_{2.5}$. Thus, previous analyses that only focus on this sector will capture most of the air quality benefits. Nonetheless, non-electricity sector effects add an additional 15% reduction in primary $PM_{2.5}$, and one third to two thirds additional reductions of CO, VOC and ammonia, which are important precursors of ozone and secondary fine particulate matter. Thus, capturing these economy-wide emissions can increase air quality co-benefits estimates compared to previous analyses.

Economy-wide, the largest percent emissions reductions compared to BAU are from SO_2 (7%), NO_X (1%), and primary $PM_{2.5}$ (1%). Emissions are fairly similar under all three implementation scenarios. National reduces fewer emissions than State or Regional for nearly all pollutants, which follows from its relative effects on economic activity. The exception is NO_X, which is slightly higher under State than National or Regional. This small increase in NO_X from electricity generation and industrial sectors under a carbon policy was also found by Rudokas et al. (2015). In industry, they attribute this to a switch to combined heat and power. In the electricity sector, they cite delayed investments, reduced efficiency due to carbon capture, and leakage outside the Clean Air Interstate Rule area. In our case, State has higher NO_X emissions perhaps due to its higher switching from coal to natural gas fired electricity that can have higher NO_X emission factors even though natural gas combustion typically emits less SO_2 and primary $PM_{2.5}$.

The effect of the implementation scenario on costs is much higher than the effect on emissions, on a percentage basis. National costs about 50% less than State, while it loses around 10% of SO_2 and $PM_{2.5}$ reductions, but gains about 10% in NO_X reductions. This suggests that allowing a flexible implementation could increase net co-benefits. However, when examining the

distribution of emissions by location, it is also possible that National could tend to shift emissions to high-damage areas where mortality risks from fine particulate are already relatively elevated. In order to examine these competing effects, a commensurate comparison of costs and co-benefits would be needed to compute the economic benefits from these foregone emissions reductions.

Our estimated emissions reductions are comparable with EPA's analysis of the Clean Power Plan, which quotes results for NO_X, SO_2, and $PM_{2.5}$. For the regional compliance case, our reductions are 8% higher for NO_X and 13% lower for $PM_{2.5}$. For state-level compliance, our reductions are 11% and 22% lower for NO_X and $PM_{2.5}$, respectively. Conversely, our estimates of SO_2 are higher by 71% and 66% for Regional and State, respectively. Our higher estimates of SO_2 reductions are mainly attributed to our use of 2005-level pollution control and policy, which does not include regulations like the Mercury and Air Toxics Standard (MATS) that is expected to reduce power-sector SO_2 emissions by 40% by 2015.

There are several additional reasons why our reductions may differ from previous analyses, including how we represent the policy and its baseline. First, we do not include heat rate improvements, which will lower our reductions. Second, we do not include renewable energy targets in our baseline, which would increase our baseline emissions and reductions.
Our use of an economy-wide CGE model will also affect our emissions reductions and policy costs compared to the EPA's analysis. Our approach captures indirect changes to non-energy sector emissions, which are minor for SO_2 and NO_X but are appreciable for CO and VOC. Our costs also include these general equilibrium effects, which previous studies show can increase cost estimates (Goulder and Williams 2003). Our initial estimates of the policy cost are higher than $25 per capita in 2006$, though these are sensitive to the representation of renewables. These initial estimates are higher than the EPA's estimated median compliance cost in 2020, which is less than $20 per capita in 2011$ (U.S. Environmental Protection Agency 2014a).

While our methods offer advantages in representing economy wide costs and emissions changes, our analysis has multiple limitations. While USREP has a detailed electricity sector, it still relies on simplified production and cannot model capacity expansion or account explicitly for

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renewable intermittency or energy efficiency measures. The general equilibrium approach is meant to capture large effects, and is not computationally well suited to small markets, as in small U.S. states. This CGE model does not include pollution abatement options, which could allow it to respond to future air quality policy and represent the choice between controlling pollutant emissions and carbon dioxide (Nam et al. 2013). This limitation allows pollution emissions to increase beyond levels than would be predicted if the model incorporated the option to increase pollution control in response to air quality regulations. We are also limited in our ability to predict regional compliance patterns. National and State can be seen as bracketing the potential implementations of these state-based limits. There are many potential regional compliance patterns, which will depend on factors like incentives and practical barriers. What our results suggest are that regional compliance could offer cost savings without many pollutant emissions changes, while National represents the least change from the status quo.

While we have compared costs and emissions changes, a future extension of this work will estimate health-related co-benefits from emissions reductions and compare them to costs. Previous studies estimate co-benefits directly from emissions with linearized relationships between emissions and impacts. These studies are based on simplified atmospheric models (Muller and Mendelsohn 2009), surface response models (Fann, Fulcher, and Baker 2013; Fann, Baker, and Fulcher 2012), adjoint methods (Mesbah, Hakami, and Schott 2013), or reduced form models based on regional chemical transport models (Buonocore et al. 2014; Fann, Fulcher, and Hubbell 2009). Because damages from air pollution vary with timing, source, and location (Fann, Fulcher, and Hubbell 2009), these linearized estimates also vary by location and source, and may lose their accuracy over time, as atmospheric conditions (and sources) change (Holt, Selin, and Solomon 2015). To capture the effects of changing regional atmospheric chemistry on ground-level pollutant concentrations, the approach accepted by the U.S. regulatory community is to use regional chemical transport models, including CMAQ and CAMx (U.S. Environmental Protection Agency 2007). Future work could use advanced air quality modeling as in Thompson et al. (2014) to compare the co-benefits of these emissions changes to the costs saved from a regional or national implementation.

CONCLUSION

We link a CGE model of the United States with state-level detail to an advanced pollutant emissions model. We apply it to study the effect of an electricity sector carbon policy on economy-wide pollutant emissions changes. Specifically, we examine the effect of state, regional, or national compliance on pollutant emissions. We estimate that imposing state-level carbon intensity limits on the U.S. electricity sector will reduce fossil fuel use in electricity, reduce fossil-based electricity sector output, impose a policy cost, and reduce pollutant emissions. The largest emissions reductions are to SO_2, NO_X, CO, and $PM_{2.5}$ from fossil-based electricity generation; however up to 15% of primary $PM_{2.5}$ and up to 68% of CO reductions are from non-electricity sources. Compared to state-level implementation, a more flexible national or regional implementation is found to lower costs, increase economic output, and increase pollutant emissions. A flexible implementation may increase net co-benefits, as the cost savings are larger than the emissions changes on a percentage basis; however, future work is needed to derive co-benefits from emissions changes.

Second Essay

2. A Self-Consistent Method to Assess Air Quality Co-Benefits from U.S. Climate Policies

ABSTRACT[1]

Air quality co-benefits can potentially reduce the costs of greenhouse gas mitigation. However, while many studies of the cost of greenhouse gas mitigation model the economy-wide welfare impacts of mitigation, most studies of air quality co-benefits do not. We employ a US computable general equilibrium economic model previously linked to an air quality modeling system, and enhance it to represent the economy-wide welfare impacts of fine particulate matter. We present a first application of this method to explore the efficiency and distributional implications of a Clean Energy Standard (CES) and a Cap and Trade (CAT) program that both reduce CO_2 emissions by 10% in 2030 relative to 2006. We find that co-benefits from fine particulate matter reduction (median \$6; \$2 to \$10/tCO_2) completely offset policy costs by 110% (40% to 190%), transforming the net welfare impact of the CAT into a gain of \$1 (-\$5 to \$7) billion 2005\$. For the CES, the corresponding co-benefit (median \$8; \$3 to \$14)/tCO_2 is a smaller fraction (median 5%; 2% to 9%) of its higher policy cost. The eastern US garners 78% and 71% of co-benefits for the CES and CAT, respectively. By representing the effects of pollution-related morbidities and mortalities as an impact to labor and the demand for health services, we find that the welfare impact per unit of reduced pollution varies by region. These interregional differences can enhance the preference of some regions, like Texas, for a CAT over a CES, or switch the calculation of which policy yields higher co-benefits, compared to an approach that uses one valuation for all regions. This framework could be applied to quantify consistent air quality impacts of other pricing instruments, sub-national trading programs, or green tax swaps.

INTRODUCTION

Policies for cutting CO_2 emissions to mitigate climate change can improve regional air quality by incidentally reducing polluting activities. These air quality improvements can have welfare co-benefits (or ancillary benefits) that help to compensate for the cost of carbon policies. A growing body of literature has quantified the air quality co-benefits of carbon policy, in part to help identify policies that benefit air quality and climate simultaneously. However, studies of air

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quality co-benefits often use different methods to assess costs and benefits, precluding direct cost-benefit comparisons. Here, we develop a consistent approach to assess costs and economy-wide air quality co-benefits, by extending and applying an economic model developed to estimate emissions changes and policy costs of US climate policies. Specifically, we implement an approach to model and quantify the economy-wide welfare implications of air pollution reductions, and compare these air quality impacts to the costs of two US carbon policies.

There is mounting evidence that air quality co-benefits significantly offset the costs of greenhouse gas mitigation (Muller 2012; Jack and Kinney 2010; Ravishankara, Dawson, and Winner 2012). Nemet et al. (2010) summarize 37 peer-reviewed studies that estimate the air quality co-benefits of climate change policy, with results ranging from $2-147/tCO$_2$. Many assessments of the co-benefits of climate policies use partial equilibrium or computable general equilibrium (CGE) economic models to estimate the costs of climate policies (Nemet, Holloway, and Meier 2010; Bell, Morgenstern, and Harrington 2011; Burtraw et al. 2003). CGE models use general equilibrium theory to assess the long-run dynamics of resource allocation and income distribution in market economies. They have been widely applied since the early 1990s to evaluate the efficiency of environmental and energy policy (Bergman 2005), including studies to estimate the cost of the US Clean Air Act and the Acid Rain Trading program (Hazilla and Kopp 1990; Goulder, Parry, and Burtraw 1997). By simulating the entire economy, CGE models offer the advantage of estimating changes to emissions from all sectors, including non-regulated sectors that respond to changing prices (Scheraga and Leary 1994). Accounting for relative price changes throughout the economy is particularly important when projecting substantial climate or energy policy (Sue Wing 2009; Bhattacharyya 1996).

In contrast to the well-developed literature on the macroeconomic costs of climate policies, studies estimating air quality benefits use more varied methodologies, most of which do not capture macroeconomic effects. Early studies of the air quality co-benefits of climate policy quantified benefits by applying linearized $/ton estimates of avoided damages from pollutant emissions (Goulder 1993; Scheraga and Leary 1994; Boyd, Krutilla, and Viscusi 1995). Later studies applied more detailed emissions-impact relationships, including information from source-receptor atmospheric modeling and updated information on concentration-response functions and

associated costs (Burtraw et al. 2003; Holmes, Keinath, and Sussman 1995; Dowlatabadi 1993; Rowe and et al. 1995). Health damages are most often valued using estimates of the willingness to pay (WTP) for reduced health risks (Bell, Morgenstern, and Harrington 2011). WTP estimates for reduced mortality risk, termed Value of a Statistical Life (VSL), comprise the majority of these benefits estimates (OIRA 2013). Macroeconomic CGE analysis attempts to incorporate and build on this approach by including the constraints of multiple policies, limited resources, and changing prices, which can lead to significant indirect gains or losses (Smith and Carbone 2007). Since top-down economic modeling approaches like CGE are commonly applied to estimate the costs of climate policy (Paltsev and Capros 2013), a consistent assessment of the air quality co-benefits would use a similar approach to capture indirect gains as well as indirect losses.

A growing number of studies have used CGE models to estimate the macroeconomic and welfare impacts of air pollution. These studies link the human health impacts from fine particulate matter and ozone to welfare loss through increased medical expenses, lost wages, pain and suffering, and reductions in the supply of labor. CGE models have been used to evaluate benefits from the US Clean Air Act (CAA) from 1975 to 2000 (Matus et al. 2008), global ozone impacts under future climate and mitigation scenarios (Selin et al. 2009), and the historical burden of air pollution in Europe (Nam et al. 2010), and China (Matus et al. 2012); however, none of these studies assessed policy costs. The US EPA's Second Prospective Analysis of the CAA evaluated both human health benefits and costs using a CGE framework, but it used pollution changes generated outside the CGE model (U.S. Environmental Protection Agency 2011b). The studies discussed above have used CGE models either to estimate the cost of environmental policy, or the benefits of air pollution reductions, but not both.

Here, we present a method for the consistent evaluation of costs and co-benefits of carbon policies. We present an approach to quantify the economy-wide welfare impacts of air pollution reductions in the same macroeconomic model used to assess emissions changes and policy costs. We adapt a multi-region, multi-sector, multi-household CGE model of the US economy, the US Regional Energy Policy Model (USREP). This model was previously linked with a detailed emissions inventory and air quality modeling system (Thompson et al. 2014) to estimate policy costs, emissions changes, pollution changes, and human health impacts. We present a first

application of this method to a national Clean Energy Standard and an equivalent Cap and Trade program. We compare the economy-wide labor and health impacts from fine particulate matter reductions that arise incidentally from each policy. We explore how these co-benefits affect both the efficiency (by reducing policy costs) and the distributional implications of each policy, by modeling how net co-benefits of a national policy are distributed across the continental US. With our more consistent co-benefits, we re-examine the question: can the impacts of air quality co-benefits on economic resources "pay for" a climate or clean energy policy in the US?

INTEGRATED ASSESSMENT METHODS

We use an integrated assessment framework to model policies, emissions, and impacts, shown in Figure 2-1. The United States Regional Energy Policy (USREP) economic model (Rausch et al. 2010) is used at the beginning and the end of our analysis process. At the beginning, USREP is used to implement climate policies, quantify costs, and estimate emissions changes (Rausch et al., 2011). The Comprehensive Air Quality Model with Extensions (CAMx) (Environ International Corporation 2013) is next used to link emissions to atmospheric concentrations. The Environmental Benefits Mapping and Analysis Program (BenMAP) (Abt Associates, Inc. 2012) is used to estimate human health impacts related to fine particulate matter. As the methodological contribution of this paper, we add a final step to the analysis by using BenMAP-derived health impacts to derive estimates of economy-wide co-benefits in USREP.

This section presents the USREP model, its link to our air quality modeling system and health impacts assessment, our extension of the USREP model to include economy-wide air quality co-benefits, and our application of this new integrated approach to a US national Clean Energy Standard and Cap and Trade program.
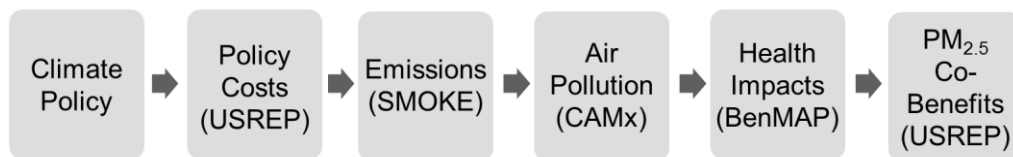
Figure 2-1: Integrated assessment framework for estimating air quality co-benefits of US climate policy.
This framework implements policies in the economic model (USREP), then estimates the impacts to welfare, production, and emissions. Emissions in SMOKE are input to the air quality model CAMx to yield concentrations of fine particulate matter. Those concentrations are input to BenMAP to estimate human health impacts. Lastly, those health impacts are input to USREP to estimate the welfare impacts of fine particulate matter pollution.

The US Regional Energy Policy Model

We use USREP to analyze two US-wide carbon policies and to estimate their costs, their effects on pollutant emissions, and their respective air quality co-benefits. USREP is a multi-region, multi-commodity, multi-household recursive-dynamic computable general equilibrium (CGE) model of the US economy, whose input and structure are described in detail in Rausch et al. (2010) and Rausch et al. (2011). USREP calculates commodity prices that support equilibrium between supply and demand in all markets based on the microeconomic decisions of rational agents (i.e., it is a classical Arrow-Debreu model). By including rich detail in the energy sector, and by relating production to emissions of greenhouse gases, USREP is designed to explore the long-run dynamics and the economy-wide costs, emissions impacts, and distributional implications of both national and sub-national energy and climate change policies. USREP has been applied previously to study climate change and energy policies, exploring their effects on economic growth, efficiency, distribution, and interactions with existing distortionary taxes (Rausch et al. 2010; Rausch, Metcalf, and Reilly 2011; Caron, Rausch, and Winchester 2015; Rausch and Mowers 2014).

USREP assesses equilibrium conditions over 5-year periods among profit-maximizing firms and utility-maximizing consumers that receive income from supplying four factors of production (labor, capital, land, and resources). USREP is a full employment model, with the labor supply determined by the household choice between labor and leisure. Taxes are collected by the government and spent on consumption and transfers to households (Rausch et al., 2010).

As shown in Table 2-1, USREP contains 12 geographic regions, 9 household income groups, 5 energy commodities, 5 non-energy sectors and advanced "backstop" energy technologies (e.g., advanced solar energy is a "backstop" for fossil energy as it can produce a close substitute for this non-renewable resource). Production is characterized by nested constant-elasticity-of-substitution (CES) functions, the details of which are in Rausch et al. (2010). The geographic regions include California, Texas, and Florida, and several multi-state composites, shown in Figure 2-2.

Table 2-1: USREP Model Details. Regional, Household, and Sectoral Breakdown and Primary Input Factors

Sectors	Regions	Primary production factors
Non-Energy	Pacific (PACIF)	Capital
Agriculture (AGR)	California (CA)	Labor
Services (SRV)	Alaska (AK)	Coal resources
Energy-intensive products (EIS)	Mountain (MOUNT)	Natural gas resources
Other industries products (OTH)	North Central (NCENT)	Crude oil resources
Commercial transportation (TRN)	Texas (TX)	Hydro resources
Passenger vehicle transportation (TRN)	South Central (SCENT)	Nuclear resources
Final demand sectors	North East (NEAS)	Land
Household demand	South East (SEAST)	Wind
Government demand	Florida (FL)	
Investment demand	New York (NY)	Household income classes
Energy	New England (NENGL)	($1,000 of annual income)
Coal (COL)		<10
Natural gas (GAS)		10-15
Crude oil (CRU)		15-25
Refined oil (OIL)		25-30
Electric: Fossil (ELE)		30-50
Electric: Nuclear (NUC)		50-75
Electric: Hydro (HYD)		75-100
Advanced Technologies		100-150
		>150

Figure 2-2: Regions of USREP.
They are the aggregation of the following states: NEW ENGLAND = Maine, New Hampshire, Vermont, Massachusetts, Connecticut, Rhode Island; SOUTH EAST = Virginia, Kentucky, North Carolina, Tennessee, South Carolina, Georgia, Alabama, Mississippi; NORTH EAST = West Virginia, Delaware, Maryland, Wisconsin, Illinois, Michigan, Indiana, Ohio, Pennsylvania, New Jersey, District of Columbia; SOUTH CENTRAL = Oklahoma, Arkansas, Louisiana; NORTH CENTRAL = Missouri, North Dakota, South Dakota, Nebraska, Kansas, Minnesota, Iowa; MOUNTAIN = Montana, Idaho, Wyoming, Nevada, Utah, Colorado, Arizona, New Mexico; PACIFIC = Oregon, Washington, Hawaii.

