Air Quality Co-benefits of Renewable Energy Policy in the U.S.

by

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Submitted to the Institute for Data, Systems, and Society
in partial fulfillment of the requirements for the degree of

Master of Science in Technology and Policy

at the

MASSACHUSETTS INSTITUTE OF TECHNOLOGY

September 2018

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Abstract

Despite lawmaker interest in transitioning electricity systems toward renewable energy sources and in mitigating harmful air pollution, the extent to which sub-national renewable energy policies in the U.S. can improve air quality and human health remains unclear. This thesis develops a systemic modeling framework to assess the impacts of future renewable energy policy on air quality, as well as on the economy and on climate change, employing the framework of cost-benefit analysis. To model the chain of policy effects from impacts on the economy to power plant emissions, human health, and climate change, I integrate an economy-wide computable general equilibrium model, an atmospheric chemistry model, and methodologies for the economic valuation of health impacts. I apply this modeling framework to study the potential future impacts of the existing Renewable Portfolio Standards (RPSs) in the U.S. Rust Belt region. This thesis also tests the impacts of alternative RPS stringency levels and assesses RPS impacts compared to carbon pricing, a climate policy favored by many economists.

I estimate that existing RPSs in this region generate health co-benefits that, in economic terms, exceed the climate change mitigation benefits of these policies. Estimated health co-benefits also outweigh the economic costs of the modeled policies, indicating that air quality co-benefits alone may justify RPS implementation. This work further finds that raising RPS stringency in the Rust Belt increases net policy benefits (air quality and climate benefits minus costs). However, I show that air quality co-benefits are highly sensitive to several assumptions such as the economic value assigned to premature mortalities and the magnitude of the health response expected from a given level of pollution. This thesis also estimates that carbon pricing generates greater air quality co-benefits for every ton of CO_2 abated compared to an RPS, suggesting that carbon pricing may be more economically efficient (greater net benefits) relative to an RPS than previously thought. Finally, I show that RPSs have far-reaching economic impacts that have implications for their overall costs and benefits. This finding demonstrates the value of employing economy-wide models to understand the overall economic and environmental impacts of such sector-specific policies, and makes the case for a comprehensive, economy-wide approach for addressing air pollution and climate change.

Thesis Supervisor: Noelle E. Selin
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Acknowledgements

This thesis would not have been possible without the previous work of numerable scholars. I would like to acknowledge Sebastian Rausch for developing the economic model used in this work. Christopher Tessum also deserves recognition for creating the air pollution model used in this thesis and for his unfailing readiness to help. I am also grateful to Daniel Rothenberg and Mei Yuan for their patient assistance with programming challenges.

I am particularly thankful to Noelle Selin who gave me this opportunity and provided me with frequent advice and astute feedback at all stages of this thesis, and to Sergey Paltsev who generously offered critical advice on computable general equilibrium modeling.

I also feel fortunate to have been part of the stimulating intellectual environments of the Technology and Policy Program and the Joint Program. I am grateful to fellow students for their patient listening and helpful feedback.

Finally, I would like to thank my wife Annie for agreeing to move to Boston with me and for steadfastly supporting me through these years.

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Chapter 1

Introduction

1.1 Problem introduction

Air pollution is an enduring problem in the U.S. Although it has been alleviated by policy responses, such as the Clean Air Act, and market forces, such as the increasing competitiveness of natural gas vis-à-vis coal, air pollution continues to harm human health. In 2016, ambient concentrations of $PM_{2.5}$ were associated with just over 93,000 premature deaths, resulting in approximately 1,556,000 years of life lost (IHME, 2016).

The problem of air pollution shares commonalities with the problem of climate change. Air pollutants and CO_2 are both byproducts of fossil fuel consumption. This overlap has led researchers to consider whether reducing CO_2 emissions could, as a side effect, lead to decreases in air pollutant emissions, and as a result, cleaner air and improved human health. Previous studies have found that the clean air related health co-benefits of climate policy can be considerable (Nemet, Holloway and Meier, 2010), significant enough to exceed policy costs (Thompson *et al.*, 2014). Therefore, accounting for air quality effects is an important step toward understanding the impacts of climate policy.

In the U.S., climate change mitigation is currently driven to a large degree by state-level lawmaking due to a lack of political willingness at the federal level. For lawmakers at the state level, a variety of possible policies exist but renewable energy instruments have emerged as particularly popular (Leiserowitz et al., 2018). In this category, Renewable Portfolio Standards (RPSs) are among the most prevalent renewable support policies (Carley and Miller, 2012). An RPS requires electricity suppliers to source a given percent of electricity from renewable power generating technologies. Such policies exist in 29 states and the District of Columbia. Across these jurisdictions, RPSs are the subject of frequent debates. State legislatures deliberated on a total of 181 RPS bills throughout the 2016 and early 2017 legislative sessions (Barbose, 2017).

Future lawmaking in this area calls for an understanding of the costs and benefits of state RPSs. While not sufficient for the purposes of policy making, a cost-benefit analysis can aid decision-making by providing a systematic way of weighing different policy effects and comparing alternative policy designs (Arrow et al., 1996; Viscusi, Vernon and Harrington, 2005). This analytical framework further enables the comparison of RPSs to alternative policies. Although some researchers have critiqued the use of such analyses on practical and moral grounds (Ackerman and Heinzerling, 2002), they remain a fixture in U.S. policy making. Federal agencies conduct them under the direction of executive order (Graham, 2007). State legislatures have also requested them specifically in regard to RPSs (Heeter et al., 2014).

Previous work on the costs and benefits of RPSs has only rarely addressed potential health co-benefits. A meta-analysis of state-commissioned legislative assessments found only one estimate of health benefits made by a Delaware utility (Heeter *et al.*, 2014). Peer-reviewed literature has studied air quality co-benefits using a variety of approaches including retrospective and prospective modeling, as well as statistical methods, but has focused on aggregate or average impacts across all U.S. RPSs (Eastin, 2014; Mai *et al.*, 2016; Wiser *et al.*, 2016).

Due to the national scope of these analyses, it is not well understood how specific state RPS policies may impact local air quality or how such impacts factor in their overall costs and benefits. The effects of sub-national policies can differ substantially from national averages as marginal damages of pollution vary by source and location (Tietenberg, 1995; Siler-evans *et al.*, 2013; Saari *et al.*, 2015). The location of damages may also differ from the location of emission sources. The transboundary nature of air pollutants implies that the benefits of emission reductions in one state may lower pollution concentrations in a downwind neighbor. Therefore, an assessment of local costs and benefits requires a sub-national modeling approach. The previous peer-reviewed literature includes two sub-national modeling studies undertaken for California and Colorado (Rouhani *et al.*, 2016; Hannum *et al.*, 2017).