The USREP model is based on 2006 state-level economic data from the Impact analysis for PLANning (IMPLAN) dataset and energy data from the Energy Information Administration's State Energy Data System (SEDS). The energy supply is regionalized with data on regional fossil fuel reserves from the US Geological Service and the Department of Energy. Further details on the input data are contained in Rausch and Mowers (2014). Several studies examine the effect of varying model inputs and structure, like the source of household income data (Rausch et al., 2011), and the structure of the energy system model (Rausch and Mowers 2014; Lanz and Rausch 2011). Paltsev and Capros (2013) list numerous studies that have explored the effects on climate mitigation costs of assumptions about innovation, low-carbon technologies, flexibility to substitute energy to low-carbon options, other regulations and regulatory credibility to trigger long-term investment, timing of actions, and the reference scenario.

Linking USREP to Emissions and Health Outcomes

Economic activity was linked to emissions, concentrations, and health outcomes by coupling USREP to an air quality modeling system and health impacts model. The details of this

approach, including projected pollutant emissions and concentrations under selected carbon policies, are described in full by Thompson et al. (2014) and summarized below.

Emissions to Concentrations

USREP was linked to an air quality modeling system with a national emissions inventory for 2005. The inventory was speciated and temporally processed using Spare Matrix Operating Kernel Emissions (SMOKE) (CMAS 2010). Production in USREP was linked to the relevant emissions sources in the detailed emissions inventory by mapping each USREP sector and region to its corresponding sources in the emissions inventory. To estimate future emissions in 2030, the relative change in production for each sector and region in USREP was used to scale the corresponding sources in the 2005 emissions inventory. For example, an increase in electricity generation from natural gas in USREP caused a proportional increase in all relevant pollutant emissions associated with the production, transportation, and use of natural gas for electricity. All relevant anthropogenic emission sources – including point sources and area sources – were scaled and run through SMOKE to produce gridded, hourly emissions for each scenario.

Projected 2030 emissions were used to estimate future hourly fine particulate matter concentrations on a 36 km grid of the continental US using the Comprehensive Air Quality Model with Extensions (CAMx) version 5.3 (Environ International Corporation 2013). To isolate the effects of policy efforts on the emissions of fine particulate matter and its precursors, we did not incorporate climate change in our analysis; emissions are expected to exceed the effect of climate change on US fine particulate matter in 2030 (Penrod et al. 2014). Instead, meteorological input for the year 2005 was used for both present and future simulations, and was developed with the fifth generation Penn State/NCAR mesoscale model MM5 (Grell, Dudhia, and Stauffer 1994). CAMx has been used in numerous evaluations of US air quality policy (US EPA 2011; US EPA 2012). The year-long air quality modeling episode for 2005 that we use as our base year was developed as part of a base case to evaluate the proposed Cross-State Air Pollution Rule, which was documented and evaluated in US EPA (2011).

Concentrations to Health Ou<span id="zotero-drag"/>tcomes

We calculate mortality and morbidity resulting from fine particulate matter concentrations using the Environmental Benefits Mapping and Analysis Program (BenMAP) v4.0. Previous studies using this air pollution episode analyzed benefits of both ozone and fine particulate matter reductions (Thompson and Selin 2012; Thompson, Saari, and Selin 2014; Thompson et al. 2014). Here, we focus on fine particulate matter, estimating both morbidity and mortality following the methods used in the Regulatory Impact Assessment (RIA) for the fine particulate matter National Ambient Air Quality Standard (US EPA, 2012). Table 2-2 lists the endpoints and concentration-response functions applied. Following the RIA's approach, we have estimated the lower and upper bound of the number of health impacts based on both the selection of health impact functions, and the uncertainty in those functions, as specified in Table 2-2.

Table 2-2. Endpoints, epidemiologic studies, and valuations used for fine particulate matter health impacts, following US EPA (2012)

Endpoint / Endpoint Group		Ages (yrs)	Individual Studies	Pooling and Lower/Upper Bounds	Valuation $2006USD
Premature Mortality		>30	Krewski et al. (2009) and Lepeule et al. (2012)	**Upper bound:** Lepeule (2012) 95th percentile **Lower bound:** Krewski et al (2009) 5th percentile	N/A
Nonfatal Myocardial Infarction			Peters et al. (2001). Pope et al. (2006), Sullivan et al. (2005), Zanobetti et al. (2009), Zanobetti and Schwartz (2006)	**Upper bound:** Peters et al. (2001) 95th percentile **Lower bound:** 5th percentile of equal-weights pooling of 4 other studies	$100,000
Respiratory Hospital Admissions	All	>64	Zanobetti et al. (2009)—ICD 460-519 (All respiratory) Kloog et al. (2012)—ICD 460-519 (All respiratory)	**Pooling of:** **All Respiratory:** Pooling of Zanobetti et al. (2009) and Kloog et al. (2012)	$23,711
	COPD	18–64	Moolgavkar (2000)—ICD 490–492, 494-496 (COPD, less asthma)	**Asthma:** Pooling Babin et al. (2007) and Sheppard (2003)	$15,903
	Asthma	<18	Babin et al. (2007)—ICD 493 (asthma), Sheppard (2003)—ICD 493 (asthma)	**COPD (less asthma):** Moolgavkar (2000)	$10,040

Endpoint / Endpoint Group	Ages (yrs)	Individual Studies	Pooling and Lower/Upper Bounds	Valuation $2006USD
Cardiovascular Hospital Admissions	>64	Zanobetti et al. (2009)—ICD 390-459 (all cardiovascular) Peng et al. (2009)—ICD 426-427; 428; 430-438; 410-414; 429; 440-449 (Cardio-, cerebro- and peripheral vascular disease) Peng et al. (2008)—ICD 426-427; 428; 430-438; 410-414; 429; 440-449 (Cardio-, cerebro- and peripheral vascular disease) Bell et al. (2008)—ICD 426-427; 428; 430-438; 410-414; 429; 440-449 (Cardio-, cerebro- and peripheral vascular disease)	Pooling of all 4 studies for ages > 64 added to Moolgavkar (2000) for ages 18-64	$27,319
	18–64	Moolgavkar (2000)—ICD 390–429 (all cardiovascular)		$29,364
Asthma-related ER Visits	<18	Mar et al. (2010), Slaughter et al. (2005), Glad et al. (2012)	Pooling of all 3 studies	$370
Acute bronchitis	8–12	Dockery et al. (1996)	N/A	$416
Lower Respiratory Symptoms	7–14	Schwartz and Neas (2000)	N/A	$18
Upper Respiratory Symptoms	9–11	Pope et al. (1991)	N/A	$29
Asthma Exacerbation	6–18	Ostro et al. (2001) (cough, wheeze, shortness of breath), Mar et al. (2004) (cough, shortness of breath)	Pooling of Ostro et al. (2001) and Mar et al. (2004)	$50
Minor Restricted-Activity Days	18–64	Ostro and Rothschild (1989)	N/A	$60
Work Loss Days	18–64	Ostro (1987)	N/A	$150

ER = Emergency Room
ICD = International Statistical Classification of Disease
COPD = Chronic Obstructive Pulmonary Disease

Economic Modeling of Air Quality Co-Benefits

For this study, we incorporate the economic and welfare effects of pollution-related health outcomes into USREP, accounting for morbidities and mortalities with separate techniques. Our focus is on the change in consumer welfare, which can be generally understood as the income amount that would be necessary to compensate consumers for losses under a policy. Our welfare index includes the change in macroeconomic consumption (capturing market-based activities), and the change in leisure (i.e., the monetary value of the change in non-working time) in response to policy (Paltsev and Capros 2013). We account for morbidities through lost wages, lost leisure, and medical expenses that vary with pollution levels. We account for mortality by reducing the supply of labor accordingly.

Morbidities in USREP

We account for morbidities related to fine particulate matter by representing the change in medical expenditures and lost wages through a new sector added to USREP. We add a household production sector for "pollution health services" whose production is determined by the pollution level and the valuation of the resulting health outcomes. The valuation of each morbidity endpoint is shown in Table 2-2, following US EPA (2012). These valuations are derived from estimates of willingness-to-pay (i.e., for asthma exacerbation and upper respiratory symptoms), medical costs, and lost wages, using US data. Similar approaches incorporate willingness-to-pay in CGE models to estimate air quality impacts, representing non-market losses as lost leisure (Matus et al. 2008; Selin et al. 2009; Nam et al. 2010; Matus et al. 2012). Smith and Carbone (2008) discuss the theoretically preferred approach and remaining empirical challenges to incorporating air quality preferences in CGE models. We follow US EPA (2012) as these valuations are based on US data and studies, and have been applied in evaluating US air quality regulations (U.S. Environmental Protection Agency 2012; U.S. Environmental Protection Agency 2011b). Within USREP, we apply the valuation per case to the number of cases estimated using BenMAP, and use the total valuation to calculate the demand for resources for the new "pollution health services" sector.

The "pollution health services" sector tracks the demand for economic resources in response to pollution-related health outcomes. Higher pollution reduces welfare by requiring more resources per health outcome. Each outcome creates economic impacts comprised of medical costs, lost labor, and other disutility (e.g., pain and suffering). We map these impacts to demand for sectoral inputs of services and labor using functions for each endpoint developed by Yang (2004) and Matus et al. (2008). The fine particulate matter pollution levels affect the output of the pollution health services sector, with higher pollution drawing more resources per unit of output (termed a Hicks neutral negative technical change). Policies that reduce pollution increase welfare as lower pollution increases the productivity of this sector.

We add this new sector to those listed in Table 2-1, following Matus et al. (2008) and Nam et al. (2010). It requires inputs of service, which are drawn from the services ("SRV") sector, and of labor, which are drawn from the household labor supply. The output of this new sector is

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included in private consumption. It thus forms a component of household welfare, i.e., the sum of consumption and leisure.

Mortalities in USREP

We do not value pollution-related mortalities directly (e.g., with a VSL estimate), but instead estimate how they affect welfare by reducing the supply of labor. Higher pollution-related mortality reduces welfare in a region by reducing the supply of labor thereby increasing production costs and decreasing consumption. We first estimate the change in the adult (> 30 years) population by dynamically reducing the census-based population projections in USREP by the amount of pollution-related mortalities from BenMAP. We apply a 2/3 labor participation rate of adults (i.e., the employment-to-population ratio) to estimate the percent change in the labor force from the percent change in population, as in Matus et al. (2008). We apply the change in labor force to the year in which the death took place, effectively accounting for one year of life lost per death. Estimating the actual years of life lost would increase our estimates of the labor impact of mortality. By reducing the labor supply, we affect wage rates, which in turn affect workers' decisions on how to use their total time endowment, represented in USREP as a substitution between labor and leisure (Rausch et al. 2010).

Climate Change Policies

Clean Energy Standard and Equivalent Cap and Trade Program

We apply our modeling framework to estimate economy-wide co-benefits from fine particulate matter reductions under two national climate policies, previously implemented in USREP (Rausch and Mowers 2014). These policies' air quality implications and estimated co-benefits were previously assessed by applying VSL measures to mortalities from fine particulate matter and ozone (Thompson et al. 2014).

Our first policy is a Clean Energy Standard (CES) similar to the proposed Clean Energy Standard Act of 2012 (Bingaman et al. 2012). This policy doubles clean energy from 42% to 80% by 2035, beginning in 2012, by setting specified percentages of electricity sales from qualified energy sources. The second policy is an equivalent US economy-wide Cap and Trade program (CAT). The revenue from auctioned emissions permits for the CAT is returned lump-sum to households on a per capita basis (Rausch and Mowers 2014). Both policies reduce

equivalent CO_2 emissions, i.e., 500 million metric tons CO_2, or a 10% reduction in 2030 relative to 2006 emissions. Both the CAT and CES are compared to a Business As Usual (BAU) scenario in which CO_2 emissions grow to 6,200 mmt by 2030. We analyze these policies as they were implemented in USREP by Thompson et al. (2014). We estimate the costs of each policy as the cumulative, undiscounted change in welfare (i.e., material consumption and leisure) for all regions from 2006-2030 compared to BAU.

Estimating Welfare Impacts of Policies from Fine Particulate Matter Reductions over Time
To estimate co-benefits from fine particulate matter reductions, we first average our CAMx output to the temporal and spatial scales of USREP. From CAMx we obtain daily concentrations in 2005 as well as in 2030 for our BAU, CAT, and CES scenarios. We combine those concentrations with census data to obtain population-weighted annual average concentrations for each USREP region contained in our air quality modeling grid. Because we model air quality in the continental US only, we do not estimate co-benefits for Alaska and Hawaii.

To capture cumulative impacts as reductions are gradually realized, we interpolate pollution reductions between 2005 and 2030 and their health effects over our analysis period of 2006-2030. For morbidities, that pollution change (in every 5-year period and region) becomes the (Hicks neutral) negative scaling factor that affects the productivity of the health services sector. Under BAU, we assume population-weighted pollution levels follow a linear progression from 2006 to 2030 levels. Our policies begin implementation in 2012. We assume pollution levels follow BAU from 2006-2012, and then assume a linear implementation of the remaining reductions from 2012-2030 for each policy. For mortalities, we estimate total deaths from the change in fine particulate matter between 2006 and 2030 in BenMAP for BAU and both policies. We then follow the same interpolation process for mortalities as for the pollution levels to allocate those deaths over the periods in BenMAP between 2006 and 2030. Over time, increases in labor productivity and population size and age each serve to increase the value of a policy's pollution reductions compared to BAU.

To estimate economy-wide co-benefits, we run USREP six more times from 2006-2030. We run USREP twice for each of the three scenarios, BAU, CES, and CAT, using the corresponding pollution levels and the lower and upper bounds of the health effects estimates, respectively. The

upper and lower bounds are determined following US EPA (2012), and are based on the 95% confidence intervals of individual or pooled studies in combination with the selection of different studies to assess the lower and upper bounds of outcomes for each endpoint. We estimate the co-benefits of each policy as the cumulative, undiscounted change in welfare (i.e., material consumption and leisure) compared to BAU. This change in welfare is our estimate of each policy's air quality co-benefit.

Integrated Assessment Process: Policy Costs to Air Quality Co-Benefits in one Framework
To summarize this process, depicted in Figure 2-1, results from USREP are used to estimate the policy costs and economic activity under BAU, CAT, and CES, respectively. Those economic activities were mapped to emissions of particulate matter and its precursors in 2030 by scaling a detailed emissions inventory for 2005 in SMOKE, and fine particulate matter concentrations in 2005 and 2030 were estimated by CAMx (Thompson et al. 2014). Based on these previous results, we create upper and lower bound estimates of morbidities and mortalities over time with BenMAP. We then run USREP again to estimate the lower and upper bounds of air quality welfare impacts for each of BAU, CES, and CAT. Thus, we use the economy-wide impacts of complying with each policy to model future pollutant concentrations and their economy-wide impacts due to human health responses. We estimate net co-benefits by subtracting air quality co-benefits from the policy cost.

CO-BENEFITS FROM FINE PARTICULATE MATTER REDUCTIONS

We present estimates of air quality co-benefits for each policy, on a total, per capita, and per ton of CO_2 basis. We then sum the welfare impacts of pollution and policy implementation to calculate net co-benefits. We compare co-benefits to costs to estimate the fraction by which co-benefits reduce policy costs. We present results at the national scale followed by the regional scale. Finally, we explore how general equilibrium and cumulative effects contribute to our results, both in terms of policy efficiency (i.e., net co-benefits) and distributional implications. Intermediate results describing changes in emissions, mortalities, and the value of health outcomes are provided in the supplemental information.

National Air Quality Co-Benefits and Net Co-Benefits by Policy

National Co-Benefits

Co-benefits from fine particulate matter reductions for each policy compared to BAU are presented in Table 2-3. We show the median and the upper and lower bounds derived from the uncertainty in the health estimates, as in Table 2-2. All values are denominated in constant year 2005 US dollars (i.e., 2005$). Nationally, we estimate that the CES yields higher co-benefits ($13 billion, with a range $4 to $21 billion) than the CAT program ($9 billion, range $3 to $15 billion), measured as cumulative benefits by 2030 in 2005$. This reflects the greater fine particulate matter reductions under the CES than under the CAT (reduction in population-weighted annual average daily mean of 0.97 µg m^{-3} for CES, compared to 0.56 µg m^{-3} for CAT) (Thompson et al. 2014). These co-benefits correspond to $8 ($3 to $14)/tCO$_2$ for the CES, and $6 ($2 to $10)/tCO$_2$ for the CAT.

National Net Co-Benefits

We compare the air quality co-benefits of each policy to its respective cost, reported as the cumulative, undiscounted change in welfare (i.e., material consumption and leisure) compared to BAU, shown in Table 2-4. We calculate net co-benefits as the sum of the modeled welfare changes due to fine particulate matter reductions (always positive) and due to policy implementation (usually negative). Co-benefits are the change in welfare from fine particulate matter changes. Policy costs are the change in welfare from policy implementation; policies can impart welfare gains to some regions, which we term a "positive cost". Positive net co-benefits indicate a net welfare gain when air quality co-benefits are included in policy costs. The CES is the more expensive policy, costing $242 billion (2005$) compared to $8 billion (2005$) for a CAT that reduces the same amount of CO$_2$.

Table 2-3: Median co-benefits of each policy (total, per capita, and per ton of mitigated CO_2 emissions) [range in square brackets].

The CES has higher co-benefits than the CAT, nationally and across regions. Co-benefits from fine particulate matter reductions by each policy are expressed in terms of: (1) the cumulative, undiscounted change in welfare (i.e., material consumption and leisure) compared to BAU from 2006-2030 (million 2005$); (2) the equivalent welfare change per capita (2005$); (3) the welfare change per ton of CO_2 mitigated. The median appears along with the range estimated from the uncertainty in health estimates in square brackets. Negative values are in round brackets. West and East are divided by the Mississippi River, and their values are the sums of their respective regions.

Region	Co-Benefits (Million 2005$)		Co-Benefits Per Capita (2005$)		Co-Benefits Per Ton of CO_2 mitigated ($2005/tCO_2$)	
	CES	CAT	CES	CAT	CES	CAT
National	$12,800 [$4,500, $21,000]	$8,700 [$2,800, $14,600]	$35 [$12, $57]	$24 [$8, $40]	$8 [$3, $14]	$6 [$2, $10]
West	$2,800 [$900, $4,600]	$2,500 [$630, $4,370]	$100 [$40, $150]	$81 [$22, $140]	$4.1 [$1.2, $7.0]	$29 [$7, $51]
East	$10,000 [$3,500, $17,000]	$6,200 [$2,200, $10,200]	$260 [$90, $430]	$190 [$70, $320]	$140 [$40, $240]	$40 [$10, $60]
Pacific	$81 [$81, $81]	$55 [$19, $90]	$5 [$5, $5]	$4 [$1, $6]	$2 [$1, $4]	$4 [$1, $6]
California	$270 [$80, $450]	$440 [$70, $810]	$6 [$2, $10]	$9 [$2, $17]	$4 [$1, $6]	$10 [$2, $19]
Mountain	$200 [$60, $340]	$260 [$40, $490]	$7 [$2, $11]	$9 [$1, $16]	$1 [$0, $2]	$1.5 [$0.2, $2.7]
North Central	$980 [$400, $1,600]	$460 [$190, $730]	$45 [$18, $71]	$21 [$9, $33]	$6* [$2, $9]	$4 [$1, $6]
Texas	$1,100 [$300, $1,800]	$1,280 [$310, $2,250]	$32 [$10, $55]	$38 [$9, $68]	$4 [$1, $7]	$10 [$2, $18]
North East	$3,900 [$1,400, $6,400]	$2,200 [$800, $3,500]	$48 [$17, $79]	$26 [$9, $44]	$11 [$4, $19]	$9 [$3, $14]
New England	$560 [$140, $980]	$1,000 [$400, $1,700]	$36 [$9, $63]	$66 [$24, $109]	$98 [$24, $170]	$7 [$2, $11]
New York	$680 [$240, $1,100]	$660 [$260, $1,050]	$35 [$12, $58]	$34 [$13, $54]	$10 [$3, $16]	$7 [$3, $11]
South Central	$660 [$290, $1,000]	$280 [$100, $470]	$55 [$24, $87]	$24 [$8, $39]	$6 [$2, $9]	$3 [$1, $5]
South East	$3,400 [$1,100, $5,600]	$1,700 [$600, $2,700]	$57 [$19, $95]	$28 [$10, $46]	$8 [$3, $14]	$7 [$3, $12]
Florida	$840 [$340, $1,300]	$440 [$120, $750]	$29 [$12, $47]	$15 [$4, $26]	$7 [$3, $12]	$2 [$1, $4]

*In the North Central region, CO_2 emissions actually rise slightly (by 5%) under the CES due to increased coal and gas use in the electricity sector.

Table 2-4: Co-benefits, costs, and net co-benefits in billions 2005$.

Region	Co-Benefit (billion 2005$)		Cost (billion 2005$)		Net Co-Benefit (billion 2005$)	
	CES	CAT	CES	CAT	CES	CAT
National	$13 [$5, $21]	$9 [$3, $15]	($242)	($8)	($229) [($237), ($221)]	$1 [($5), $7]
West	$2.8 [$0.9, $4.6]	$3 [$0.6, $4]	($92)	($8.2)	($89) [($91), ($87)]	($5.7) [($7.6), ($3.8)]
East	$10 [$3.5, $16.5]	$6 [$2, $10]	($150)	$0.4	($140) [($146), ($133)]	$6.7 [$2.7, $10.7]
Pacific	$0.25 [$0.08, $0.41]	$0.05 [$0.02, $0.09]	$3.2	$0.5	$3 [$3, $4]	$0.6 [$0.5, $0.6]
California	$0.27 [$0.08, $0.45]	$0.44 [$0.07, $0.09]	($25)	$1.4	($24) [($25), ($24)]	$1.8 [$1.4, $2.2]
Mountain	$0.20 [$0.06, $0.34]	$0.26 [$0.04, $0.49]	($17)	($2.5)	($17) [($17), ($17)]	($2.2) [($2.4), ($2.0)]
North Central	$0.98 [$0.40, $1.6]	$0.46 [$0.19, $0.73]	($22)	($2.2)	($21) [($21), ($20)]	($1.7) [($2.0), ($1.5)]
Texas	$1.1 [$0.3, $1.8]	$1.28 [$0.31, $2.25]	($31)	($5.4)	($30) [($31), ($29)]	($4.1) [($5.1), ($3.2)]
Northeast	$3.9 [$1.4, $6.4]	$2.2 [$0.8, $3.5]	($60)	($1.5)	($56) [($59), ($54)]	$0.7 [($0.7), $2.1]
New England	$0.6 [$0.1, $1.0]	$1.0 [$0.4, $1.7]	($6.5)	$2.5	($6.0) [($6.4), ($5.6)]	$3.5 [$2.8, $4.2]
New York	$0.7 [$0.2, $1.1]	$0.7 [$0.3, $1.1]	($6.9)	$0.3	($6.2) [($6.7), ($5.8)]	$1.0 [$0.6, $1.4]
South Central	$0.7 [$0.3, $1.0]	$0.28 [$0.10, $0.47]	($13)	($1.4)	($12) [($13), ($12)]	($1.1) [($1.3), ($0.9)]
Southeast	$3.4 [$1.1, $5.6]	$1.7 [$0.6, $2.7]	($37)	$0.2	($34) [($36), ($32)]	$1.9 [$0.8, $3.0]
Florida	$0.8 [$0.3, $1.3]	$0.4 [$0.1, $0.7]	($26)	$0.3	($25) [($26), ($25)]	$0.7 [$0.4, $1.1]

The CAT is more cost-effective than the CES, and its co-benefits can offset more of its costs, up to 190% (median 110%; 10% to 690%). (1) Co-benefits of each policy are the change in welfare (consumption and leisure) from BAU due to fine particulate matter reductions, expressed in billions 2005$. (2) Costs of each policy are the change (usually a negative change) in welfare (consumption and leisure) from BAU due to the policy implementation, expressed in billions 2005$. Regions that gain from policy implementation (i.e. costs are positive) are highlighted with double outlines. (3) Net co-benefits are the sum of these two welfare changes. Where net co-benefits are positive, cells are emphasized with double outlines. The median appears along with the range estimated from the uncertainty in health estimates. Negative values are in round brackets. West and East are divided by the Mississippi River, and their values are the sums of their respective regions.

We compare co-benefits to policy costs in two ways. We sum co-benefits and policy costs to estimate net co-benefits. Summing co-benefits and costs estimates the amount by which air quality impacts reduce the apparent mitigation costs. If net co-benefits are positive, then fine particulate matter co-benefits completely offset policy costs. Each policy has different costs and co-benefits (e.g., the CES has higher co-benefits and higher costs). Thus, to indicate the relative importance of air quality, we estimate what fraction of policy costs are offset by air quality co-benefits.

We find that the CES has net co-benefits of -\$230 (-\$237 to -\$221) billion 2005\$. The CAT has net co-benefits of \$1 (-\$5 to \$7) billion 2005\$. This implies that the policy costs of the CES are reduced by 5% (2% to 9%) by ancillary fine particulate matter reductions. For the CAT, up to 190% of the costs are offset by co-benefits (median 110%; 40% to 190%).

Regional Air Quality Co-Benefits and Net Co-Benefits by Policy

Regional Co-Benefits
Table 2-3 shows the distribution of the calculated co-benefits of each policy across the continental US (total, per capita, and per ton of mitigated CO_2 emissions). Most co-benefits go to regions east of the Mississippi River. For the CES, per capita gains are \$260 in 2005\$ in the East compared to \$95 in the West, with 78% of all gains accruing to eastern regions. For the CAT, 71% of all gains accrue to the East. The distribution of co-benefits is a combination of the pattern of pollution reductions and general equilibrium (GE) economic effects. Because fine particulate matter reductions are greatest in the eastern states under both policies (Thompson et al. 2014), most co-benefits accrue to those regions. We discuss GE effects in a later results section on the Contribution of General Equilibrium Economic Effects.

Some regions experience high or low co-benefits on a per ton basis. This pattern is the distributional effect of a national policy. To understand the effects of a regional policy, a separate analysis would be required. For example, under the CES, in one region – North Central – CO_2 emissions actually rise by 5%. Thus, the co-benefits per ton of CO_2 in North Central (median \$6/tCO_2; range \$2/tCO_2 to \$9/tCO_2 2005\$) are actually expressed with respect to an increase in CO_2 emissions. CO_2 emissions rise under the CES in North Central due to increased

coal and gas use in the electricity sector. This outcome is only possible because compliance with the CES is counted on a national basis and not for each region: if each region had to meet the clean energy standard on its own, this effect would not occur. Similarly, New England only reduces a small amount of CO_2 under the CES, as more cost-effective reductions are realized elsewhere. At the same time, New England still benefits from upwind pollutant reductions, and, consequently, appears to have high benefits per ton of $98/tCO_2$ ($24/tCO_2$ to $170/tCO_2$). As with North Central, if New England had to meet a regional CES of its own, its co-benefits per ton would likely drop as its required CO_2 reductions would rise. If New England unilaterally adopted a CES, its local pollutant emissions might decrease, but its transport of pollution from unconstrained upwind regions might rise compared to a national CES. Given these interregional interactions, the costs and co-benefits in a given region do not depend only on the local impacts of policy or pollution.

Regional Net Air Quality Co-Benefits

Table 2-4 displays the co-benefits, the costs, and the net co-benefits by policy and region. The costs are defined as the cumulative change in welfare (consumption and leisure) resulting from compliance with each policy, expressed in billions 2005$. Since "costs" are defined as a welfare change (usually negative), the regions that benefit from implementing these policies have a positive "cost". Those regions are highlighted in Table 2-4. Net co-benefits are the sum of the welfare changes due to fine particulate matter reductions (always positive) and due to policy implementation (usually negative). Regions with positive net co-benefits are also highlighted in Table 2-4.

Implementing each policy does benefit some regions, i.e. their policy "costs" are actually a welfare gain. For the CAT, this is true of the East which gains $0.4 billion. Individual regions that gain from the CAT are the coastal regions, excepting the North East, i.e., Pacific, California, New England, New York, South East, and Florida. As explained in Rausch and Mowers (2014), coastal regions appear to gain from the CAT because of how we treat its revenue, which we return to households on a per capita basis. Our per capita allocation of carbon revenue over-compensates people in these populous, largely de-carbonized areas (Rausch and Mowers 2014). For the CES, it is only the Pacific region that gains a relative advantage and reaps $0.5 billion 2005$ relative to BAU.

In terms of net co-benefits, the CAT favors the East while the CES favors the West. For the CAT, the East nets a gain of $6.7 ($2.7 to $11) billion 2005$. The West posts a net loss of –$5.7 (-$7.6 to -$3.8) billion 2005$. For the CES, the East and West post net losses of $140 ($133 to $146) and $89 ($87 to $91) billion 2005$, respectively. Under the CES, the East fares worse than West because its higher co-benefits ($10 billion in the East versus $3 billion in the West) are countered by even higher costs (-$150 billion in the East versus -$92 billion in the West).

For each policy, some regions receive a net gain. For the CAT, net co-benefits are positive in the coastal regions (which gain from policy implementation, i.e., have positive costs), including the North East (where costs are negative). Median net co-benefits in these regions range from $0.7 billion $2005 USD in the North East to $3.5 billion 2005$ in New England. For the CES, median net co-benefits are positive only in the Pacific, at $0.4 billion 2005$. Apart from the North East, the regions with positive net co-benefits are the regions that gain under their respective policy implementations, i.e., the regions with positive costs. Thus, the North East under the CAT is the one instance where a region's positive co-benefits (median $2.2; $0.8 to $3.5) billion 2005$ offset its negative costs of implementation (-$1.5 billion 2005$).

We compare co-benefits of the CES to the magnitude of its policy costs (whether negative or positive) in Figure 2-3. Everywhere, the welfare impact of implementing a CES is greater than that of the resulting reduction in fine particulate matter. Median CES co-benefits range from 1% (in California) to 10% (in New York) of the magnitude of policy costs. This pattern combines the relative importance of both co-benefits and costs. For example, the Pacific region has a ratio of 8% because its costs are less than in California and Mountain, which have similar co-benefits (co-benefits are Pacific: $0.25, California: $0.26, Mountain: $0.20 billion 2005$), and ratios of 1%. New York and New England similarly reach ratios of 10% and 9% by having the 3rd and 2nd lowest costs. Conversely, the South East has a high ratio compared to other regions because it has the highest co-benefits ($3.4 billion 2005$).

Figure 2-3: Ratio of median co-benefits to the magnitude of policy costs for the CES (%). Median CES co-benefits range from 1% (in California) to 10% (in New York) of the magnitude of policy costs.

CES Ratio of Co-Benefits to Magnitude of Costs (%)
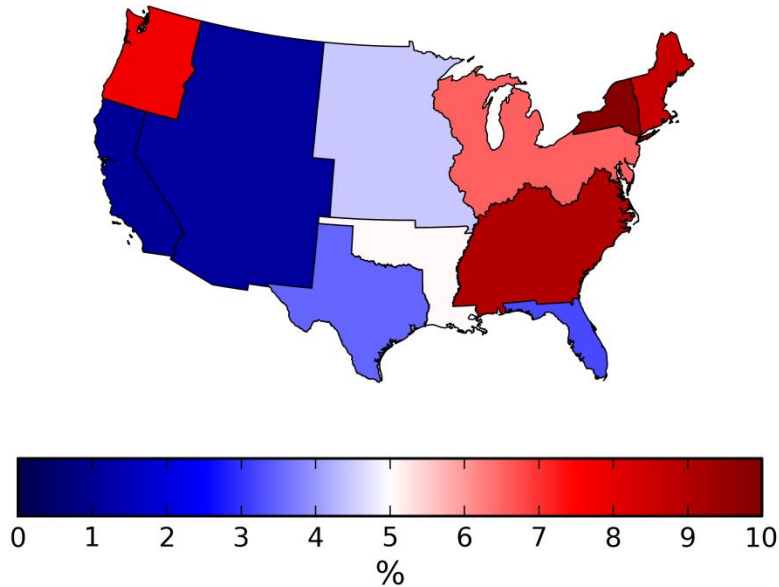
Figure 2-4 shows the ratios of co-benefits to the magnitude of costs for the CAT, which range from 11% to 690%. Co-benefits exceed the magnitude of policy costs in the East, and in four eastern regions. In the West, they are 0.3 times smaller overall, and range from 11% of costs (in Pacific and Mountain) to 32% in North Central. Co-benefits from pollution reduction in the East are 14 times the welfare impact of compliance. Co-benefits are greater than the magnitude of the cost in Florida, North East, New York, and South East by a factor of 1.4, 1.4, 1.9, and 6.9, respectively.

Figure 2-4: Ratio of median co-benefits to the magnitude of policy costs for the CAT (%). The relative welfare impact of pollution to policy implementation is greatest in the East, where co-benefits are 14 times greater than costs. Median values are plotted; for the CAT these range from 11% to 690%, and are >100% for Florida, New York, North East, and South East.

CAT Ratio of Co-Benefits to Magnitude of Costs (%)
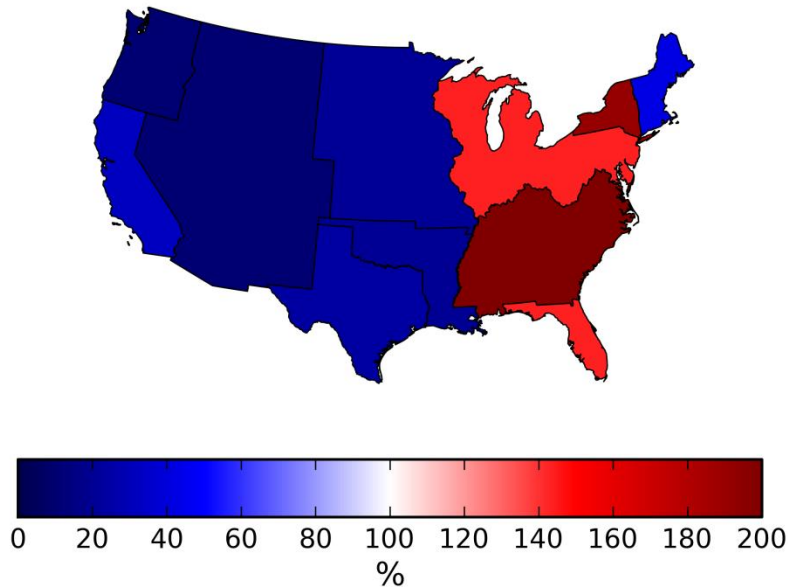
Contribution of General Equilibrium Economic Effects

National Co-Benefits: Contribution of Cumulative and Indirect Effects

In addition to estimating direct morbidity costs, this approach represents cumulative and indirect welfare gains as fine particulate matter is gradually reduced compared to BAU under each policy. Over the entire 2006-2030 period, the direct effects of morbidities are 9% and 7% of the total co-benefits for the CES and CAT. The remaining 91% and 93% of welfare impacts are the effects of price adjustments and labor productivity (from avoided mortality) that compound over time as they are applied to successively larger populations. This compounding of co-benefits from years prior to 2030 amounts to 42% and 45% of cumulative co-benefits for the CES and CAT, respectively.

Regional Co-Benefits: Patterns of Direct and Indirect Effects

We compare our distribution of co-benefits to one that values mortalities directly. This comparison illustrates how our approach yields a different regional pattern of co-benefits than

would be found using a typical VSL approach. VSL valuations of avoided mortality comprise the majority of fine particulate matter benefits in impact assessments that use them. For example, the Office of Management and Budget's Office of Information and Regulatory Affairs (OIRA 2013) cited air quality regulations as having the greatest benefits of all regulations reviewed, and those regulations have > 90% of benefits due to VSL valuations of fine particulate matter related mortality (e.g., US EPA 2012). Thus, the distributional implications of air quality benefits evaluated this way will have a pattern that largely follows the pattern of avoided mortality. Here we represent mortality as a labor impact, and we also account for the indirect effects of market interactions. Therefore, we expect our distribution of co-benefits to differ from our distribution of avoided mortalities. We explore that difference for each policy.