Another challenge concerning RPS evaluation is the quantification of costs. Modeling studies have most commonly focused on estimating electricity system costs (Mai *et al.*, 2016; Rouhani *et al.*, 2016; Wiser *et al.*, 2016), thus leaving out considerations of the ripple effects that such policies can have beyond the electricity sector. Sector-specific cost estimates may significantly underestimate the social costs of air pollution abatement (Hazilla and Kopp, 1990; Goulder, Parry and Burtraw, 1996). An alternative approach is the use general equilibrium approaches, which quantify economy-wide policy impacts (Thompson *et al.*, 2014, Saari *et al.*, 2015, Hannum *et al.*, 2017). To the author's knowledge, Hannum *et al.* (2017) represents the only sub-national RPS study to quantify health co-benefits and total economic costs.

It can also be relevant to understand how RPSs compare to other climate policies. Economists often recommend carbon pricing as the most cost-effective mitigation policy (Pigou, 1932; Stern, 2006; Stiglitz *et al.*, 2017). Modeling by Rausch and Mowers (2014) showed that a carbon price reduces CO_2 emissions at a lower cost than an RPS. However, studies that account for air quality effects found that factoring in such co-benefits alters the relative cost-effectiveness of carbon pricing relative to other (non-RPS) policies (Boyce and Pastor 2013, Thompson *et al.*, 2014, Driscoll *et al.*, 2015, Knittel and Sandler 2011). This strand of the literature raises the question of how RPS policies compare to carbon pricing once air quality co-benefits are considered.

1.2 Research questions

This thesis builds on this literature by presenting a modeling assessment of the air quality co-benefits of sub-national RPSs, as well as their climate benefits and economic costs. This work focuses on the region comprising the following states: Pennsylvania, Ohio, Wisconsin, Michigan, Illinois, Indiana, West Virginia, New Jersey, Maryland, and Delaware. This regional definition, which spans parts of the American Midwest, Rust Belt, and Mid-Atlantic regions, is based on the spatial aggregation of the economic model used in this thesis. For simplicity, the rest of this thesis refers to this region as the Rust Belt.

Information about RPS impacts may be particularly relevant in this region. RPS bills frequently feature on the legislative agenda of individual states, proposing to roll back, strengthen, or otherwise modify current policies (Colorado State University, 2018). The decisions taken on such bills may have important implications for human health due to the relative severity of air pollution in this region (Caiazzo *et al.*, 2013; Jaramillo and Muller, 2016).

This work takes a prospective approach and explores projected future implications of RPSs. I compare projected costs and benefits across a number of policy scenarios informed by the current state of RPS debates. The scenarios are designed to assess the effects of existing RPSs in the Rust Belt region, the effects of alternative RPS stringency levels, and the effects of RPSs relative to carbon pricing. The specific research questions addressed in this thesis are outlined below:

1. How do projected 2030 health and climate mitigation benefits of the currently implemented RPSs in the Rust Belt region compare to their total economic cost?
2. How do projected costs and benefits vary with policy stringency?
3. How do the projected costs and benefits differ between RPSs and carbon pricing?

1.3 Approach and structure

To address these questions, I develop a modeling framework that integrates a socioeconomic general equilibrium model and an air quality model. I use the economic model - the United States Regional Energy Policy (USREP) model – to explore how RPSs influence economic variables such as fuel use, production, consumption, the make-up of the electricity mix, and CO_2 emissions. The air quality model – the Intervention Model for Air Pollution (InMAP) – simulates how changes in air pollutant emissions influence concentrations of $PM_{2.5}$ and the resulting amount of premature mortalities. I further quantify the climate mitigation and health benefits in monetary terms for the purpose of cost-benefit analysis. These policy impacts are modeled for a number of policy scenarios as well as number of modeling cases representing different modeling assumptions.

The remainder of this paper is organized as follows. Chapter 2 introduces the characteristics of the air pollution effects of power generation in the U.S. and the Rust Belt region and discusses the link between the problems of air pollution and climate change. It goes on to outline policy approaches and describes the design of RPS policies in the U.S. The chapter then provides a review of the literature quantifying the costs and benefits of climate policy in general and RPS in particular, with a focus on the climate and air quality co-benefits and economic costs.

Chapter 3 describes the modeling framework developed to simulate the chain of policy impacts from changes in the economy to emissions, pollution concentrations, avoided mortalities, and resulting economic benefits of avoiding mortalities. It provides an overview of the USREP model, with an emphasis on how the electricity system is represented and the modeling of RPS impacts. The chapter then details how economic impacts are translated into impacts on air pollutant emissions, how InMAP is used to model pollution concentrations and mortalities, and how mortalities are translated into monetary impacts. It also discusses the quantification of climate benefits and closes with a description of the modeled policy scenarios. The assumptions presented in this chapter represent Base Case modeling assumptions (the sensitivity of results to alternative assumptions are discussed in Chapter 4).

Chapter 4 presents the modeling results for the chosen policy scenarios. It further includes results for all policy scenarios based on alternative modeling cases using assumptions alternative to the Base Case to test the robustness of the presented results. The chapter closes with a discussion that places the results of this thesis in the context of the previous literature.

Finally, Chapter 5 discusses policy implications for the costs and benefits of RPSs, the relative merits of RPSs relative to carbon pricing, the political economy of climate policy, and the evaluation of climate policy. The chapter concludes with a discussion of the limitations of this study and how they may be addressed by future work.

Chapter 2

Power generation and human health

Power generation affects human health through a chain of events that lead from the power system to the emission of air pollutants, human exposure to air pollutants, and eventual health outcomes. This chapter begins with an introduction of the process through which power generation impacts air pollution and human health (Section 2.1) and discusses the way in which air pollution is linked to the problem of climate change. The chapter then discusses the severity of the problems of air pollution and climate change in the Rust Belt region in particular (Section 2.2). It further reviews policy approaches to air pollution and climate change, including Renewable Portfolio Standards (RPSs) (Section 2.3). Finally, this chapter provides a literature review on the costs and benefits of climate policy and RPSs (Section 2.4).

2.1 Air pollution impacts of power generation

This section introduces the way power generation influences air quality and human health. It first provides an overview of the Rust Belt power sector and next reviews the way in which power generation impacts human health by discussing the factors that determine power sector emissions, the path from emissions to concentrations, and the epidemiology of how pollutant concentrations affect human health. Finally, this section provides an overview of the relationship between the air pollution and climate change impacts of power generation.

2.1.1 Power sector in the Rust Belt

An important factor that determines the effect power generation may have on air pollution and human health is the mix of technologies used in the power system. In the Rust Belt region, electricity generation is currently skewed toward coal and natural gas burning power plants. In 2016, coal plants supplied 42% of the power generated in the region. Overall, fossil fuels provided 66% of the power. In comparison, the U.S. generated a similar portion of power from fossil fuel plants (65%) but used more natural gas (34%) than coal (30%) generation.

Renewable technologies, in contrast, supply a relatively small share of power generated in the Rust Belt region. In 2016, the renewable share equaled 5% compared to 15% in the U.S. Wind is the most common renewable power source in the Rust Belt, supplying 3% of the power generated.