Under policy, each region avoids a certain number of mortalities, which is a percentage of the total avoided mortalities. Similarly, each region gains a particular share of our estimated co-benefits. Here, we explore the difference in the regional patterns of co-benefits and avoided mortalities by calculating the percentage by which the share of co-benefits differs from the share of avoided mortalities.

If the effect of mortality on welfare were identical in each region, then the difference in these patterns should be zero everywhere. Positive differences would mean that the co-benefits we calculate would be underestimated using one VSL for all regions, while negative values mean they would be overestimated.

Morbidities could explain small differences between the shares of co-benefits and mortalities by region. The direct costs of morbidities contribute less than 9% to co-benefits for either policy, and are highly spatially correlated with avoided mortalities (97% correlation of avoided morbidities and mortalities by region). Thus, we attribute differences of 10% or more in the share of co-benefits and avoided deaths to differences in the welfare impact of mortality by region, which can arise through differences in relative labor productivity and abundance, and the effects of trade.

For the CES, the distribution of co-benefits differs from the distribution of mortalities by as much as -25% in California to 25% in New York. Figure 2-5 shows the percent difference in the share of median co-benefits from the share of median mortalities, cumulative from 2006-2030. We find, for example, that New York has 4% of all mortalities avoided by the CES, but reaps 5% of the total welfare gain. Thus, the welfare gain from avoided mortalities in New York is greater than in other regions for the CES. The pattern of mortalities is a good predictor of co-benefits, explaining 99% of the variance in co-benefits between regions. However, if we were to apply the national average welfare gain from avoided mortalities to New York, we would underestimate its co-benefits by $132 million 2005$. Conversely, we would overestimate co-benefits in Texas by $138 million 2005$.

Figure 2-5: CES median difference in the share of co-benefits from the share of mortalities (%). For the CES, each region's share of co-benefits is different than its share of avoided mortalities. Compared to valuing mortality directly with VSL, this approach gives a distributional pattern of co-benefits that differs by as much as -25% in California to 25% in New York.
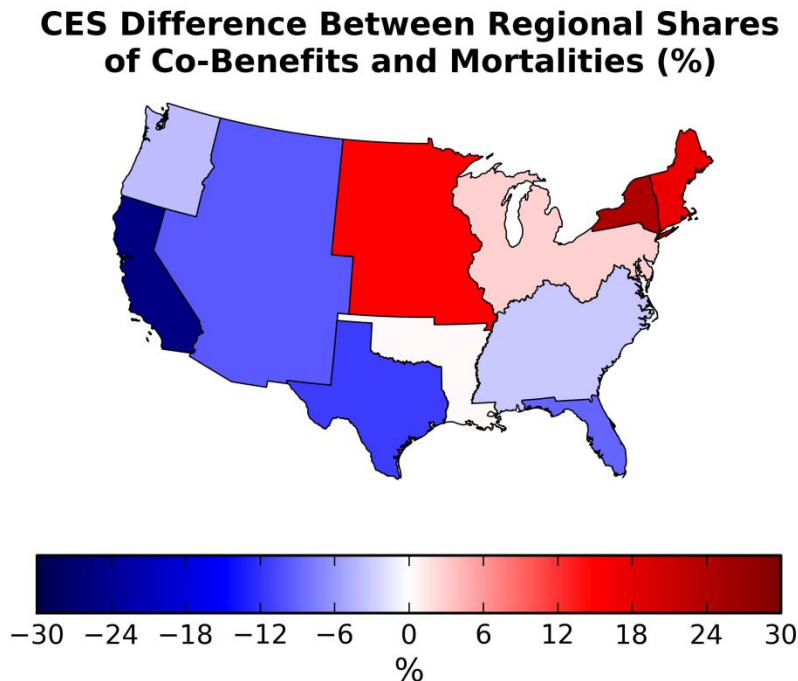
For the CAT, the distribution of co-benefits agrees with the distribution of avoided mortalities within 10% for seven regions, as shown in Figure 6. Using one valuation for mortality risk in all regions would explain over 98% of the variance in co-benefits between regions. It would, however, overestimate co-benefits in every region except North Central, New York, and New

England, and would differ in the share of co-benefits by as much as -15% in South Central to 27% in New England. Using the average welfare gain from avoided mortalities would under-estimate co-benefits in New York and New England by $100 million 2005$ and $220 million 2005$, respectively.

Figure 2-6: CAT median difference in the share of co-benefits from the share of mortalities (%). For seven regions, the pattern of the share of co-benefits matches that of mortalities within 10%. Using one valuation for mortality risk in all regions would overestimate co-benefits in every region except North Central, New York, and New England, and would differ in the share of co-benefits by as much as -15% in South Central to 27% in New England.

CAT Difference Between Regional Shares of Co-Benefits and Mortalities (%)
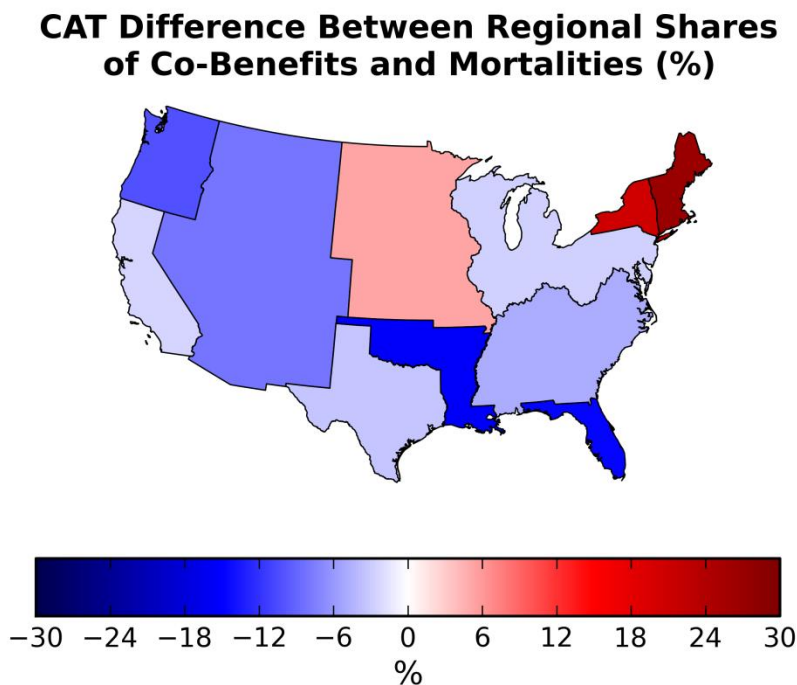
DISCUSSION AND CONCLUSION

We present a self-consistent integrated modeling framework to quantify the economy-wide co-benefits of fine particulate matter reductions under climate change and energy policy in the US. We employ a US economic model previously linked to an air quality modeling system, and enhance it to represent the economy-wide welfare impacts of fine particulate matter. We present a first application of this method to explore the efficiency and distributional implications of a Clean Energy Standard and a Cap and Trade program that both reduce CO_2 emissions by 10% in 2030 relative to 2006, including their ancillary impacts to fine particulate matter.

Based on our consistent methodological treatment of both climate policy costs and the co-benefits of air pollution reductions, we find that avoided damages from fine particulate matter alone can completely offset the costs of reducing CO_2 through cap and trade in the US. Up to 190% (median 110%; 40% to 190%) of the CAT's policy cost are offset by co-benefits of $6 ($2 to $10)/$tCO_2$ for the CAT. In the process of reducing CO_2 emissions by 10%, the CAT reduces fine particulate matter concentrations by 0.56 µg m^{-3} in 2030 relative to 2006, yielding health-related economic benefits that offset the cumulative welfare impact of the CAT to yield a net impact of $1 (-$5 to $7) billion 2005$.

Though the equivalent Clean Energy Standard yields higher pollution reductions (0.97 µg m^{-3}, population weighted annual average reduction in 2030 versus 2005), the corresponding co-benefit (median $8; $3 to $14)/$tCO_2$ is a smaller fraction (median 5%; 2% to 9%) of its higher policy costs. A less cost-effective means of reducing CO_2 than the CAT, the CES has a net welfare impact (co-benefits minus costs) of -$229 (-$237 to -$221) billion 2005$.

Including air quality co-benefits affects not only the efficiency but the distributional implications of each policy. The distributional pattern of co-benefits favors the East for both policies, which garners 78% and 71% of gains for the CES and CAT, respectively. On a net co-benefits basis, the CAT favors the East while the CES favors the West. The only area with a net gain under either policy is the East, which nets a gain of $6.7 ($2.7 to $10.7) billion 2005$ under the CAT. In several regions in the East under the CAT, the impacts of air quality are greater than the impacts of the policy by a factor of 1.4, 1.4, 1.9, and 6.9 for Florida, North East, New York, and South East, respectively. Overall, co-benefits from pollution reductions in the East are 14 times the welfare impact of compliance with the CAT.

General equilibrium effects shift our distributional implications compared to a traditional analysis that values mortalities directly. Over 90% of our co-benefits arise from a combination of labor and price adjustments as they compound over time, i.e., a combination of the direct labor effects of mortalities with general equilibrium effects. For both policies, avoided mortalities are a good predictor of the co-benefits within a region, but the welfare impact of a healthier labor

force differs across regions. General equilibrium effects and interregional differences in labor productivity and supply shift the distribution of our co-benefits compared to the pattern of avoided mortalities. For the CES, the patterns of co-benefits and mortality differ by as much as – 25% in California to 25% in New York. For the CAT, the distribution of co-benefits agrees with the distribution of mortalities within 10% for seven regions. However, using a single value for the welfare impact of mortality would overestimate co-benefits in most regions excepting New York and New England, where co-benefits would be understated by $100 million 2005$ and $220 million 2005$, respectively.

We track the economy-wide effect of mortality related to pollution as a reduction in the supply of labor, which yields lower benefits compared to studies that value mortality directly using VSL. Per ton of CO_2 emissions avoided, our co-benefits of $6 ($2 to $10)/$tCO_2$ for the CAT and $8 ($3 to $14)/$tCO_2$ for the CES are lower than previous work by Thompson et al. (2014) and reviewed by Nemet et al. (2010). Thompson et al. (2014) examined the same policies and particulate matter reductions studied here. They estimated the effects of morbidities and mortalities, which were valued using a VSL approach, yielding $170/$tCO_2$ for the CAT and $302/$tCO_2$ for the CES. Our results also fall at the low end of the range of $2-196/$tCO_2$ from 37 studies of air quality co-benefits of climate policy reviewed by Nemet et al. (2010), who note that the higher values of this range derived primarily from developing nations. We attribute this primarily to our treatment of mortality, in addition to our relatively clean setting and moderate policy stringency. In spite of our conservative valuation approach and policy setting, we find that air quality co-benefits can "pay for" 110% (40% to 190%) of the costs of a CAT, indicating the importance of co-benefits even in developed nations.

Our conclusions agree with and complement previous findings on the air quality co-benefits of climate policy. Air quality co-benefits are significant for both climate and energy policies, especially in regions with high pollutant emissions. The CAT policy is the lower-cost option per ton of CO_2 abated. The CES, which targets the polluting energy sector, reduces more fine particulate matter. However, we observe that the CES yields similar co-benefits to the CAT at 30 times the cost. When modeling the effects of a healthier labor force, ancillary pollution

reductions have welfare impacts that are nearly commensurate with policy costs for cost-effective, national quantity instruments, like a cap and trade policy.

Our approach yields complementary insights to past work by representing the impacts of a healthier labor force within different, interconnected regions. By allowing for market interactions and tracking labor effects, our approach suggests that the welfare effects of mortality are distributed differently than mortalities themselves. While mortality is a good predictor of co-benefits in a region, using a single value for the welfare impact of mortality would over- or under- estimate shares of co-benefits by up to 25%. That difference could switch the calculation of which policy – the CAT or the CES – yields higher co-benefits in certain regions, like New York. Traditional co-benefits analyses value mortalities directly, and will not capture these interregional differences and inter-market economic interactions that can serve to raise or lower co-benefits in a given region.

Our modeling approach yields insights rather than predictions of policy impacts. It relies on simplified production and behavior, and there are relevant processes we do not capture, such as future climate and future air quality. There are also uncertainties in the processes we do capture, for example, in our emissions, atmospheric concentrations, health impacts, valuations, economic growth, and the costs of low-carbon technologies. Some of these uncertainties are explored through sensitivity analysis by Thompson et al. (2014). The effect of another factor – model resolution – was explored for the air quality modeling in our base year of 2005 in previous work by (Thompson and Selin 2012) and (Thompson, Saari, and Selin 2014). Each of these factors is likely to alter our estimate of the mean and range of co-benefits, but not the conclusion that they can be important compared to policy costs for a CAT.

Implications for Policy Analysis

This work has implications for the design and evaluation of domestic carbon policies. Our results indicate that particulate matter reductions are sufficient to completely offset the costs of efficient carbon policy instruments, like a national cap and trade program. In this study, co-benefits transform the CAT's cumulative welfare impact from a net loss of -$8 billion 2005$ to a net gain of $1 billion 2005$. They also transfer gains from the West to the East, thereby increasing the set of regional "winners" to include the North East. Co-benefits exceed the costs of the CAT despite

consisting of welfare impacts of mortality represented simply by a reduction in labor supply for a single year, and of morbidity represented by a change in the demand for health services. This approach captures interregional differences in the effect of mortality on welfare, which, compared to using a single valuation of mortality, can alter the calculation of which policy yields greater co-benefits in a region. This approach also predicts relatively higher gains per unit of pollution reduction in high-productivity regions like New York, and lower gains in lower-productivity regions like Texas. This approach does not replace the use of VSL to value reduced mortalities, but it does identify interregional differences in the welfare impacts resulting from pollution-related labor impacts.

Regional planners might note that regional costs and co-benefits are affected by interregional differences and interactions, and not just local impacts. Flows across regions of pollutants and goods (including energy and CO_2 permits) also affect costs and co-benefits within a region. To maximize welfare under each national policy constraint, different regions will realize different CO_2 reductions and compliance costs. Some regions may mitigate little to no CO_2 if more cost-effective reductions are available elsewhere; for example, both New England and North Central reduce little to no CO_2 under the CES. At the same time, flows of pollutants, interregional differences in labor productivity and supply, and market interactions affect the pattern of co-benefits. New England has high benefits per ton under the CES both from its low reductions of CO_2, and its relatively high co-benefits as its upwind regions reduce fine particulate matter.

Regional planners might also note that the welfare impact per reduction of fine particulate matter varies across regions. For a region with high labor productivity, like New York, the impact of pollution reductions on consumer welfare is higher than elsewhere. Using the national average valuation for avoided mortalities for each policy would underestimate New York's co-benefits for the CES and CAT by 20% and 16%, respectively. Those underestimates mean that the policy leading to larger co-benefits in New York is the CES, not the CAT as might be estimated from the pattern of mortalities. For other regions, like Texas, capturing the relative importance of labor in the region strengthens its preference for a CAT, both on a co-benefits and a net co-benefits basis.

In future studies, we can apply this new framework directly to explore the air quality impacts of other climate or energy policies. We can explore other pricing instruments, such as a carbon tax, or regional instruments like a California CAT (as were previously studied with USREP in Rausch et al. 2011 and Caron, Rausch, and Winchester 2015; the co-benefits of California's AB32 were also studied by Zapata, Muller and Kleeman (2013). We can examine the air quality impacts of carbon tax swap policies that use revenue from a carbon price to reduce distortionary taxes (as studied with USREP in Rauch, Metcalf and Reilly (2011) and Rausch and Reilly (2012). Building on the work presented here, we could use our approach iteratively to quantify feedbacks between air quality and private consumption, the importance of which are discussed in, for e.g., Smith and Carbone (2007) and Goulder and Williams (2003).

Next, we can extend our work in order to better understand the interactions of air pollution with US environmental and energy policy. We can use our detailed air quality modeling system to include other pollutants, such as ozone, nitrogen oxides and sulfur dioxides. By combining those pollutants with an endogenous representation of pollution abatement (as in Nam et al. (2013)), we can explore the interaction of climate and air quality policy. This includes avoided pollution mitigation costs, potentially competing effects of markets for carbon and pollution, and potentially diminishing co-benefits as air quality improves. With this economy-wide, integrated assessment framework as a basis, we can quantify consistent air quality implications and their effect on the efficiency and distribution of domestic energy and environmental policy.

SUPPLEMENTAL INFORMATION

Median Change vs. BAU	CES	CAT
Emissions Reduction in 2030 (%)		
NOx	-5.6%	-2.9%
CO	-1.2%	-1.5%
SO_2	-24.4%	-9.6%
VOC	-1.3%	-1.5%
NH_3	-0.9%	-0.4%
PM_{10}	-2.9%	-2.5%
$PM_{2.5}$	-4.9%	-4.6%
Mortalities Avoided in 2030*	36,591	24,917
Cumulative Pollution Health Demand (million 2005$)	$1,090	$570

* Calculated as by first estimating the change in mortalities in 2030 vs. 2005 PM2.5 levels for the policy, then subtracting the change in mortalities in 2030 vs. 2005 PM2.5 levels under BAU For example: Mortalities avoided for CES = CES_mortalities(PM_CES_2030 - PM_2005) - BAU_mortalities(PM_BAU_2030 - PM_2005)

Third Essay

3. Human Health and Economic Impacts of Ozone Reductions by Income Group

ABSTRACT

Background Ground-level ozone may disproportionately affect the health and economic welfare of low-income households. Methods are needed that can simultaneously track the health-related impacts of ozone policies and their distributional implications.

Objectives We quantify how inequality in ozone levels, delayed policy action, and approaches to economic valuation affect the relative economic value of ozone policies with household income.

Methods We use an advanced air quality modeling system and health impacts model along with Census data for the continental U.S. to estimate ozone levels and health incidence rates by income group. We extend a U.S. economic model to represent the health-related economic impacts of ozone, and apply it to assess the relative economic value of ozone reductions for low-income households.

Results Using modeled ozone levels in 2005, we estimate a median mortality incidence rate of 11 deaths/1000,000 people that decreases monotonically with income. The EPA's planned reductions for 2014 reduce these rates by 13%, favoring reductions among the lowest income households. Treating exposure as equal by income group understates relative welfare gains for the poor (by about 4%), and overstates them for the rich (by up to 8%). Delaying those reductions until 2025 is relatively twice as harmful for the lowest income households. Using an income-based approach to estimate the impact of mortality instead of the Value of a Statistical Life drops the median annual value of reductions in 2015 from $30 billion to $1 billion. Under both valuation approaches, ozone reductions are relatively more valuable to low-income households, who are also the most relatively affected by delay. The relative value of ozone reductions is most affected by the valuation approach, followed by delay, and finally by the inequality in ozone reductions across income groups.

Conclusions Modeled inequality in ozone levels shifted the relative gains of ozone policies toward low-income households.

INTRODUCTION

Tropospheric ozone is a harmful pollutant that affects human health and economic welfare (Schwartz 2005; Bell, Dominici, and Samet 2005; Selin et al. 2009). U.S. ozone levels vary by household income group (Miranda et al. 2011; Marshall 2008; Liu 1996; Liu 1998; Grineski, Bolin, and Boone 2007). Correspondingly, policies to reduce ozone can affect household income groups differently (Bento, Freedman, and Lang 2014). Such findings, both in the inequality of exposure and the effect of policy, motivate the U.S.-wide assessment of ozone policy with household income. Further ozone reductions are planned or proposed under EPA policy (Berman et al. 2012; U.S. Environmental Protection Agency 2015b), some of which have already been delayed from their planned start date (U.S. Environmental Protection Agency 2015a). Previously, the EPA has explored strategies to target reductions based on poverty rates (Fann et al. 2011), and has analyzed the effects to the lowest 10% of household incomes (U.S. Environmental Protection Agency 2012), but it does not systematically evaluate effects by income group (Fraas 2011). Here, we perform a systematic assessment of the relative health and economic impacts of ozone reductions by household income group. We enhance an integrated modeling framework designed to assess the air quality impacts of environmental policy to track the health-related impacts of ozone with income. For a set of ozone policies that were planned for 2014, we model ozone levels by income group and assess the relative economic value of reductions for low-income households.

Both ozone levels and ozone policies can have different effects across income groups. Health risks from ozone might be higher for low-income households if they experience higher ozone levels (Bell and Dominici 2008). U.S. studies have found that low income households can be either more (Grineski, Bolin, and Boone 2007; Liu 1998) or less (Miranda et al. 2011; Marshall 2008; Liu 1996; Liu 1998) likely to be exposed to high levels of ozone. These findings apply to different regions of study, including counties with high ozone (Liu 1998; Miranda et al. 2011), and specific urban areas (Liu 1996; Grineski, Bolin, and Boone 2007; Marshall 2008). Inequality in exposures can affect the relative value of policies of low-income households. For example, Bento, Freedman and Lang (2014) suggest that the Clean Air Act Amendments favored reducing exposure among low-income households, and were consequently twice as valuable for the poor.

Such a posteriori studies rely on measurements, whereas models are needed to assess the effect of future policies with income.

Assessing the relative health and economic importance of future ozone policy by income group remains a challenge. The current U.S. regulatory approach is to treat the valuation of pollution-related health impacts equally with household income. While both health effects and environmental preferences may vary with income, there are gaps and challenges for quantifying this variation, as evidenced by reviews of income and ozone-related health effects (Bell and Dominici 2008), of income and health services (Goddard and Smith 2001; van Doorslaer et al. 2000), and income and risk preference or environmental preference (van Kippersluis and Galama 2014; Viscusi and Aldy 2003; Chemingui and Thabet 2014; Alberini and Ščasný 2013). Using the regulatory approach alongside other approaches could offer insight about the effect of these debates on the relative value of ozone reductions for low-income households. Despite these gaps, we can still develop tools that will tell us what the current techniques imply for regulatory policy, and explore the implications of alternative methods.

One method for assessing the economic impact of ozone policy with income is integrated assessment modeling. Advances in these techniques using Computable General Equilibrium (CGE) economic modeling can complement traditional regulatory Partial Equilibrium (PE) policy assessments to provide new insights about the relative impacts and long-run dynamics of environmental policy. An advantage of this approach is its capacity to put the value of ozone reductions in the context of other economic factors, forces, and decisions, so that the relative economic impact of policies can be assessed with household income. There are other reasons which make CGE analysis a useful complement to the regulatory use PE approaches. Partial equilibrium estimates can differ significantly from general equilibrium (GE) for large, diverse environmental changes, like significant ozone reductions (Sieg et al. 2004; Tra 2010). CGE modeling of air pollution reductions estimate the social costs of reductions to be much higher than PE estimates (Hazilla and Kopp 1990; Goulder, Parry, and Burtraw 1997). PE and GE estimates of the benefits of ozone reductions in Los Angeles were found empirically to differ substantially in magnitude and sign because the GE estimates accounted for price responses (Sieg et al. 2004). CGE modelling has been used in regulatory reports, including the EPA Second

Prospective Report on CAA (U.S. Environmental Protection Agency 2011a), which estimated impacts across four household income categories, but did not include ozone mortality. Other authors have included ozone mortality in CGE estimates of pollution impacts but did not examine effects by income group (Matus et al. 2008; Selin et al. 2009; Nam et al. 2010; Matus et al. 2012).

In this study, we seek to quantify the relative health and economic impacts of planned ozone reductions with household income across the continental United States. We systematically model the effects across nine household income categories of a scenario evaluated by the U.S. EPA for the Cross State Air Pollution Rule comprising a set of policies that were planned for 2014 (U.S. Environmental Protection Agency 2011b). We use this modeling experiment to examine the relative importance of U.S. policies for low-income households, and to explore the importance of policy delays, unequal ozone levels, and methodological choices on this result. To perform our integrated modeling, we extend a framework, elaborated elsewhere (Thompson et al. 2014; Saari et al. 2015), that connects a regional chemical transport model (Comprehensive Air Quality Model with extensions (CAMx)) and a health impacts model (Benefits Mapping and Analysis System (BenMAP)) with a CGE model of the U.S. energy and economic system (U.S. Regional Energy Policy model (USREP)). USREP can assess the long-run dynamics of the economy under policy, including the relative economic impacts of policies over nine household income categories. Here, we enhance this framework to be able to examine the health and economic effects of ozone by income group. We ask, what do modeled U.S. ozone levels in 2005 and reductions planned for 2014 imply for health outcomes by income group? What is the relative economic benefit of planned ozone reductions for 2014 by income group? How is this affected by delay, by unequal ozone reductions with income, and by the valuation approach?

METHODS

We model ozone concentrations and their economic impacts using an integrated assessment framework similar to Thompson et al. (2014) and Saari et al. (2015). This framework links an advanced air quality modeling system to an economic model capable of analyzing impacts across income groups. This sequential system comprises an emissions model, the Sparse Matrix Operating Kernel Emissions model (SMOKE), an air quality model, the Comprehensive Air Quality Model with Extensions (CAMx), a health impacts model, the Environmental Benefits

Mapping and Analysis Program (BenMAP), and an economic model, The U.S. Regional Energy Policy model (USREP). This section describes our methods for modeling ozone, health impacts, and economic impacts, and their application to analyzing ozone reductions.

Ozone Modeling

We use CAMx version 5.3 to estimate future hourly ozone concentrations on a 36 km grid of the continental U.S. using the same year-long 2005 modeling episode described in Thompson et al. (2014), and documented and evaluated against ambient monitoring data in U.S. EPA (2011b). CAMx is used regularly in air quality policy analysis (U.S. Environmental Protection Agency 2011b; U.S. Environmental Protection Agency 2007). We temporally process and speciate emissions for the years 2005 and 2014 using the Spare Matrix Operating Kernel Emissions (SMOKE) (CMAS 2010). We apply meteorological input for the year 2005 in both present and future simulations, which was developed with the fifth generation Penn State/NCAR mesoscale model MM5 (Grell, Dudhia, and Stauffer 1994).

Health Outcomes and Valuations

We calculate mortality and morbidity resulting from ozone concentrations using the Environmental Benefits Mapping and Analysis Program (BenMAP) v4.0 following the methods used in U.S. EPA (2012). Table 3-1 lists the endpoints and concentration-response functions applied. We have estimated the lower and upper bound of the number of health impacts based on 1000 Monte Carlo simulations of the 95% uncertainty bounds of the concentration-response functions as specified in Table 3-1. We assign the resulting estimates to household income groups based on Census data.

Table 3-1: Health impact functions and valuations

Endpoint / Endpoint Group	Ages (yrs)	Individual Studies	Pooling and Lower/Upper Bounds	Valuation 2006$
Premature Mortality	0-99	Ito et al. (2005) Schwartz (2005) Bell et al. (2004) Bell et al. (2005) Levy et al. (2005) Huang et al. (2005)	Equal weight pooling of all studies	$8 million

Respiratory Hospital Admissions	>65	Schwartz (1995)—ICD 460–519 (all respiratory) Schwartz (1994a; 1994b)—ICD 480–486 (pneumonia) Moolgavkar et al. (1997)—ICD 480–487, 490–496 (pneumonia, COPD) Schwartz (1994b)—ICD 491–492, 494–496 (COPD)	Random effects pooling of outcomes from four cities: Tacoma, New Haven, Detroit, and Minneapolis	$28,000
	<2	Burnett et al. (2001)	N/A	$10,000
Asthma-related ER Visits	5-34 All ages	Jaffe et al (2003) Peel et al (2005) Wilson et al (2005)	Random/fixed effects pooling of all 3 studies	$370
Minor Restricted-Activity Days	18–64	Ostro and Rothschild (1989)	N/A	$60
School Loss Days	5-17	Gilliland et al. (2001) Chen et al. (2000)	Random/fixed effects pooling of both studies	$90

ER= Emergency Room
ICD = International Statistical Classification of Disease

The US Regional Energy Policy Model

USREP Model Description

USREP is a recursive dynamic computable general equilibrium (CGE) economic model designed to explore the environmental impacts and distributional implications of environmental and energy policy. It calculates the commodity prices that support equilibrium between supply and demand in all markets across 5-year time periods to assess the long-run dynamic effects of policy on resource allocation and income distribution. USREP has been described and applied in the literature, including: tests of its structure, inputs, and assumptions; model inter-comparisons; economic and distributional impacts of climate change and energy policies; and economy-wide impacts of air pollutants including ozone, fine particulate matter, and mercury (Rausch et al. 2010; Rausch, Metcalf, and Reilly 2011; Lanz and Rausch 2011; Rausch and Karplus 2014; Caron, Rausch, and Winchester 2015; Rausch and Mowers 2014; Thompson et al. 2014; Saari et al. 2015; Giang 2013).

We estimate outcomes for consumers in nine household income categories, divided across 12 geographic regions shown in Table 3-2. These utility-maximizing consumers supply four factors of production (labor, capital, land, and resources) to profit-maximizing firms. In this full

employment model, consumers have a time endowment they dedicate to labor or leisure. They pay taxes and collect transfers from the government (Rausch et al., 2010).

Table 3-2: USREP Model Details: Regional, Household, and Sectoral Breakdown and Primary Input Factors

Sectors	Regions	Primary production factors
Non-Energy	Pacific (PACIF)	Capital
Agriculture (AGR)	California (CA)	Labor
Services (SRV)	Alaska (AK)	Coal resources
Energy-intensive products (EIS)	Mountain (MOUNT)	Natural gas resources
Other industries products (OTH)	North Central (NCENT)	Crude oil resources
Commercial transportation (TRN)	Texas (TX)	Hydro resources
Passenger vehicle transportation (TRN)	South Central (SCENT)	Nuclear resources
Pollution-related Health Services (PH)	North East (NEAS)	Land
Final demand sectors	South East (SEAST)	Wind
Household demand	Florida (FL)	
Government demand	New York (NY)	Household income classes
Investment demand	New England (NENGL)	($1,000 of annual income)
Energy		<10
Coal (COL)		10-15
Natural gas (GAS)		15-25
Crude oil (CRU)		25-30
Refined oil (OIL)		30-50
Electric: Fossil (ELE)		50-75
Electric: Nuclear (NUC)		75-100
Electric: Hydro (HYD)		100-150
Advanced Technologies		>150

We estimate the relative economic impact of policies through their effect on consumer welfare, which comprises consumption (capturing market-based activities) and leisure (capturing non-working time) (Paltsev and Capros 2013). We estimate the change in consumer welfare as the equivalent variation, or the income amount that consumers would pay to avert (or ensure) the price effects of a policy; in this case, due to health-related impacts from ambient ozone. We present the equivalent variation across income groups as an advantage of this modeling approach, which can track the importance of policies compared to all other pre-existing interacting policies, resource allocations, and price responses that affect our estimates of consumer welfare (Rausch et al. 2010).

Production also responds to policy. USREP represents production in nested constant-elasticity-of-substitution (CES) functions involving 5 energy commodities, 5 non-energy sectors and advanced energy technologies shown in Table 2 and described in Rausch et al. (2010).

USREP has a base year of 2006 and solves in 5-year periods to 2100. It is based on state-level economic data from the Impact analysis for PLANning (IMPLAN). It incorporates energy data from the Energy Information Administration's State Energy Data System (SEDS), and regional fossil fuel reserves from the US Geological Service and the Department of Energy, with details in Rausch and Mowers (2014).

Pollution Health Services Sector in USREP

To represent the economic and welfare impacts of health-related ozone, we enhance the pollution health services sector in USREP. This sector accounts for morbidities and mortalities in the economy through lost wages, lost leisure, and medical expenses that vary with pollution levels. It draws inputs from services ("SRV") sector and from the household labor supply. It affects consumer welfare through private consumption and leisure. This sector was introduced in Saari et al. (2015) and is similar in approach to previous work using global CGE models (Matus et al. 2008; Selin et al. 2009; Nam et al. 2010; Matus et al. 2012).

Valuation of Health Endpoints

We present results from two approaches to valuing health endpoints with income in our CGE modeling framework. The first is meant to represent the current regulatory approach, which is supported by theoretical arguments and an increasing number of empirical estimates (Viscusi and Aldy 2003; Viscusi 2009; Cropper, Hammitt, and Robinson 2011). The second represents a typical approach employed in economy-wide models; theoretical and empirical questions remain regarding how to best represent human capital or environmental preferences in economy-wide assessments (Carbone and Smith 2013; Arrow et al. 2013). Both approaches are implemented within our CGE framework. They represent different quantities and are not comparable. However, presenting both approaches places these valuation challenges in the context of the policy-to-impacts pathway, of ozone inequality, and of the effect of delay. It also demonstrates that our modeling framework can be adapted to a variety of valuation techniques to aid the distributional analysis of policy. We discuss each approach in turn.

We first assign the valuation of each endpoint following U.S. EPA (2012) as shown in Table 3-1. We apply the same valuation to each outcome regardless of household income. While the EPA Guidelines for Economic Analysis recognize potential heterogeneity in environmental preferences across the population, citing income elasticity specifically (U.S. Environmental Protection Agency 2014b), agencies typically apply a uniform Value of a Statistical Life (VSL) to all groups across regulations (Viscusi 2009). The VSL has also been used in CGE assessments of the health impacts of air pollution reductions (Chemingui and Thabet 2014).

Our second valuation approach follows previous analyses using CGE models, including the EPA's benefits assessment of the Clean Air Act Amendments (Matus et al. 2008; Selin et al. 2009; Nam et al. 2010; Matus et al. 2012; Saari et al. 2015; U.S. Environmental Protection Agency 2011a). The valuations are the same as in Table 3-1, but mortality is represented as 0.5 years of lost income, differentiated by household income category. Reduced mortality risk is the only valuation we vary with household income as we lack relevant empirical relationships between income and cost of illness (for hospitalizations and ER visits) and income and willingness-to-pay (for minor restricted activity days). The morbidity endpoint valuations are the same as the regulatory approach in US EPA (2012). They are based on cost of illness estimates (for hospitalizations), lost wages (for school lost days), and willingness-to-pay (for minor restricted activity days). The valuation per outcome determines the demand for "pollution health services" in USREP.

Assessing Ozone Exposure and Impacts under Planned Reductions

We analyze a scenario of ozone reductions that the EPA developed to evaluate the Cross State Air Pollution Rule (CSAPR) (U.S. Environmental Protection Agency 2011b). This hypothetical 2014 scenario includes not only the CSAPR, which has been delayed by judicial actions (U.S. Environmental Protection Agency 2015a), but other policies and plant closures, with details in EPA (2011b). These reductions were planned for 2014 starting from a base year of 2005. We use these hypothetical scenarios as a modeling experiment to assess the value of the reductions with income. In our Policy Scenario, we implement the 2014 reductions linearly between 2005 and 2014. Recognizing that planned implementation can be delayed, we include a case where implementation spans 2015 through 2025. Finally, we isolate the role of unequal ozone

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reductions with income. We compute separate USREP runs that apply the average ozone reduction for each region to all households regardless of income group. These scenarios are enumerated below:

- Base Case 2005: constant 2005 ozone levels

- Policy Scenario 2014: reductions implemented between 2005 and 2014

- Policy Scenario 2025: reductions implemented between 2015 and 2025

- Equal Ozone Reductions with Income: reductions are adjusted to be equal across household income groups within a given region

RESULTS

Ozone-Related Mortality Incidence Rates by Income Group

Figure 3-1 shows the estimates of mortality incidence rates from ambient ozone by household income group in the Base Case and Policy Scenario 2014. Ozone levels in 2005 imply a median mortality incidence rate of 11 deaths/100,000 people. Based only on modeled differences in ambient ozone levels, this incidence rate is 3% higher for lowest income than highest income households. At the national scale, mortality incidence rates decrease monotonically with income, but this pattern varies by region. In all regions, the pattern of the population-weighted annual mean of the 8-hour daily maximum ozone level spanned a range of about 2 ppb across income groups. Refer to supplemental information for regional mortality incidence rates with income.