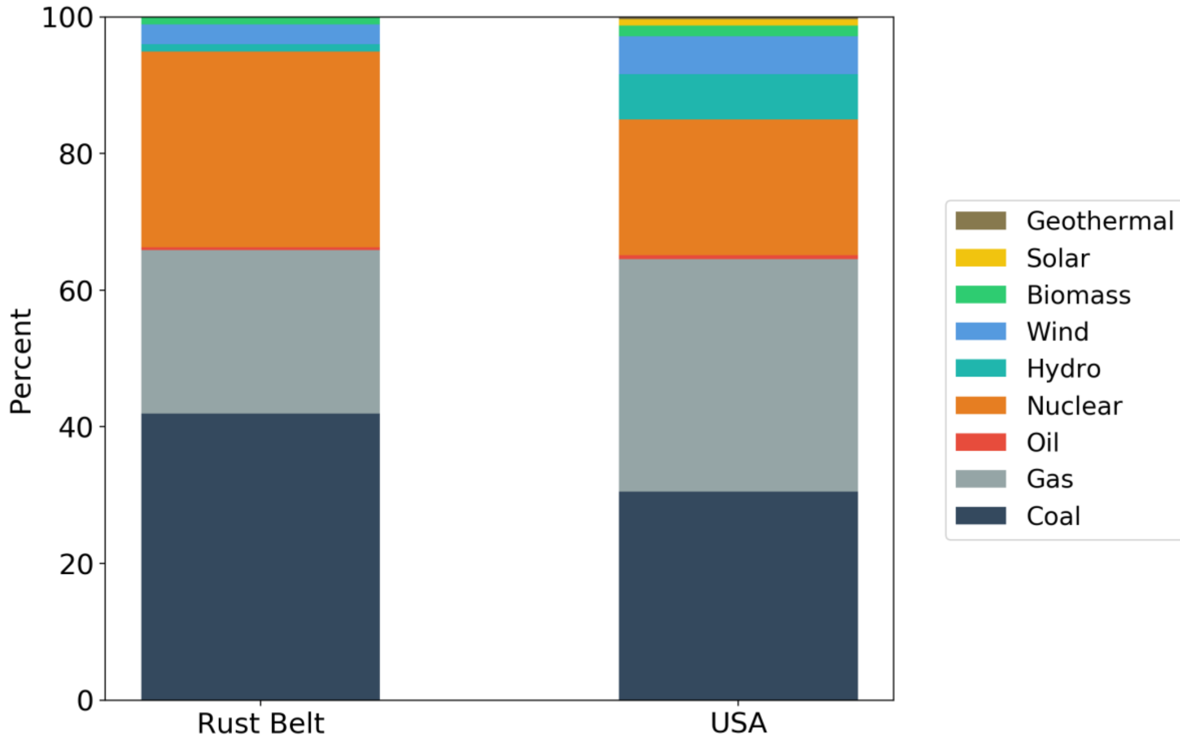

Figure 1: Power mix by region in 2016. Source: EIA 2017.

2.1.2 Factors determining power sector emissions

The combustion of fossil fuels leads to the formation and emission of a number of different air pollutants. When coal is burned to drive steam turbines, the sulfur contained in the fuel reacts with oxygen, resulting in SO_2 emissions. The combustion process also releases nitrogen oxides (NO_x), Particulate Matter (PM), referred to as primary PM, and heavy metals such as mercury. PM is a mixture of very small particles and liquid droplets, which is classified as either primary: released directly into the atmosphere; or secondary: formed by physical and chemical reactions from other pollutants (referred to as precursors). PM is further categorized by its aerodynamic particle diameter. The most commonly monitored types are PM_{10} (less than or equal to a diameter of 10 micrometers) and $PM_{2.5}$. $PM_{2.5}$ particles are the subject of particular concern and regulatory focus as discussed below due to their small size, their diameter being 20 times smaller than that of a human hair, allowing particles to enter deep into the lungs and some to enter the bloodstream (EPA, 2018f). The combustion of natural gas in gas turbines releases NO_x, PM, and negligible amounts of SO_2 and mercury. Oil burning can release significant amounts of SO_2 and NO_x.

In addition to fuel type, power plant emission rates depend on the heat content of the fuel, a plant's thermal efficiency, and the availability of pollution control technologies. Pollution controls in particular make a substantial difference. Existing technologies include Fluidized Bed Combustion, which removes up to 95% of sulfur from coal during combustion, and Flue Gas Desulfurization, which scrubs SO_2 post-combustion, with new units having a 95-98% removal rate. Low-NO_x burners and catalytic and non-catalytic converters and are also available to limit

NO_x formation during combustion or remove it post-combustion. As of 2015, 40% of coal power plants in the U.S. lacked any SO_2 pollution control technology, while 9% operated without NO_x controls (Massetti *et al.*, 2017).

When all factors underlying emission rates are accounted for, oil plants emerge as the highest SO_2 and NO_x emitting power plants in the U.S. per unit of energy. As of 2011, oil burning power plants released 2% more SO_2 per MWh relative to coal, and approximately three times as much NO_x (Massetti *et al.*, 2017). With coal in the middle, natural gas plants have the lowest SO_2 and NO_x emission rates, emitting almost no SO_2 and about six times less NO_x than coal.

The total amount of emissions depends on both the emission rates and the prevalence of different power generating technologies. Overall, coal-burning power plants were responsible for 98%, 86%, and 83% of SO_2, NO_x, and $PM_{2.5}$ emissions in the U.S. power sector respectively in 2011 (Massetti *et al.*, 2017). This is a result of coal's relatively high emission rates as well as coal's significant share of the U.S. power mix, which stood at 44% in 2011. Natural gas power plants, which supplied 24% of the power demand in 2011, emitted <1%, 8%, and 12% of these air pollutants respectively. The remainders were mostly emitted by oil plants, which supplied less than 1% of electricity in 2011.

As a result of this heterogeneity in emission intensity of different power plants, the air quality impacts of the power sector are sensitive to the composition of the power mix. The share of coal, and the way in which it may be influenced by different policies, is a particularly relevant factor. Economic modeling of the power sector can be performed with a variety of tools including statistical and optimization approaches. In this thesis, I use optimization modeling. The last section of this chapter discusses the use of different approaches in the relevant literature.

2.1.3 The path from emissions to pollution concentrations

Once pollutants are emitted, they may impact humans directly or indirectly. Carried by winds, compounds can come in direct contact with humans. For instance, SO_2 has been causally linked to respiratory problems, particularly for individuals with asthma (EPA, 2008). Pollutants may also react with other compounds in the atmosphere to form new chemical species, thereby impacting humans indirectly.

SO_2 and NO_x are particularly harmful as precursors to $PM_{2.5}$ (EPA, 2011). These compounds react with ammonia (NH_3) in the atmosphere to form inorganic secondary $PM_{2.5}$, which is a major portion of overall $PM_{2.5}$ concentrations that impact human health (EPA, 2009). Secondary organic $PM_{2.5}$ concentrations can also result from emissions of Volatile Organic Compounds (VOCs).