The planned ozone reductions affect both the magnitude and pattern of mortality incidence rates with income. Under the modeled Policy Scenario, national ozone-related mortality incidence rates decrease by about 1.3 deaths per 100,000 people per year, from 10.8 to 9.5 deaths per 100,000 people per year. The pattern of mortality incidence rates still decreases with increasing income, but it is slightly flatter; the Policy Scenario decreases the incidence rate by 13% for the lowest income households and by 12% for the highest income households.
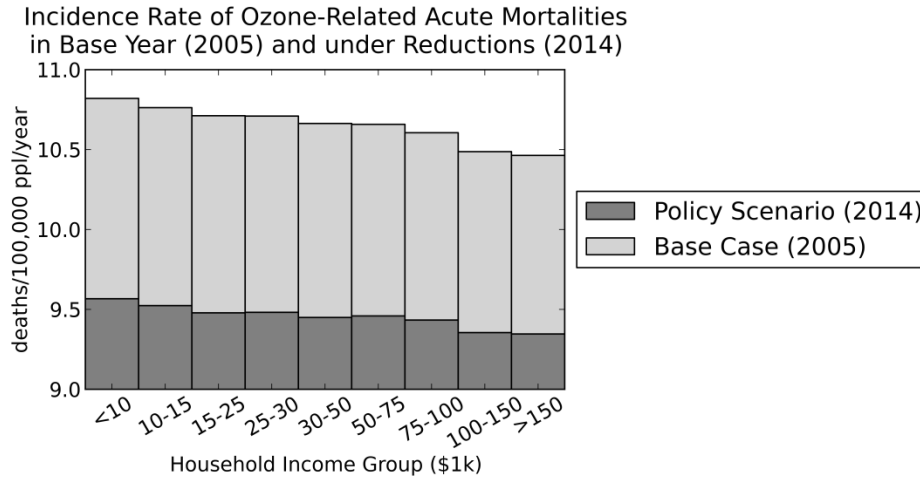
Figure 3-1: Decreasing pattern with household income of U.S. national incidence rate of ozone-related mortalities in the base year (2005) and Policy Scenario (2014).

Relative Economic Impacts of Reductions with Income Group

In this section, we first discuss results using a single VSL to value reduced mortality risk, then discuss the effect of accounting for lost income instead. The solid line in Figure 3-2(a) shows the relative per capita welfare gain (i.e., equivalent variation) from the ozone reductions with income. This is based on the net present value of annual welfare gains between 2005 and 2100 discounted at 7%. Refer to supplemental information for the undiscounted annual welfare gains over time of reductions compared to constant 2005 ozone levels.

As shown in Figure 3-2(a), in the solid line, the median per capita welfare gain decreases with increasing income, with the 95% confidence interval in the supplemental information. In this relative sense, the lowest income households gain twice as much as highest incomes (0.21% vs. 0.11% of per capita welfare). Similarly, the relative welfare loss from delaying regulations is twice as harmful for low as high income households. Figure 3-2(a), in the red shading between the solid line and the dot-dashed line, shows the welfare gain foregone by delay. The delay to 2025 foregoes about 50% of the potential gains from the Policy Scenario discounted at 7%. This amounts to a reduction in the equivalent variation of 0.1% for the lowest income households, and 0.05% for the highest income households.
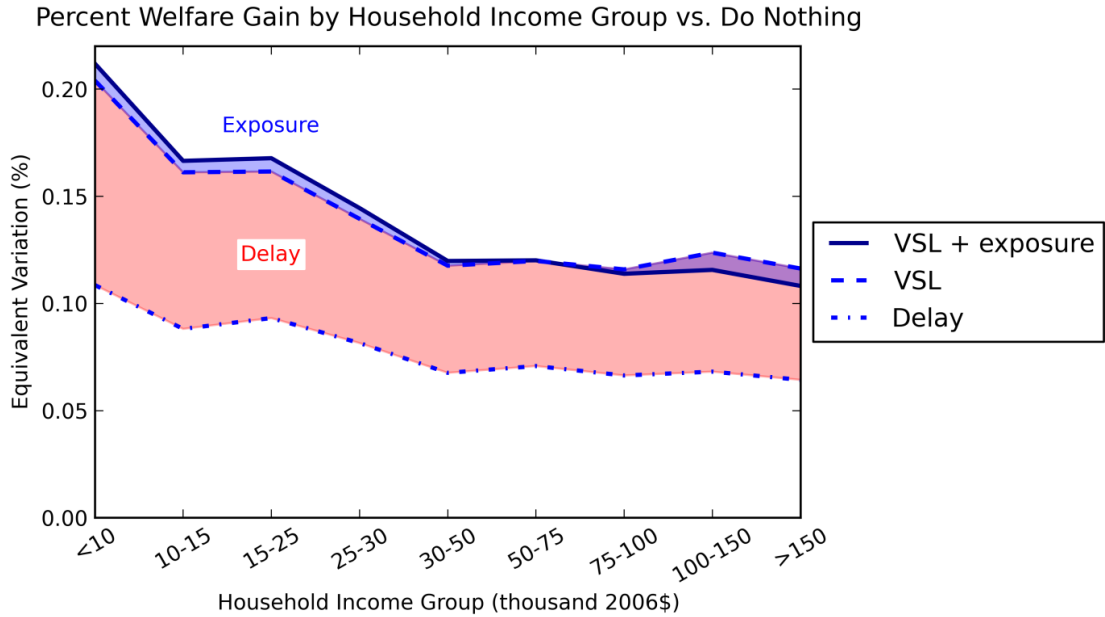
(a)
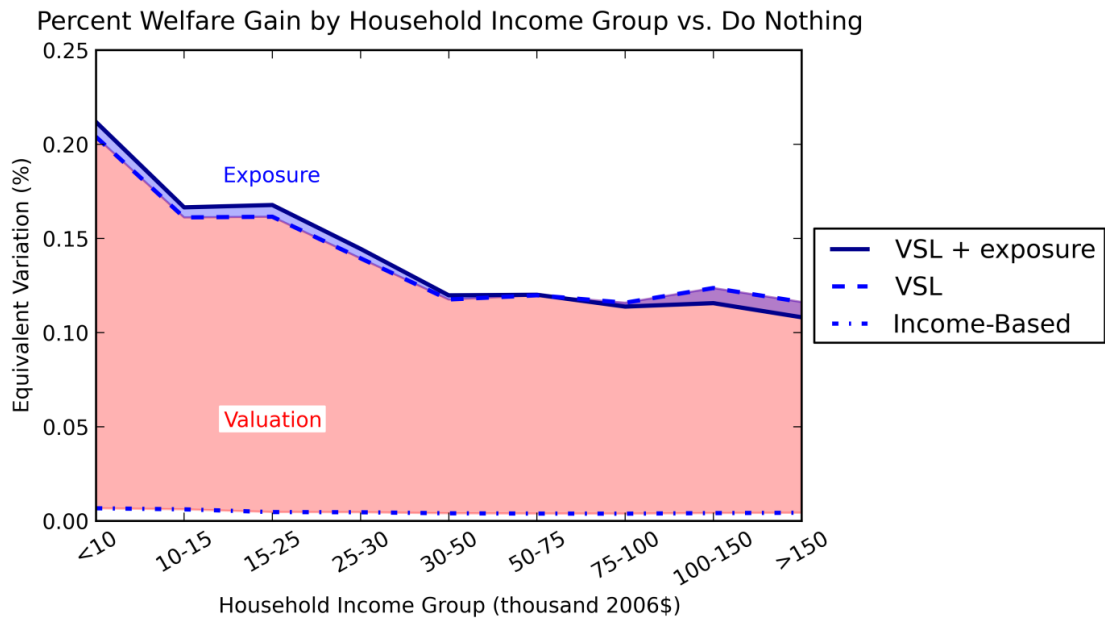
(b)

Figure 3-2: Percent welfare gain by household income group. Solid blue line uses VSL valuation for reduced mortality risk.

Blue dashed line shows run with ozone reductions equal across income. (a) Dot-dash line shows implementation delayed to 2025. Red shading is welfare loss from delay. (b) Dot-dash line shows income-based valuation. Red shading is welfare difference between valuations.

Figure 3-2 also depicts the effect of the difference in ozone reductions with household income. Figure 3-1 showed that the ozone reductions in the Policy Scenario happened to favor low-income households slightly. In Figure 3-2(a) the dashed line presents a separate USREP model analysis that assigns the regional average change in health outcomes to all households within a region. The shaded blue area between the solid and dashed lines shows the difference between these runs. To see this effect more clearly, refer to Figure 3-3, which shows the percent change in the relative per capita welfare between the Policy Scenario and the Equal Ozone Reductions with Income runs. Treating the reductions in ozone levels as equal by income group would understate relative welfare gains for the poor (by about 4%), and overstate them for the rich (by up to 8%).

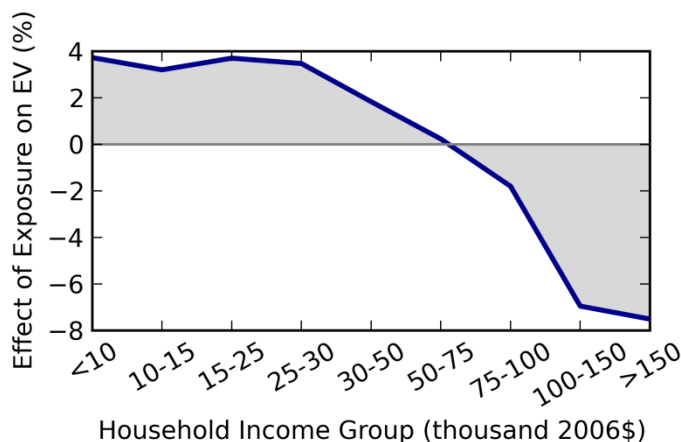Percent Effect of Ozone Differences on Relative Welfare Gain

Figure 3-3: Effect of accounting for differential ozone reductions across household income groups on relative welfare gain.

Figure 3-2(b) also presents results using our second valuation approach, which accounts for lost income from premature mortality based on household income category. With this approach, the highest relative per capita welfare gain still goes to households with less than $10k in annual income, but its value is 0.007% instead of 0.22%. The annual gain in 2014 is $1 billion compared to $30 billion using the VSL. The choice of valuation approach has a larger effect on the relative policy gains than the effect of delay or differences in ozone reductions with income.

Across income groups, however, certain findings hold with both valuation approaches. Figure 3-4 places the normalized relative gains under both valuations on the same scale. With the income-based approach, households with the lowest incomes still have the highest relative gains;

however, instead of monotonically decreasing with income, the relative value of reductions begins to increase for households with incomes higher than $75k. Delay has a similar effect for both types of valuation, foregoing half of the relative gains for the lowest income households, and being about twice as harmful (factor of 1.8) for the lowest compared to the highest income households.
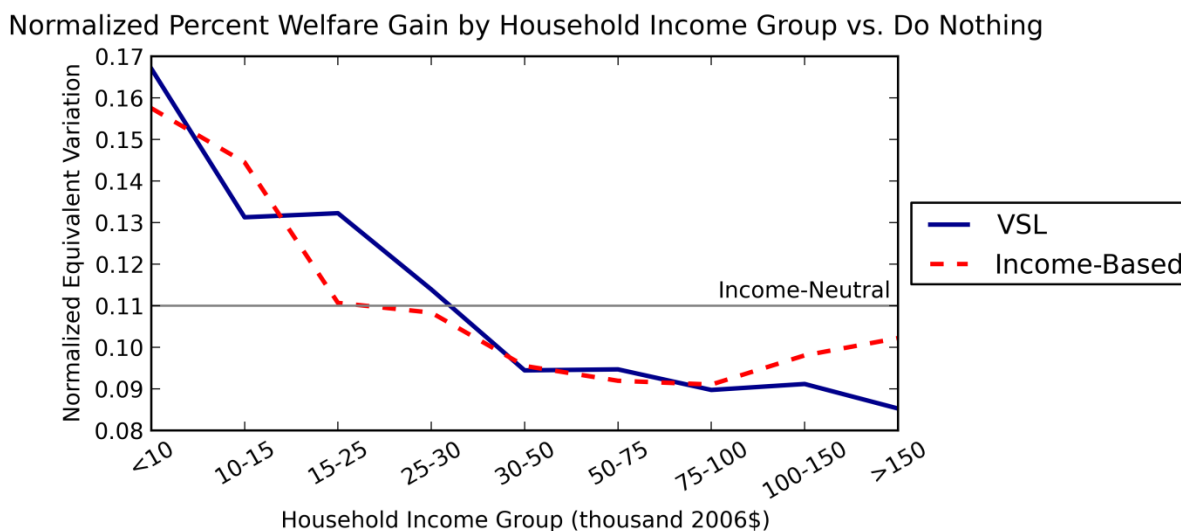
Normalized Percent Welfare Gain by Household Income Group vs. Do Nothing

Figure 3-4: Normalized percent per capita welfare gain of the Policy Scenario by household income group. Blue solid line employs the VSL-based valuation; red dashed line employs the income-based valuation.

DISCUSSION

Ozone-Related Mortality Incidence Rates by Income Group

Previous studies using air quality models or ambient monitors have found different relationships between ambient ozone and income in the United States, depending on the region of study (Miranda et al. 2011; Marshall 2008; Liu 1996; Liu 1998; Grineski, Bolin, and Boone 2007). We find, at the national scale, in 2005, that short-term ambient ozone levels implied a higher mortality rate for low-income households. Consistent with previous studies, we find that this pattern of ozone and income varies between regions. In all regions, the pattern of the population-weighted annual mean of the 8-hour daily maximum ozone level spanned a range of about 2 ppb across income groups, which is significant from a regulatory perspective, though it falls within the 95%CI of our uncertainty estimates. By using a modeling study at 36-km, we are able to cover a greater proportion of the population at the expense of higher resolution.

Resolution is one of several factors that could make actual ozone-related mortality incidence rates different from our findings. We focus only on the effect of modeled ozone levels at 36-km resolution at the household's location. Bell and Dominici (2008) note that the effect of ozone on health risks will potentially be affected by underlying health status, difference in exposure, or other factors. Indeed, underlying health status is linked to socioeconomic status (Marmot 2007). Bell and Dominici (2008) did not find their measure of income – specifically, the median poverty rate – to modify the effect of ozone on mortality, though it correlated with another effect modifier, unemployment.

Relative economic impacts of reductions with income group

USREP can categorize general equilibrium effects to economic welfare with household income. The relative general equilibrium impacts can encompass interactions between policies, and assess the relative economic importance of ozone reductions. Empirical studies have shown that general equilibrium effects can have an important influence on the benefits of ozone reductions (Tra 2010; Tra 2013). Our results also account for cumulative effects, which previous CGE studies of ozone benefits showed to be important (Selin et al. 2009). Our approach offers the flexibility of changing the timing of policy implementation, allowing us to explore, for example, the effect of delay.

We find, using two different valuation approaches, that ozone reductions can be relatively more valuable for low-income households. Both valuations also yield the result that an 11 year delay of ozone reductions discounted at 7% is relatively twice as harmful for the lowest income households as the highest income households. This relative analysis places the importance of ozone reductions in context of other sources of economic welfare. In magnitude, rather than in the relative sense, our benefits per capita do increase with income, which is consistent with the empirical findings of Tra (2010) that willingness-to-pay for ozone reductions increased with income.

By developing this method, we are able to explore what the regulatory approach implies for the value of reductions, and to test the effect of other approaches and factors. Specifically, for this Policy Scenario, we find that the difference in the median benefits estimates between the two

valuation approaches was larger than the loss from delay and the effect of unequal reductions across income groups.

This study has implications for the air quality policy community. It supports the notion of including analysis by household income as part of the equity assessment of air quality policy. Fann et al. (2011) describe the potential for targeting fine particulate matter reductions in vulnerable and susceptible sub-populations to improve metrics of risk inequality. Our study explored a wider region at a lower resolution for a different pollutant, ozone, and we examined the effect of a Policy Scenario that did not purposefully attempt to reduce inequality. We did find that differential reductions with income favored low income households in a relative sense; however, our results suggest there is more potential to address inequality through social and economic policy than through targeted reductions in ambient ozone.

Our results highlight the need to develop empirical relationships between environmental preferences and health outcomes that can represent the full range of income inequality. Various findings suggest that risk preferences, health care access, and health outcomes could vary with income, region, and insurance status (Viscusi and Aldy 2003; Van Ourti, van Doorslaer, and Koolman 2009; Jones et al. 2011; Wilper et al. 2009; Schoen et al. 2013), though we lack specific empirical relationships to apply to this case. Even with those relationships, we would still be restricted to results based on the Census household income categories, which do not capture the full range of income disparities, especially for top incomes (Piketty and Saez 2013), and do not capture the considerable variability of consumption within income groups (Rausch, Metcalf, and Reilly 2011).

In addition to those already discussed, there are numerous uncertainties and factors that affect this analysis. There are uncertainties in the estimates of ambient ozone levels, and in future meteorology. USREP is based on simple equations of production, and assumptions about economic behavior. Our projections of future demographics are based only on population growth and do not capture the potential interaction of migration and economic mobility. Our estimates of the value of delay will depend on the discount rate. The dynamic element of our approach could be improved by, for example, accounting for climate feedbacks (Knowlton et al. 2004; Bell et al.

2007; West, Szopa, and Hauglustaine 2007; Chang, Zhou, and Fuentes 2010), or introducing endogenous implementation of pollution control (Nam et al. 2013). This study serves to demonstrate these tools and their potential to complement and extend current health and economic impacts assessment. It agrees with previous findings that ozone levels can differ with income, and that policies may produce unequal reductions with income.


CONCLUSIONS

We enhance an integrated modeling framework to represent the health-related economic impacts of ozone pollution and the related benefits of ozone policy. We quantify how inequality in ozone levels, delayed policy action, and approaches to economic valuation affect the relative economic value of ozone policies with household income. We find that ozone reductions in the Policy Scenario favored reduced mortality risks among low-income households, and that reductions were relatively twice as beneficial for the lowest compared to the highest income households. In attempting to bracket current valuation approaches used in CGE modeling, we find that the valuation approach had a greater impact on the median benefits than the effect of an 11 year policy delay or unequal ozone reductions with income. Empirical and theoretical challenges remain in assessing environmental and health preferences with income in an economy-wide framework. This study demonstrates the potential for differences in relative economic gains across income groups from ozone reductions, and supports including analysis across income groups as a complement to current analysis of environmental policy. It is not meant to identify the actual relationship between income and health-related ozone impacts, but to develop an approach that can explore the effect of ozone reductions under policy with income. As the empirical relationships between ozone, human health, and in come are improved, they can be incorporated into this type of approach to better estimate the effect of policy. Our findings suggest that ozone policies may differentially affect health outcomes with income, and that ignoring these differences could understate the importance of reductions for low-income households.

SUPPLEMENTAL INFORMATION

Regional mortality incidence rates

Within regions, the pattern of ozone-related mortality incidence rates can be increasing, decreasing, or flat with household income. The patterns for four regions are shown below. Finding that this pattern varies across regions is consistent with previous studies, which find differing patterns in different locations.