NO_x emissions can also be harmful as precursors to tropospheric ozone, which has been linked to respiratory diseases and premature death (Bell, Dominici and Samet, 2005; EPA, 2013a) NO_x compounds form tropospheric ozone by reacting with anthropogenic and biogenic VOCs. This thesis does not estimate policy effects on tropospheric ozone but instead focuses on $PM_{2.5}$, due to its greater impact on human health as discussed in the following section.

In addition to the emission of precursors, concentrations of secondary pollutants such as $PM_{2.5}$ depend on meteorology. Temperature and humidity have non-linear effects on $PM_{2.5}$ formation. Circulation patterns determine the long-range transport of compounds. $PM_{2.5}$ particles can travel hundreds of miles (EPA, 2018g) due to their small size. Precipitation impacts the deposition of particles, thus influencing atmospheric concentrations in a given location.

2.1.4 Epidemiology of health effects

The next link in the chain of emissions to impacts is the effect of pollutant concentrations on human exposure and health outcomes. Human exposure to $PM_{2.5}$ (for which researchers use $PM_{2.5}$ concentrations as a proxy) has been documented to have a number of adverse health effects. A large body of epidemiological literature concludes that there is an association between $PM_{2.5}$ concentrations and premature death (resulting from lung and cardiovascular diseases), which cannot be explained by chance or observable confounding factors (Dockery et al. 1993; Krewski et al. 2009; Lepeule et al. 2012). The presence of this epidemiological relationship between $PM_{2.5}$ exposure and mortality is also supported by human exposure studies (e.g Mills et al. 2007; Peretz et al. 2008) and experimental biology work (e.g. Bouthillier et al. 1998; Vincent et al. 2001). On the basis of the evidence, the EPA has concluded that $PM_{2.5}$ exposure is causally linked to premature mortality (EPA, 2009).

In the U.S., it has been estimated that $PM_{2.5}$ was responsible for just over 93,000 premature deaths in 2016, resulting in just over 1,556,000 years of life lost (YLL) (IHME 2017). This makes it the main source of premature deaths due to air pollution. The other main contributor to early deaths, tropospheric ozone, has been linked to just over 12,000 premature deaths and approximately 172,000 YLL in the U.S. in 2016 (IHME 2017). Globally, it has been estimated that $PM_{2.5}$ is responsible for more than 90% of mortalities related to air quality (Lim *et al.*, 2012; Lelieveld *et al.*, 2015).

The power sector is among the main contributors to $PM_{2.5}$ related premature mortality. Dedoussi and Barrett (2014) estimated that approximately a quarter of premature deaths in 2005 are attributable to electricity generation. The majority of the mortalities linked to the sector (75%) came from SO_2 emissions. The study found that the power sector was either the first or second largest contributor to early deaths, depending on the model used. Other major contributing sectors were the road transportation, industry and commercial/residential sectors.

2.1.5 Air pollution and climate change

The air pollution effects of the power sector are closely linked to the impact of power generation on climate change. Air pollutants and greenhouse gases (GHGs) are co-emitted in the process of fossil fuel combustion. The power sector in particular is a common contributor to both problems. As in the case of air pollution, the power sector is among the leading contributors to climate change in the U.S. In 2016, electricity generation was responsible for 28% of national GHG Emissions. It tied for first place with the transportation sector (EPA, 2018e).

Within the power sector, most GHG emissions resulted from coal burning, which released around 70% of GHGs in 2016. The remainder was emitted almost entirely by gas burning power plants, with oil contributing a negligible portion. An important determinant of this distribution is the relative emissions intensity of different fuels. Coal combustion emits almost twice the amount of CO_2 per unit of energy compared to gas (EPA, 2018e).

Air pollution is also linked to climate change due to the latter's effect on weather patterns. As meteorology can influence the formation of air pollutants, climate change may indirectly affect air pollution. Atmospheric modeling and statistical analyses have suggested that future climate change may worsen air quality in the U.S., leading to a so-called climate penalty (Wu *et al.*, 2008; Garcia-Menendez *et al.*, 2015; Shen, Mickley and Murray, 2017).

2.2 Rust Belt power sector impacts on air pollution and climate change

The impacts of power generation on air pollution and climate change are particularly acute in the Rust Belt region. Due to its greater use of coal, the Rust Belt region emits larger quantities of air pollutants than the U.S. per unit of power produced (Figure 2, EPA 2017). The difference in emissions intensity is greatest for primary $PM_{2.5}$ where it exceeds the U.S. average intensity by 72% and for SO_x (which includes SO_2 and SO_4) emissions where it is 63% greater.

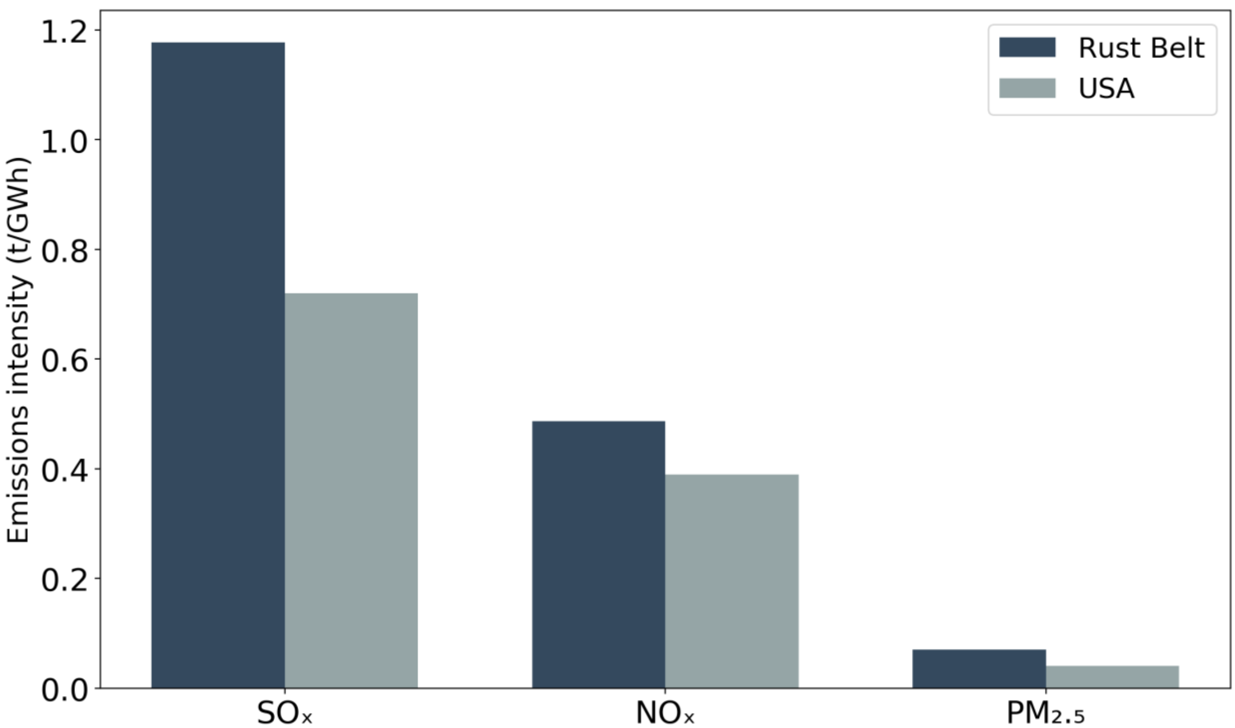

Figure 2: Air pollutant emissions intensity of the electricity sector, 2014. Sources: 2014 National Emissions Inventory (EPA, 2017a).

An additional factor contributing to the relatively high emissions of SO_x is the higher sulfur content of Appalachian coal used by eastern power plants (EIA, 1993, 2001). The relatively lower disparity in NO_x emission intensities can be partially explained by the greater share of gas generation in the U.S. power mix. Disparities in the prevalence of emission control technologies also play a role in determining differences in emission intensities (EIA, 2011).

The differences in these emissions intensities imply that air pollution impacts may be more severe in the Rust Belt region. Modeling work has confirmed this hypothesis. Caiazzo et al. (2013) estimated premature mortalities by state that are associated with $PM_{2.5}$ and ground level ozone exposure. They used atmospheric chemical modeling to simulate individual sector contributions to 2005 mortalities. As illustrated in Figure 3, the estimates for annual early deaths associated with the power sector are highest for Ohio (4,223) followed by Pennsylvania (3854), New York (3744) and Illinois (3161). Jaramillo and Muller (2016) also estimated air quality related premature mortalities by state, finding that Pennsylvania, Ohio, and Indiana bear the highest number of mortalities, a key driver being the amount of coal-fired power generation in these states.

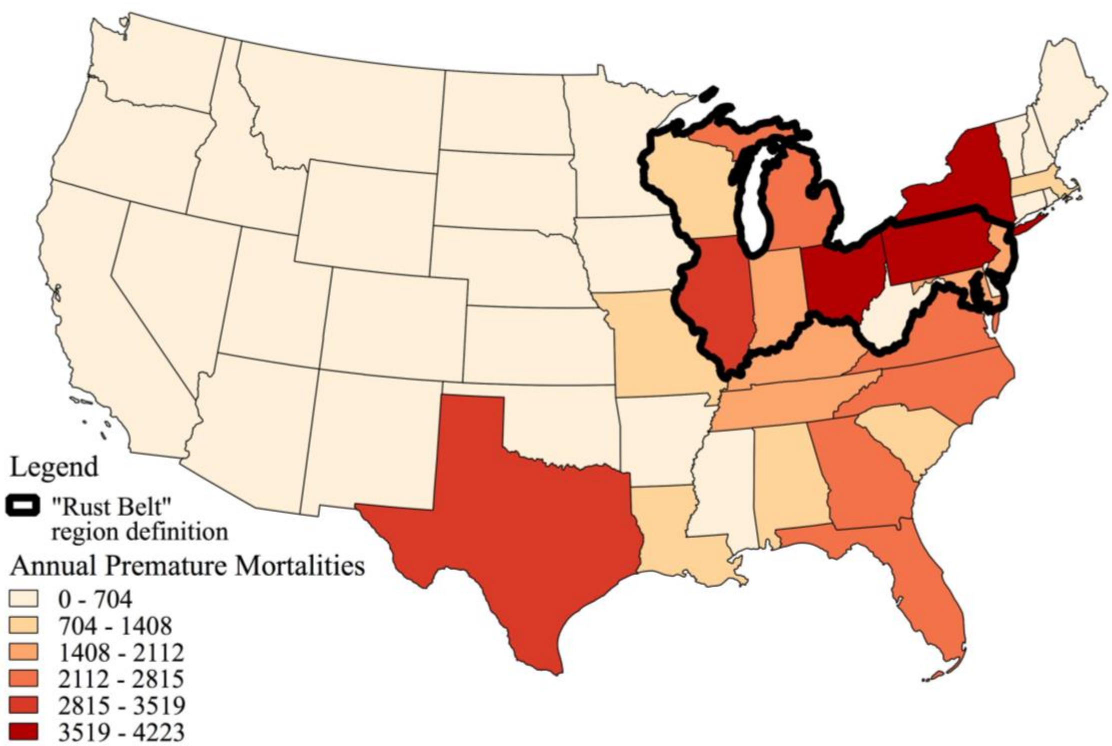

Figure 3: Annual premature mortalities attributed to power generation, estimated using 2005 emissions. Source: Caiazzo et al. (2013).

The Rust Belt region also stands out with regard to the CO_2 emissions intensity of its power generation. In 2015, power sector emissions in the region equaled 542 Mt CO_2 (EIA, 2018), while the region generated 1,005 TWh of electricity (EIA, 2017b), indicating an intensity of 0.54 Mt/TWh. In contrast, U.S.-wide power generation in 2015 had CO_2 emissions intensity of 0.46 Mt/TWh.

2.3 Policy approaches

2.3.1 Overview of air pollution policy in the U.S.

The U.S. addresses air pollution through the Clean Air Act (CAA). Pollutants such as SO_2, NO_2, $PM_{2.5}$, and ozone are classified as criteria pollutants. For these, the CAA grants the EPA authority to set National Ambient Air Quality Standards (NAAQS). NAAQS define a level of atmospheric concentration “requisite to protect public health with an adequate margin of safety” (EPA, 2018c). To ensure compliance with NAAQS, the EPA has implemented additional policies, such as the Cross-State Air Pollution Rule (CSAPR) cap-and-trade markets for SO_2 and NO_x emissions (another initiative under the CAA with impact on SO_2 emissions is the Acid Rain cap-and-trade program, implemented specifically to control acid deposition).

The current NAAQS for $PM_{2.5}$ as of 2012 specifies that concentrations cannot exceed an annual mean of 12 $\mu g/m^3$. Most areas in the U.S. meet this standard. In the Rust Belt, only four counties are in non-attainment of the $PM_{2.5}$ NAAQS, but these include relatively populous urban centers such as Cleveland, Ohio and Allegheny County, Pennsylvania (EPA, 2018d). In both states, the number of people who live in non-attainment areas exceeds 10% of the population (EPA, 2018d).

For areas in attainment of the NAAQS, air pollution continues to cause harm. This is in part due the limited authority of the EPA under the CAA as determined by case law. In *Lead Industries v. EPA*, the court ruled that the CAA does not require the EPA to set NAAQS at a zero-risk level (D.C. Circuit, 1980). Instead, the EPA promulgates standards that avoid “unacceptable risk of harm” (EPA, 2013b). Yet, the epidemiological literature has found no threshold for $PM_{2.5}$ concentrations, below which exposure ceases to be harmful (EPA, 2009). On the contrary, the marginal mortality risk has been found to be higher at exposures below the 12 $\mu g/m^3$ NAAQS (Di *et al.*, 2017). Therefore, adverse health effects caused by $PM_{2.5}$ can be expected to continue to occur despite the existence of the CAA.