Incidence Rate of Ozone-Related Acute Mortalities in 2005
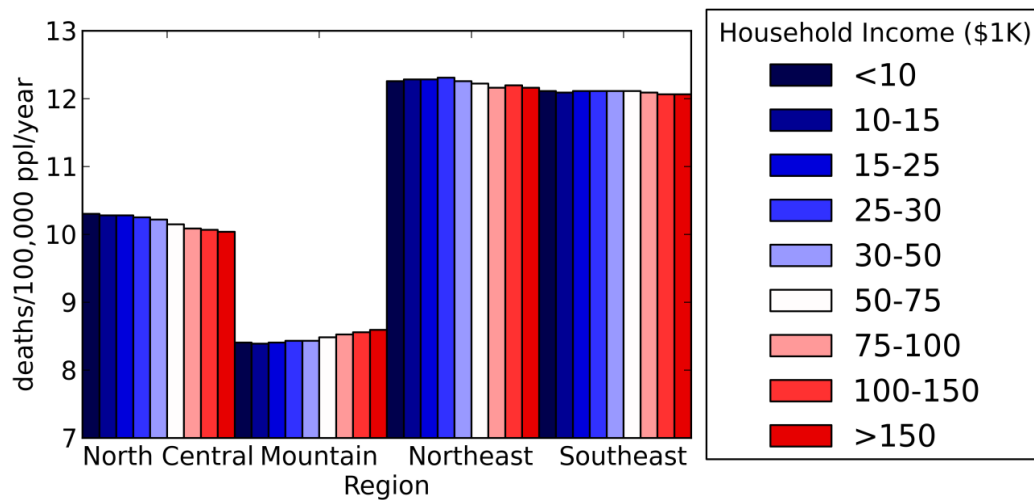
Annual welfare gain over time

The graph below shows the annual economic gain for the 2014 reductions compared to a scenario where we have constant 2005 ozone levels. One can see a steep rise in the annual gain from the policy as it is implemented. Afterwards, the economy continues to gain from clean air at a rate that grows basically at the rate of the rest of the economy. This plot employs the VSL valuation approach. The income-based approach looks similar, but rises from $1 billion in 2005 to $5 billion in 2100.

Annual Welfare Gain Over Time (2006$)
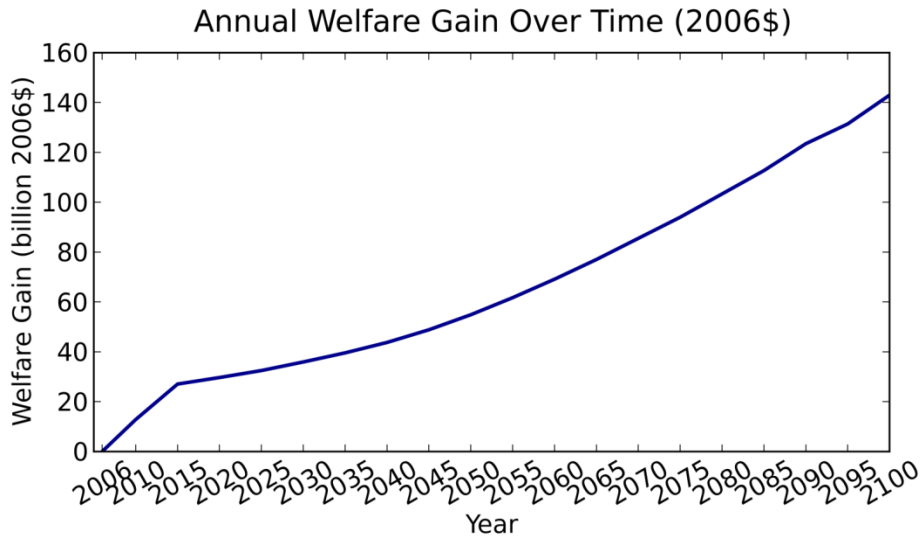
95% CI in welfare gain

The plot below shows the 95% CI in welfare gain for the VSL valuation. This 95%CI is based on 1000 Monte Carlo simulations representing the uncertainty in our pooled concentration-response functions.

Percent Welfare Gain by Household Income Group vs. Do Nothing
2015Feb12WTPMVSLpminc5pdisc7
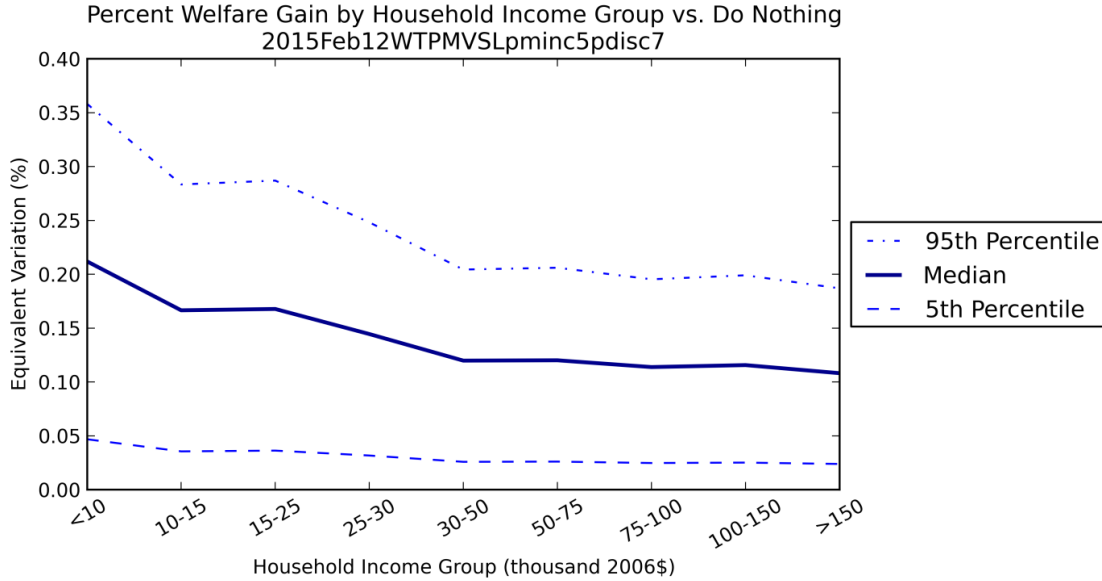
4. REFERENCES

Abt Associates, Inc. 2012. "Environmental Benefits and Mapping Program (Version 4.0), 2012." Research Triangle Park, NC.: U.S. Environmental Protection Agency Office of Air Quality Planning and Standards. http://www.epa.gov/air/benmap.

Adelman, David E. 2013. "The Collective Origins of Toxic Air Pollution: Implications for Greenhouse Gas Trading and Toxic Hotspots." SSRN Scholarly Paper ID 2015375. Rochester, NY: Social Science Research Network. http://papers.ssrn.com/abstract=2015375.

Alberini, Anna, and Milan Ščasný. 2013. "Exploring Heterogeneity in the Value of a Statistical Life: Cause of Death v. Risk Perceptions." *Ecological Economics* 94 (October): 143–55. doi:10.1016/j.ecolecon.2013.07.012.

Arrow, Kenneth J., Partha Dasgupta, Lawrence H. Goulder, Kevin J. Mumford, and Kirsten Oleson. 2013. "Sustainability and the Measurement of Wealth: Further Reflections." *Environment and Development Economics* 18 (04): 504–16. doi:10.1017/S1355770X13000193.

Bellas, Allen S., and Ian Lange. 2008. "Did State Regulation of Power Plants Cause an Increase in Pollution Exportation?" *Economics Letters* 98 (2): 213–19. doi:10.1016/j.econlet.2007.04.032.

Bell, Michelle L., and Francesca Dominici. 2008. "Effect Modification by Community Characteristics on the Short-Term Effects of Ozone Exposure and Mortality in 98 US Communities." *American Journal of Epidemiology* 167 (8): 986–97. doi:10.1093/aje/kwm396.

Bell, Michelle L., Francesca Dominici, and Jonathan M. Samet. 2005. "A Meta-Analysis of Time-Series Studies of Ozone and Mortality With Comparison to the National Morbidity, Mortality, and Air Pollution Study." *Epidemiology (Cambridge, Mass.)* 16 (4): 436.

Bell, Michelle L., Richard Goldberg, Christian Hogrefe, Patrick L. Kinney, Kim Knowlton, Barry Lynn, Joyce Rosenthal, Cynthia Rosenzweig, and Jonathan A. Patz. 2007. "Climate Change, Ambient Ozone, and Health in 50 US Cities." *Climatic Change* 82 (1-2): 61–76. doi:10.1007/s10584-006-9166-7.

Bell, Michelle L., Richard D. Morgenstern, and Winston Harrington. 2011. "Quantifying the Human Health Benefits of Air Pollution Policies: Review of Recent Studies and New Directions in Accountability Research." *Environmental Science & Policy* 14 (4): 357–68. doi:10.1016/j.envsci.2011.02.006.

Bento, Antonio, Matthew Freedman, and Corey Lang. 2014. "Who Benefits from Environmental Regulation? Evidence from the Clean Air Act Amendments." *Review of Economics and Statistics*, November. doi:10.1162/REST_a_00493.

Bergman, Lars. 2005. "CGE Modeling of Environmental Policy and Resource Management." In *Handbook of Environmental Economics*, 1273–1306. Economywide and International Environmental Issues 3. Elsevier.

Berman, Jesse D., Neal Fann, John W. Hollingsworth, Kent E. Pinkerton, William N. Rom, Anthony M. Szema, Patrick N. Breysse, Ronald H. White, and Frank C. Curriero. 2012. "Health Benefits from Large-Scale Ozone Reduction in the United States." *Environmental Health Perspectives* 120 (10): 1404–10. doi:10.1289/ehp.1104851.

Bhattacharyya, Subhes C. 1996. "Applied General Equilibrium Models for Energy Studies: A Survey." *Energy Economics* 18 (3): 145–64. doi:10.1016/0140-9883(96)00013-8.

Bingaman, Jeff, Ron Wyden, Bernard Sanders, Mark Udall, Al Franken, Christopher A. Coons, John Kerry, Sheldon Whitehouse, and Tom Udall. 2012. *Clean Energy Standard Act of 2012*. http://thomas.loc.gov/cgi-bin/bdquery/z?d112:S.2146:

Boyd, Roy, Kerry Krutilla, and W.Kip Viscusi. 1995. "Energy Taxation as a Policy Instrument to Reduce CO2 Emissions: A Net Benefit Analysis." *Journal of Environmental Economics and Management* 29 (1): 1–24. doi:10.1006/jeem.1995.1028.

Buonocore, Jonathan J., Xinyi Dong, John D. Spengler, Joshua S. Fu, and Jonathan I. Levy. 2014. "Using the Community Multiscale Air Quality (CMAQ) Model to Estimate Public Health Impacts of PM2.5 from Individual Power Plants." *Environment International* 68 (July): 200–208. doi:10.1016/j.envint.2014.03.031.

Burtraw, Dallas, Alan Krupnick, Karen Palmer, Anthony Paul, Michael Toman, and Cary Bloyd. 2003. "Ancillary Benefits of Reduced Air Pollution in the US from Moderate Greenhouse Gas Mitigation Policies in the Electricity Sector." *Journal of Environmental Economics and Management* 45 (3): 650–73. doi:10.1016/S0095-0696(02)00022-0.

Burtraw, Dallas, and E. Mansur. 1999. "Environmental Effects of SO2 Trading and Banking." *Environmental Science & Technology* 33 (20): 3489–94.

Carbone, Jared C., and V. Kerry Smith. 2008. "Evaluating Policy Interventions with General Equilibrium Externalities." *Journal of Public Economics* 92 (5–6): 1254–74. doi:10.1016/j.jpubeco.2007.06.003.

———. 2013. "Valuing Nature in General Equilibrium." *Journal of Environmental Economics and Management* 66 (1): 72–89.

Caron, Justin, Gilbert E. Metcalf, and John M. Reilly. 2015. "The General Equilibrium Implications of the EPA's Proposed Clean Power Plan." *(in Prep)*.

Caron, Justin, Sebastian Rausch, and Niven Winchester. 2015. "Leakage from Sub-National Climate Policy: The Case of California's Cap-and-Trade Program." *Energy Journal* 36 (2). doi:10.5547/01956574.36.2.8.

Chang, Howard H., Jingwen Zhou, and Montserrat Fuentes. 2010. "Impact of Climate Change on Ambient Ozone Level and Mortality in Southeastern United States." *International Journal of Environmental Research and Public Health* 7 (7): 2866–80. doi:10.3390/ijerph7072866.

Chemingui, Mohamed Abdelbasset, and Chokri Thabet. 2014. "Taxing CO2 Emissions and Its Ancillary Health Benefits: A Computable General Equilibrium Analysis for Tunisia." *Middle East Development Journal* 6 (1): 108–45. doi:10.1080/17938120.2014.887805.

CMAS. 2010. "SMOKE v2.7 User's Manual." Chapel Hill, NC: Institute for the Environment, The University of North Carolina at Chapel Hill. http://cmascenter.org/smoke/documentation/3.5/manual_smokev35.pdf.

Cropper, Maureen, James K. Hammitt, and Lisa A. Robinson. 2011. "Valuing Mortality Risk Reductions: Progress and Challenges." *Annual Review of Resource Economics* 3 (1): 313–36. doi:10.1146/annurev.resource.012809.103949.

Dowlatabadi, Hadi. 1993. "Estimating the Ancillary Benefits of Selected Carbon Dioxide Mitigation Strategies: Electricity Sector." Climate Change Division, US Environmental Protection Agency.

Driscoll, Charles, Jonathan Buonocore, Habibollah Fakhraei, and Kathy Fallon Lambert. 2014. "Part 1: Air Pollution Changes under Different 111d Options for Existing Power Plants." Co-Benefits of Carbon Standards. http://eng-cs.syr.edu/wp-content/uploads/2014/07/Carbon-cobenefits-study-6-13-14.pdf.

Environ International Corporation. 2013. "CAMx User's Guide Version 5.4." Novato, California. http://www.camx.com/files/camxusersguide_v5-40.aspx.

Fann, Neal, Kirk R. Baker, and Charles M. Fulcher. 2012. "Characterizing the PM2.5-Related Health Benefits of Emission Reductions for 17 Industrial, Area and Mobile Emission Sectors across the U.S." *Environment International* 49 (November): 141–51. doi:10.1016/j.envint.2012.08.017.

Fann, Neal, Charles M. Fulcher, and Kirk Baker. 2013. "The Recent and Future Health Burden of Air Pollution Apportioned Across U.S. Sectors." *Environmental Science & Technology*, March. doi:10.1021/es304831q.

Fann, Neal, Charles M. Fulcher, and Bryan J. Hubbell. 2009. "The Influence of Location, Source, and Emission Type in Estimates of the Human Health Benefits of Reducing a Ton of Air Pollution." *Air Quality, Atmosphere, & Health* 2 (3): 169–76. doi:10.1007/s11869-009-0044-0.

Fann, Neal, Henry A. Roman, Charles M. Fulcher, Mikael A. Gentile, Bryan J. Hubbell, Karen Wesson, and Jonathan I. Levy. 2011. "Maximizing Health Benefits and Minimizing Inequality: Incorporating Local-Scale Data in the Design and Evaluation of Air Quality Policies." *Risk Analysis* 31 (6): 908–22. doi:10.1111/j.1539-6924.2011.01629.x.

Fraas, Arthur G. 2011. "The Treatment of Uncertainty in EPA's Analysis of Air Pollution Rules: A Status Report." *Journal of Benefit-Cost Analysis* 2 (2): 1–27.

Giang, Amanda (Amanda Chi Wen). 2013. "Quantifying the Health and Economic Impacts of Mercury : An Integrated Assessment Approach." Thesis, Massachusetts Institute of Technology. http://hdl.handle.net/1721.1/81115.

Goddard, Maria, and Peter Smith. 2001. "Equity of Access to Health Care Services:: Theory and Evidence from the UK." *Social Science & Medicine* 53 (9): 1149–62. doi:10.1016/S0277-9536(00)00415-9.

Goulder, Lawrence H. 1993. "Economy-Wide Emissions Impacts of Alternative Energy Tax Proposals." Climate Change Division, US Environmental Protection Agency.

Goulder, Lawrence H., Ian W. H. Parry, and Dallas Burtraw. 1997. "Revenue-Raising versus Other Approaches to Environmental Protection: The Critical Significance of Preexisting Tax Distortions." *The RAND Journal of Economics* 28 (4): 708–31. doi:10.2307/2555783.

Goulder, Lawrence H., and Roberton C. Williams. 2003. "The Substantial Bias from Ignoring General Equilibrium Effects in Estimating Excess Burden, and a Practical Solution." *Journal of Political Economy* 111 (4): 898–927. doi:10.1086/375378.

Graff Zivin, Joshua S., Matthew J. Kotchen, and Erin T. Mansur. 2014. "Spatial and Temporal Heterogeneity of Marginal Emissions: Implications for Electric Cars and Other Electricity-Shifting Policies." *Journal of Economic Behavior & Organization* 107, Part A (November): 248–68. doi:10.1016/j.jebo.2014.03.010.

Grell, George A., Jimmy Dudhia, and David R. Stauffer. 1994. "A Description of the Fifth-Generation Penn State/NCAR Mesoscale Model (MM5)." Tech. Note NCAR/TN-398+STR. Boulder, CO: National Center for Atmospheric Research.

Grineski, Sara, Bob Bolin, and Christopher Boone. 2007. "Criteria Air Pollution and Marginalized Populations: Environmental Inequity in Metropolitan Phoenix, Arizona*." *Social Science Quarterly* 88 (2): 535–54. doi:10.1111/j.1540-6237.2007.00470.x.

Hazilla, Michael, and Raymond J. Kopp. 1990. "Social Cost of Environmental Quality Regulations: A General Equilibrium Analysis." *Journal of Political Economy* 98 (4): 853–73.

Henry, David D., Nicholas Z. Muller, and Robert O. Mendelsohn. 2011. "The Social Cost of Trading: Measuring the Increased Damages from Sulfur Dioxide Trading in the United States." *Journal of Policy Analysis and Management* 30 (3): 598–612. doi:10.1002/pam.20584.

Holmes, Rebecca, Doug Keinath, and Fran Sussman. 1995. "Ancillary Benefits of Mitigating Climate Change: Selected Actions from the Climate Change Action Plan." Contract No. 68-W2-0018, ICF Incorporated. Washington, D.C.: Final Report prepare for Adaptation Branch, Climate Change Division, Office of Policy, Planning and Evaluation, U.S. Environmental Protection Agency.