2.2.2 Overview of climate policy in the U.S.

There is potential, however, for air pollution mitigation to occur as a co-benefit of climate policy. Because $PM_{2.5}$ precursors and CO_2 result from the same process of fossil fuel combustion, climate policies that reduce fossil fuel use will abate both types of emissions. Climate policy can also influence air quality by mitigating the meteorological impact of climate change on air pollution (i.e. the climate penalty). This thesis does not model this latter effect, which is estimated to be smaller than the emission-related effects, as discussed further in Chapter 3.

In the U.S., climate policy is currently largely contingent on state-level lawmaking. Following the 2016 election, the federal government is not expected to enact stringent climate legislation. This is most evident in the EPA’s proposal to roll back the Clean Power Plan rule, the landmark federal climate policy initiative of the Obama Administration (EPA, 2015a), replacing it with requirements for modest power plant upgrades (EPA, 2017b). The Administration’s proposed withdrawal from the Paris Agreement further signals a lack of intention to meet the U.S. Nationally Determined Contribution of reducing CO_2 emissions by 26-28% from 2005 to 2025.

Additionally, incentives specific to renewables, such as the Production Tax Credit for wind and the Investment Tax Credit for solar, are scheduled to expire in 2020 and 2022 respectively. In the absence of federal support for climate policy, investment decisions are becoming more dependent on state level policies. Even if new impetus for federal climate policy arises in the future, national policy may still provide substantial latitude to states to determine their own policy approaches, following the approach employed by the Clean Power Plan rule.

A number of states have committed themselves to reducing CO_2 emissions. A bipartisan group of 16 states has formed the U.S. Climate Alliance, pledging to reduce their CO_2 emissions by 26-28% from 2005 to 2025. Ten states have signed the Under2 Memorandum of Understanding, committing to abate CO_2 emissions by 80-95% below 1990 levels by 2050 or to below 2 tons of CO_2 per capita per year. The Governors of nine other states, including Ohio, Pennsylvania, and Illinois (members of the Rust Belt region as defined in this thesis) have made no public policy commitments but have expressed their support for the Paris Agreement (EESI, 2017).

There exists a wide range of climate policies. Economists generally distinguish between market-based and regulatory approaches. The former imposes a price on CO_2, through a tax or a market for pollution permits (referred to as cap-and-trade), and lets the market determine how to reduce emissions. Another policy that can be considered to belong in this category is a subsidy that rewards CO_2 mitigation equally for the amount of CO_2 abated. In contrast, the regulatory type of policies encompasses a broad range of instruments, through which lawmakers seek to intervene in the economy directly, by imposing requirements or favoring specific CO_2 abatement options through, for example, technology-specific subsidies.

While economists have argued in favor of carbon pricing as the least-cost climate policy (Pigou, 1932; Stern, 2006; Stiglitz *et al.* 2017), attempts to implement such instruments have consistently met resistance from lawmaking bodies and been abandoned or scaled back (Grubb, Hourcade and Neuhoff, 2014; Jenkins, 2014; Rabe, 2018). As a result, the scope of carbon pricing, both globally and in the U.S., remains limited (Zechter *et al.* 2017). In the jurisdictions where carbon prices do exist, most fall short of the EPA's estimates for the marginal cost of CO_2 emissions or the levels required to keep global warming below 2°C compared to preindustrial levels (Jenkins and Karplus, 2016; Stiglitz *et al.*, 2017).

Given the low political acceptability of carbon pricing, the theory of the second best (Lipsey and Lancaster, 1956) suggests that alternatives may be warranted. Some economists have argued in favor of renewable subsidies as a way to mitigate climate change (Wagner, 2015; Jaccard, Hein and Vass, 2016). Other researchers have proposed a sequential strategy to decarbonization policy (Meckling and Kelsey, 2015). This approach leverages regulatory instruments such as clean energy subsidies, which engender a new political constituency in support of climate mitigation that can eventually lead to the implementation of least-cost climate policy.

Renewable energy policy can be considered as a distinct subset of regulatory climate policy. Policy instruments that support renewable energy have often been implemented as parts of climate policy portfolios such as California's Global Warming Solutions Act of 2006 or been part of climate policy packages such as the most recent major attempt at federal climate policy –

the American Clean Energy and Security Act of 2009, which was passed in the House but lost support before reaching the Senate floor.

Yet, it must also be acknowledged that renewable energy policy may be pursued for purposes other than climate change mitigation. For example, lawmakers may use it as a vehicle to attract new industries to their jurisdictions. Case studies have suggested that economic development and job creation are common motivating factors (Stokes, 2015; Rabe, 2018). Rabe (2006) reported that, in many cases, climate mitigation is an ancillary rationale relative to the potential economic benefits. Economists have also argued that renewable energy policy may be justified as a solution to market failures other than the emission of GHGs. In particular, researchers have contended that renewable technology is subject to technology and adoption spill-over related market failures, whereby knowledge accumulated by the investment of early movers spills over to other market participants, thus discouraging investment and encouraging free-riding (Jaffe, Newell and Stavins, 2005).

Regardless of their ostensible rationale, renewable energy policies form an important component of U.S. climate policy due to their relative popularity (Leiserowitz *et al.*, 2018). Support for renewable energy among Americans has been on the rise. In 2018, 73% of 1,041 surveyed adults said they prefer the development of alternative energy sources to the production of more coal, oil, and gas, up from 66% in 2011 (Gallup, 2018). This preference for clean energy was found to be significant across the political spectrum, varying from 51% for Republicans to 88% for Democrats. Other surveys using a variety of question formulations have showed similar results, finding general support for the development of renewable energy among a majority of Americans (Hart Research Associates, 2016; Pew Research Center, 2016; Public Opinion Strategies, 2016).

2.3.2 Renewable Portfolio Standards

RPSs are among the most prevalent renewable energy policies in the U.S. (Carley and Miller, 2012). Such policies exist in 29 states and the District of Columbia. Eight other states have voluntary RPSs (National Conference of State Legislators, 2018). An RPS requires electricity providers (technically referred to as Load Serving Entities) to provide a certain percent of energy or power from renewable sources by a given year. There is a wide diversity of RPS designs. While most require that a certain percentage of energy generation be renewable, there are two exceptions – Iowa and Texas – where the requirement is for a given amount of power capacity (expressed in MW).

It is common for RPS statutes to provide flexibility as to how renewable requirements are met. LSEs can choose to meet the requirement by installing and running their own renewable plants, purchasing Renewable Energy Credits (RECs) generated by other renewable facilities, or pay an Alternative Compliance Payment, the proceeds from which are commonly invested in state clean energy programs.

A REC is a commodity that certifies the generation of one MWh of renewable energy. Power providers comply with an RPS by surrendering the amount of RECs needed to cover the percent of renewable energy required by the statute. Regulators track the generation and retirement of

RECs through digital accounting systems. Most RPS laws allow RECs generated within the same regional grid (Levin, 2017).

RPS statutes specify the types of renewable energy technologies that qualify for REC generation. Many divide the overall renewable requirement into tiers that represent different types of renewable technologies or construction dates (to distinguish between existing and new capacity). Lawmakers often single out specific technology types, such as rooftop solar panels, by creating carve-outs: portions of the renewable requirement that can only be met with the specified renewable resource. For instance, Ohio requires 6.5% renewable generation in 2020 and calls for 0.26% of generation (4% of the renewable requirement) come from solar power (N.C. Clean Energy Technology Center, 2018b).

Table 1 (next page) displays the range of RPS requirements by state. The percentages represent total renewable requirements including both broad requirements and carve-outs for specific technologies. As illustrated, RPS targets vary from Ohio's 7% in 2020 to Vermont's 59%.

Across these states, RPSs are the subject of frequent debates. State legislatures deliberated on a total of 181 RPS bills throughout the 2016 and early 2017 legislative sessions (Barbose, 2017). Some of these bills sought to raise the renewable percentage requirement, such as Pennsylvania's Senate Bill 291 from 2017. Others proposed to weaken or altogether repeal the requirement, such as Ohio's House Bill 114, which as of this writing is being deliberated in the state's Senate, having already passed through the House.

RPS discussions are ongoing in other Rust Belt states as well (Colorado State University, 2018). In Indiana, which currently has a voluntary RPS in place, a bill for a mandatory requirement, SB0318, was introduced in 2018. Minnesota's 2018 legislative session also included an RPS bill, which proposed an increase to the renewable requirement. In Wisconsin, two RPS bills featured in the 2017 legislature. In Michigan, the Public Service Commission, which regulates the state's electricity market, will consider plans by the two largest utilities, DTE Energy and Consumers Energy, to reach a 25% renewable energy target by the year 2030 (Samilton, 2018).

Most Rust Belt states are also nearing the final year of their RPS statutes (Barbose, 2017), which raises questions about whether states will legislate new targets, or whether renewable requirements will remain at the last value stipulated in current legislation. For Pennsylvania, the last RPS target year is 2020. In Minnesota, Illinois, Delaware, the final year is 2025 and in Ohio it is 2026. In Wisconsin, the RPS requirement has not been changed since it reached its final target in 2015.

Given that RPSs will likely continue to demand space on legislative agendas, it is pertinent to understand how alternative RPS decisions compare against one another on the basis of economic costs and benefits. Studying RPSs has relevance outside of the U.S. as well since such policies are commonly used in other countries and jurisdictions. As of 2015, at least 59 jurisdictions had implemented mandatory renewable energy targets (IRENA, 2015). Such policies exist at the national level in all European Union countries, China, India, South Korea, Japan, Australia Mexico, Chile, and elsewhere (IEA/IRENA, 2018).

State	2020	2025	2030	Comment
Arizona	10%	15%		
California	33%	42%*	50%	
Colorado	30%			
Connecticut	28%			
Delaware	20%	25%		
Hawaii	30%	35%*	40%	
Illinois	18%	25%		
Maine	40%			
Maryland	25%	25%*		
Massachusetts	15%	20%	25%	
Michigan	13%	15%		
Minnesota	22%	25%		
Missouri	10%	15%		
Montana				Montana's RPS specifies a renewable requirement of 15% by 2015
Nevada	22%	25%		
New Hampshire	21%	25%		
New Jersey	21%			
New Mexico	20%			
New York	29%	40%*	50%	
North Carolina	10%	13%		
Ohio	7%	12%	13%	
Oregon	20%	27%	35%	
Pennsylvania	8%			
Rhode Island	16%	24%	31%	
Texas		10GW by 2025 (approx. 10% of 2012 summer net capacity)		
Vermont	59%	66%	72%	
Washington	15%			
Wisconsin				Wisconsin's RPS specifies a renewable requirement of 10% by 2015

Table 1: Renewable requirements by RPS as of March 2018. Iowa's standard requiring 105 MW by 1999 has been excluded. States in bold are the focus of this thesis. * denotes interpolation by the author. Source: (N.C. Clean Energy Technology Center, 2018a).

2.4 Costs and benefits of climate policy

This section provides an overview of previous work on the costs and benefits of climate policy. It begins with a critical assessment of the cost-benefit framework. Following is a description of common approaches toward quantifying climate benefits. Next, I describe previous research on quantifying air quality co-benefits of climate policy and identify a gap in the literature concerning air quality co-benefits of sub-national RPSs. The section then reviews the quantification of policy costs and limitations in existing understanding of the total economic costs of RPSs, which this thesis seeks to address. Lastly, I discuss how the cost-benefit framework has been used to compare alternative policy approaches in previous literature and the way this thesis fills a knowledge gap regarding the way RPSs compare to carbon pricing.

2.4.1 The cost-benefit framework

The cost-benefit framework has both advocates and critics. Proponents have argued that it enables a transparent assessment of alternative decisions that forces an explicit consideration of the underlying assumptions (Viscusi, Vernon and Harrington, 2005). By expressing costs and benefits in the same monetary units, it provides a consistent way of expressing disparate information (Arrow *et al.*, 1996) and protects against cognitive biases (Sunstein, 2000). On the other hand, critics like Ackerman and Heinzerling (2002) countered that assumptions are rarely made clear outside of technocratic circles. They also asserted that the framework imposes a utilitarian perspective that does not appropriately account for issues of equity.

In the context of climate change, Grubb, Hourcade and Neuhoff (2014) argued that cost-benefit analyses are made impractical by the difficulty of quantifying costs and benefits and the presence of low-probability but potentially catastrophic outcomes, suggesting that a risk-management framework may be more applicable to climate change. Morgan *et al.* (2017) similarly advocated for a framework of identifying and avoiding climate thresholds as an alternative to cost-benefit analysis.

Notwithstanding these critiques, cost-benefit analyses remain a fixture in U.S. policy making. Federal agencies conduct such studies under the direction of executive order (Graham, 2007). State governments also perform or commission cost-benefit analyses (The Pew Charitable Trusts, 2013). Some have requested such evaluations specifically with regard to RPSs (Heeter *et al.*, 2014).

Case studies confirm that, while not sufficient, a cost-benefit analysis serves an important role in evidence-based policy making. It can be useful during particular stages of the policy making process such as assessments by appropriations committees where cost concerns must be addressed before policy proposals can move forward (Mosley and Gibson, 2017). In the environmental realm, the quantification of benefits along with costs can enable policy making based on net-benefit maximization as an alternative to financial affordability criteria (Graham, 2007).

A comprehensive review of climate policy costs and benefits is beyond the scope of this thesis. Climate policy confers a wide range of benefits in addition to climate change mitigation (The

Global Commission on the Economy and Climate, 2014). The focus of this work is on quantifying the air pollution related health co-benefits of climate policies, as well as their climate mitigation benefits and economic costs.