Holt, Jareth, Noelle E. Selin, and Susan Solomon. 2015. "Changes in Inorganic Fine Particulate Matter Sensitivities to Precursors Due to Large-Scale US Emissions Reductions." *Environmental Science & Technology* 49 (8): 4834–41. doi:10.1021/acs.est.5b00008.

Jack, Darby W, and Patrick L Kinney. 2010. "Health Co-Benefits of Climate Mitigation in Urban Areas." *Current Opinion in Environmental Sustainability* 2 (3): 172–77. doi:10.1016/j.cosust.2010.06.007.

Jones, Andrew M., Nigel Rice, Silvana Robone, and Pedro Rosa Dias. 2011. "Inequality and Polarisation in Health Systems' Responsiveness: A Cross-Country Analysis." *Journal of Health Economics* 30 (4): 616–25. doi:10.1016/j.jhealeco.2011.05.003.

Kates, Robert W., William C. Clark, Robert Corell, J. Michael Hall, Carlo C. Jaeger, Ian Lowe, James J. McCarthy, et al. 2001. "Sustainability Science." *Science* 292 (5517): 641–42. doi:10.1126/science.1059386.

Knowlton, Kim, Joyce E. Rosenthal, Christian Hogrefe, Barry Lynn, Stuart Gaffin, Richard Goldberg, Cynthia Rosenzweig, Kevin Civerolo, Jia-Yeong Ku, and Patrick L. Kinney. 2004. "Assessing Ozone-Related Health Impacts under a Changing Climate." *Environmental Health Perspectives* 112 (15): 1557–63.

Lanz, Bruno, and Sebastian Rausch. 2011. "General Equilibrium, Electricity Generation Technologies and the Cost of Carbon Abatement: A Structural Sensitivity Analysis." *Energy Economics* 33 (5): 1035–47. doi:10.1016/j.eneco.2011.06.003.

Larsen, John. 2014. "Remaking American Power: Potential Energy Market Impacts of EPA's Proposed GHG Emission Performance Standards for Existing Electric Power Plants." Washington, DC: Center for Strategic and International Studies. http://rhg.com/wp-content/uploads/2014/11/RemakingAmericanPower.pdf.

Liu, Feng. 1996. "Urban Ozone Plumes and Population Distribution by Income and Race: A Case Study of New York and Philadelphia." *Journal of the Air & Waste Management Association (1995)* 46 (3): 207–15.

———. 1998. "Who Will Be Protected by EPA's New Ozone and Particulate Matter Standards?" *Environmental Science & Technology* 32 (1): 32A – 39A. doi:10.1021/es9833624.

Marmot, Michael. 2007. "Achieving Health Equity: From Root Causes to Fair Outcomes." *The Lancet* 370 (9593): 1153–63. doi:10.1016/S0140-6736(07)61385-3.

Marshall, Julian D. 2008. "Environmental Inequality: Air Pollution Exposures in California's South Coast Air Basin." *Atmospheric Environment* 42 (21): 5499–5503. doi:10.1016/j.atmosenv.2008.02.005.

Matus, Kira, Kyung-Min Nam, Noelle E. Selin, Lok N. Lamsal, John M. Reilly, and Sergey Paltsev. 2012. "Health Damages from Air Pollution in China." *Global Environmental Change* 22 (1): 55–66. doi:10.1016/j.gloenvcha.2011.08.006.

Matus, Kira, Trent Yang, Sergey Paltsev, John Reilly, and Kyung-Min Nam. 2008. "Toward Integrated Assessment of Environmental Change: Air Pollution Health Effects in the USA." *Climatic Change* 88 (1): 59–92.

Mauzerall, Denise L., Babar Sultan, Namsoug Kim, and David F. Bradford. 2005. "NOx Emissions from Large Point Sources: Variability in Ozone Production, Resulting Health Damages and Economic Costs." *Atmospheric Environment* 39 (16): 2851–66. doi:10.1016/j.atmosenv.2004.12.041.

Mesbah, S. Morteza, Amir Hakami, and Stephan Schott. 2013. "Optimal Ozone Reduction Policy Design Using Adjoint-Based NOx Marginal Damage Information." *Environmental Science & Technology* 47 (23): 13528–35. doi:10.1021/es402531n.

———. 2014. "Targeted NOx Emissions Control for Improved Environmental Performance." In *Air Pollution Modeling and Its Application XXII*, edited by Douw G. Steyn, Peter J. H. Builtjes, and Renske M. A. Timmermans, 35–40. NATO Science for Peace and Security Series C: Environmental Security. Springer Netherlands. http://link.springer.com/chapter/10.1007/978-94-007-5577-2_6.

Miranda, Marie Lynn, Sharon E. Edwards, Martha H. Keating, and Christopher J. Paul. 2011. "Making the Environmental Justice Grade: The Relative Burden of Air Pollution Exposure in the United States." *International Journal of Environmental Research and Public Health* 8 (6): 1755–71. doi:10.3390/ijerph8061755.

Muller, Nicholas Z. 2012. "The Design of Optimal Climate Policy with Air Pollution Co-Benefits." *Resource and Energy Economics* 34 (4): 696–722. doi:10.1016/j.reseneeco.2012.07.002.

Muller, Nicholas Z., and Robert Mendelsohn. 2009. "Efficient Pollution Regulation: Getting the Prices Right." *The American Economic Review* 99 (5): 1714–39.

Nam, Kyung-Min, Noelle E. Selin, John M. Reilly, and Sergey Paltsev. 2010. "Measuring Welfare Loss Caused by Air Pollution in Europe: A CGE Analysis." *Energy Policy* 38 (9): 5059–71.

Nam, Kyung-Min, Caleb J. Waugh, Sergey Paltsev, John M. Reilly, and Valerie J. Karplus. 2013. "Carbon Co-Benefits of Tighter SO2 and NOx Regulations in China." *Global Environmental Change*, October. doi:10.1016/j.gloenvcha.2013.09.003.

Nemet, G. F., T. Holloway, and P. Meier. 2010. "Implications of Incorporating Air-Quality Co-Benefits into Climate Change Policymaking." *Environmental Research Letters* 5 (1): 014007. doi:10.1088/1748-9326/5/1/014007.

"OIRA Reports to Congress | The White House." 2013. Accessed May 31. http://www.whitehouse.gov/omb/inforeg_regpol_reports_congress.

Paltsev, Sergey, and Pantelis Capros. 2013. "COST CONCEPTS FOR CLIMATE CHANGE MITIGATION." *Climate Change Economics* 04 (supp01): 1340003. doi:10.1142/S2010007813400034.

Penrod, Ashley, Yang Zhang, Kai Wang, Shiang-Yuh Wu, and L. Ruby Leung. 2014. "Impacts of Future Climate and Emission Changes on U.S. Air Quality." *Atmospheric Environment*. Accessed January 21. doi:10.1016/j.atmosenv.2014.01.001.

Piketty, Thomas, and Emmanuel Saez. 2013. "Top Incomes and the Great Recession: Recent Evolutions and Policy Implications." *IMF Economic Review* 61 (3): 456–78. doi:10.1057/imfer.2013.14.

Rausch, Sebastian, and V. J. Karplus. 2014. "Markets versus Regulation: The Efficiency and Distributional Impacts of U.S. Climate Policy Proposals." Technical Report. MIT Joint Program on the Science and Policy of Global Change. http://dspace.mit.edu/handle/1721.1/91460.

Rausch, Sebastian, Gilbert E. Metcalf, and John M. Reilly. 2011. "Distributional Impacts of Carbon Pricing: A General Equilibrium Approach with Micro-Data for Households." *Energy Economics* 33, Supplement 1 (December): S20–33. doi:10.1016/j.eneco.2011.07.023.

Rausch, Sebastian, Gilbert E. Metcalf, John M. Reilly, and Sergey Paltsev. 2010. "Distributional Impacts of a U.S. Greenhouse Gas Policy: A General Equilibrium Analysis of Carbon Pricing." In *U.S. Energy Tax Policy*, edited by Gilbert E. Metcalf. Cambridge University Press.

Rausch, Sebastian, and Matthew Mowers. 2014. "Distributional and Efficiency Impacts of Clean and Renewable Energy Standards for Electricity." *Resource and Energy Economics* 36 (2): 556–85. doi:10.1016/j.reseneeco.2013.09.001.

Rausch, Sebastian, and John Reilly. 2012. "Carbon Tax Revenue and the Budget Deficit: A Win-Win-Win Solution?" Joint Program Report Series. http://globalchange.mit.edu/files/document/MITJPSPGC_Rpt228.pdf.

Ravishankara, A.R., John P. Dawson, and Darrell A. Winner. 2012. "New Directions: Adapting Air Quality Management to Climate Change: A Must for Planning." *Atmospheric Environment* 50 (April): 387–89. doi:10.1016/j.atmosenv.2011.12.048.

Rowe, Robert, and et al. 1995. "The New York State Externalities Cost Study." Oceana Publications. Dobbs Ferry, New York: Hagler Bailly Consulting, Inc.

Rudokas, Jason, Paul J. Miller, Marcus Trail, and Armistead G. Russell. 2015. "Regional Air Quality Management Aspects of Climate Change: Impact of Climate Mitigation Options on Regional Air Emissions." *Environmental Science & Technology*, March. doi:10.1021/es505159z.

Saari, Rebecca K., Noelle E. Selin, Sebastian Rausch, and Tammy M. Thompson. 2015. "A Self-Consistent Method to Assess Air Quality Co-Benefits from U.S. Climate Policies." *Journal of the Air & Waste Management Association* 65 (1): 74–89. doi:10.1080/10962247.2014.959139.

Scheraga, Joel D., and Neal A. Leary. 1994. "Costs and Side Benefits of Using Energy Taxes to Mitigate Global Climate Change." In *Proceedings of the 86th Annual Conference of the National Tax Association*. Columbus, OH,.

Schoen, C., D.C. Radley, P. Riley, J.A. Lippa, J. Berenson, C. Dermody, and A. Shih. 2013. "Health Care in the Two Americas: Findings from the Scorecard on State Health System Performance for Low-Income Populations." http://www.commonwealthfund.org/publications/fund-reports/2013/sep/low-income-scorecard.

Schwartz, Joel. 2005. "How Sensitive Is the Association between Ozone and Daily Deaths to Control for Temperature?" *American Journal of Respiratory and Critical Care Medicine* 171 (6): 627–31. doi:10.1164/rccm.200407-933OC.

Selin, N E, S Wu, K M Nam, J M Reilly, S Paltsev, R G Prinn, and M D Webster. 2009. "Global Health and Economic Impacts of Future Ozone Pollution." *Environmental Research Letters* 4 (4): 044014. doi:10.1088/1748-9326/4/4/044014.

Shadbegian, R, W Gray, and C Morgan. 2007. "Benefits and Costs from Sulfur Dioxide Trading: A Distributional Analysis." In *Acid in the Environment*, edited by Gerald R. Visgilio and Diana M. Whitelaw, 241–59. New York: Springer Science Business Media, LLC.

Sieg, Holger, V. Kerry Smith, H. Spencer Banzhaf, and Randy Walsh. 2004. "Estimating the General Equilibrium Benefits of Large Changes in Spatially Delineated Public Goods*." *International Economic Review* 45 (4): 1047–77. doi:10.1111/j.0020-6598.2004.00297.x.

Smith, V. Kerry, and Jared C. Carbone. 2007. "Should Benefit–Cost Analyses Take Account of General Equilibrium Effects?" *Research in Law and Economics* 23 (October): 247–72. doi:10.1016/S0193-5895(07)23011-8.

Spencer Banzhaf, H., Dallas Burtraw, and Karen Palmer. 2004. "Efficient Emission Fees in the US Electricity Sector." *Resource and Energy Economics* 26 (3): 317–41. doi:10.1016/j.reseneeco.2003.10.002.

Sue Wing, Ian. 2009. "Computable General Equilibrium   Models for the Analysis of Energy and Climate Policies." In *International Handbook on The Economics Of Energy*, edited by J. Evans and L.C. Hunt, 332–66.

Thompson, Tammy M., Sebastian Rausch, Rebecca K. Saari, and Noelle E. Selin. 2014. "A Systems Approach to Evaluating the Air Quality Co-Benefits of US Carbon Policies." *Nature Climate Change* 4 (10): 917–23. doi:10.1038/nclimate2342.

Thompson, Tammy M., Rebecca K. Saari, and Noelle E. Selin. 2014. "Air Quality Resolution for Health Impact Assessment: Influence of Regional Characteristics." *Atmos. Chem. Phys.* 14 (2): 969–78. doi:10.5194/acp-14-969-2014.

Thompson, Tammy M., and Noelle E. Selin. 2012. "Influence of Air Quality Model Resolution on Uncertainty Associated with Health Impacts." *Atmos. Chem. Phys.* 12 (20): 9753–62. doi:10.5194/acp-12-9753-2012.

Tietenberg, Tom. 1995. "Tradeable Permits for Pollution Control When Emission Location Matters: What Have We Learned?" *Environmental and Resource Economics* 5 (2): 95–113. doi:10.1007/BF00693018.

Tong, Daniel Q., Nicholas Z. Muller, Denise L. Mauzerall, and Robert O. Mendelsohn. 2006. "Integrated Assessment of the Spatial Variability of Ozone Impacts from Emissions of Nitrogen Oxides." *Environmental Science & Technology* 40 (5): 1395–1400.

Tra, Constant I. 2010. "A Discrete Choice Equilibrium Approach to Valuing Large Environmental Changes." *Journal of Public Economics* 94 (1–2): 183–96. doi:10.1016/j.jpubeco.2009.10.006.

———. 2013. "Measuring the General Equilibrium Benefits of Air Quality Regulation in Small Urban Areas." *Land Economics* 89 (2): 291–307.

U.S. Environmental Protection Agency. 2007. "Guidance on the Use of Models and Other Analyses for Demonstrating Attainment of Air Quality Goals for Ozone, PM2.5, and Regional Haze." Research Triangle Park, NC 27711. http://www.epa.gov/scram001/guidance/guide/final-03-pm-rh-guidance.pdf.

———. 2011a. "The Benefits and Costs of the Clean Air Act from 1990 to 2020." U.S. Environmental Protection Agency.

———. 2011b. "Air Quality Modeling Final Rule Technical Support Document." Research Triangle Park, North Carolina 27711: Office of Air Quality Planning and Standards. http://www.epa.gov/airtransport/pdfs/AQModeling.pdf.

———. 2012. "Regulatory Impact Analysis for the Final Revisions to the National 1 Ambient Air 2 Quality Standards for Particulate Matter, Office of Air Quality Planning and Standards." EPA-452/R-12-005. Research Triangle Park, NC 27711. http://www.epa.gov/ttnecas1/regdata/RIAs/finalria.pdf.

———. 2014a. "Regulatory Impact Analysis for the Proposed Carbon Pollution Guidelines for Existing Power Plants and Emission Standards for Modified and Reconstructed Power Plants." EPA-452/R-14-002. Research Triangle Park, North Carolina 27711: Office of Air Quality Planning and Standards. http://www2.epa.gov/sites/production/files/2014-06/documents/20140602ria-clean-power-plan.pdf.

———. 2015a. "Cross-State Air Pollution Rule (CSAPR)." April 20. http://www.epa.gov/airtransport/CSAPR/.

———. 2015b. "National Ambient Air Quality Standards for Ozone." EPA-HQ-OAR-2008-0699; FRL-9918-43-OAR. Research Triangle Park, North Carolina 27711: Office of Air Quality Planning and Standards. http://www.epa.gov/airquality/ozonepollution/pdfs/20141125proposal.pdf.

U.S. Environmental Protection Agency, (first). 2014b. "Guidelines for Preparing Economic Analyses." hhttp://yosemite.epa.gov/ee/epa/eerm.nsf/vwAN/EE-0568-50.pdf/$file/EE-0568-50.pdf.

Van Doorslaer, Eddy, Adam Wagstaff, Hattem van der Burg, Terkel Christiansen, Diana De Graeve, Inge Duchesne, Ulf-G Gerdtham, et al. 2000. "Equity in the Delivery of Health Care in Europe and the US." *Journal of Health Economics* 19 (5): 553–83. doi:10.1016/S0167-6296(00)00050-3.

Van Kippersluis, Hans, and Titus J. Galama. 2014. "Wealth and Health Behavior: Testing the Concept of a Health Cost." *European Economic Review* 72 (November): 197–220. doi:10.1016/j.euroecorev.2014.10.003.

Van Ourti, Tom, Eddy van Doorslaer, and Xander Koolman. 2009. "The Effect of Income Growth and Inequality on Health Inequality: Theory and Empirical Evidence from the European Panel." *Journal of Health Economics* 28 (3): 525–39. doi:10.1016/j.jhealeco.2008.12.005.

Viscusi, W. Kip. 2009. "The Devaluation of Life." *Regulation & Governance* 3 (2): 103–27. doi:10.1111/j.1748-5991.2009.01052.x.

Viscusi, W. Kip, and Joseph E. Aldy. 2003. "The Value of a Statistical Life: A Critical Review of Market Estimates Throughout the World." *Journal of Risk and Uncertainty* 27 (1): 5–76. doi:10.1023/A:1025598106257.

West, J. Jason, Sophie Szopa, and Didier A. Hauglustaine. 2007. "Human Mortality Effects of Future Concentrations of Tropospheric Ozone." *Comptes Rendus Geoscience*, Impact du changement climatique global sur la qualité de l'air à l'échelle régionale, 339 (11–12): 775–83. doi:10.1016/j.crte.2007.08.005.

Wilper, Andrew P., Steffie Woolhandler, Karen E. Lasser, Danny McCormick, David H. Bor, and David U. Himmelstein. 2009. "Health Insurance and Mortality in US Adults." *American Journal of Public Health* 99 (12): 2289–95. doi:10.2105/AJPH.2008.157685.

Yang, Trent. 2004. "Economic and Policy Implications of Urban Air Pollution in the United States, 1970 to 2000." Thesis, Massachusetts Institute of Technology. http://dspace.mit.edu/handle/1721.1/28315.

Zapata, Christina, Nicholas Muller, and Michael J. Kleeman. 2013. "PM2.5 Co-Benefits of Climate Change Legislation Part 1: California's AB 32." *Climatic Change* 117 (1-2): 377–97. doi:10.1007/s10584-012-0545-y.