2.4.2 Quantifying climate benefits

A common approach to estimating the benefit of CO_2 emission abatement is the use of the Social Cost of Carbon (SCC). The SCC represents the discounted future monetary impacts of an incremental ton of CO_2 on the global economic system including effects on agriculture, human health, physical property, and ecosystem services. In the U.S., the Interagency Working Group on Social Cost of Greenhouse Gases (IWG) has recommended SCC values based on results from three Integrated Assessment Models (IAMs) - DICE, FUND, and PAGE - which represent the interactions between the economic system and climate processes. For instance, the central value for the SCC in 2020 is $42/t$CO_2$, expressed in 2007 dollars (IWG, 2016). The National Academy of Sciences has endorsed this approach but recommended a number of improvements such as the development of a new single IAM, more advanced treatment of feedbacks and linkages, and expert elicitation regarding key uncertainties, among others (NAS, 2017).

Some researchers have urged caution with regard to the SCC. Pindyck, (2013) asserted that "IAMs are of little or no value in evaluating alternative climate change policies and estimating SCC". The criticism centers on the fact that there is a large amount of uncertainty involved at different stages of the modeling process including emission projections, climate response, and economic impact. (Morgan *et al.*, 2017) contended that analyses of marginal impacts cannot capture the risk of catastrophic climate impacts. An estimate for the costs of climate change ceases to be useful when there is even a low probability of a high-impact outcome that would result in costs that cannot be compensated for financially.

However, SCC is commonly used for regulatory analyses in the U.S., such as the EPA's analysis of the Clean Power Plan (EPA, 2015b). As a means of capturing the marginal impact of low-probability high-impact outcomes, the IWG has recommended the use of an additional "High Impact" SCC value, which represents the 95th percentile of the SCC probability distribution computed by IAM models and is equal to $123/t$CO_2$ in 2020 (IWG, 2016).

2.4.3 Quantifying air qualify co-benefits

Assigning a monetary value to the air quality co-benefit of climate policy remains a critical challenge for researchers and policy makers. The previous literature has taken a variety of approaches and estimated values that span three orders of magnitude, with one global meta-analysis reporting a range of $2/t$CO_2$ to $196/t$CO_2$ (Nemet, Holloway and Meier, 2010).

A common approach is to translate premature mortalities into monetary terms using Willingness-To-Pay methods (Schelling, 1968). This method measures either demonstrated preferences (through contingent valuation surveys) or revealed preferences (hedonic wage studies using on-the-job risk exposure and wage data) of people's willingness to pay or accept a compensation for changes in the probability of death. Using a range of previous estimates derived from both categories of methods, the EPA estimated a central Value of Statistical Life of $7.4 million in

2006 dollars (EPA, 2014). Some researchers have argued against the notion that the value of life can be quantified (Ackerman and Heinzerling, 2002). Alternative approaches to monetizing health effects include the modeling of impacts on labor and demand for health services (Saari *et al.*, 2015) and calculating avoided cost of compliance with mandatory air pollution regulations (Bye, Kverndokk and Rosendahl, 2002). However, regulatory impact analyses often employ the VSL approach as a means of representing the value people may place on avoided mortality risks.

Modeling studies using VSL-based monetization methods suggest that health co-benefits can be on the same order of magnitude as climate benefits (as expressed by the SCC). Thompson et al. (2014) estimated co-benefits related to both $PM_{2.5}$ and ozone effects of an economy-wide carbon price in the U.S., arriving at a central estimate of \$170/tCO_2 in 2030. West *et al.*, (2013) tested the impacts of the RCP4.5 climate scenario (a global carbon pricing policy, which achieves 2100 radiative forcing of 4.5 W/m^2) on the $PM_{2.5}$ and ozone related health effects, as well as, on the health effects resulting from the mitigation of the climate penalty. The researchers found a worldwide average co-benefit of \$50-380/tCO_2, and \$30-600/tCO_2 for the US and Western Europe.

Previous studies have also estimated co-benefits of RPS policies at the national level. Wiser *et al.*, (2016) provided a retrospective comparison of the aggregate 2013 costs and benefits of all U.S. RPSs against a counterfactual scenario with no RPSs in the U.S. The authors estimated a central co-benefit of \$88/tCO_2 or ¢5 for each kWh of renewable generation triggered by RPSs. Mai *et al.*, (2016) modeled aggregate U.S. RPS costs and benefits up to 2050, estimating health co-benefits of \$11-39/tCO_2 or 1.2-4.2¢/kWh.

Despite previous interest in RPS co-benefits, the effects of specific state RPSs are not well understood as evaluations of sub-national policy impacts are rare. A survey of state-level analyses in the U.S. included only five studies on RPS effects on air pollutant emissions, only one of which, prepared by a Delaware utility, evaluates the resulting human health impacts (Heeter *et al.*, 2014). Modeling of RPS health effects has also been conducted for California and Colorado (Rouhani *et al.*, 2016; Hannum *et al.*, 2017).

A sub-national modeling approach is necessary to understand local impacts, as health effects can differ dramatically by location. Co-benefit estimates by Saari et al. (2015) for a carbon price in the U.S. exhibit a standard deviation across 11 U.S. regions of \$3/tCO_2 around a mean of \$6/tCO_2. Siler-evans *et al.*, (2013) further demonstrated the regional variation in the health benefits of renewable generation. The researchers estimated that health, environmental (such as visibility and agricultural losses), and climate damages avoided by a wind turbine or solar panel installed in a given location in the U.S. are in the range of 0.3-10¢/kWh. A significant factor influencing this variation is the type of fossil fuel generation displaced by renewable generation. The authors contrast a wind turbine in California, where the displaced fossil fuel is natural gas, estimated to provide 0.3¢/kWh in health benefits, and an identical turbine in Indiana providing a benefit of 8.3¢/kWh due to the displacement of coal-based generation.

2.4.4 Quantifying costs

The economic costs of RPSs are only partially understood, as the RPS literature has most commonly estimated policy costs using electricity system models or other approaches focused on the electricity sector (Wiser *et al.*, 2016, Mai *et al.*, 2016, Rouhani *et al.*, 2016). While sector-specific models allow for a detailed bottom-up representation of the sector of interest, they preclude the calculation of the total economic costs of a policy. For instance, they do not capture the spillover effects caused by changes in the electricity price on other sectors of the economy.

As an alternative, researchers often use economy-wide Computable General Equilibrium (CGE) models. Such models have been widely used in climate policy co-benefit studies (Thompson *et al.*, 2014, 2016; Saari *et al.*, 2015) but, to the author's knowledge, the only RPS study to use a CGE approach in the peer-reviewed literature is by Hannum *et al.* (2017) who model the impacts of Colorado's RPS.

The EPA has suggested that CGE models may be preferable when a policy can be expected to impact a wide number of sectors (EPA, 2014). Previous literature has argued that accounting for general equilibrium interactions is particularly important when analyzing the impacts of climate and energy policy (Bhattacharyya, 1996; Wing, 2009). Such approaches have showed that sector-specific model estimates substantially underestimate the social costs of air pollution abatement (Hazilla and Kopp, 1990; Goulder, Parry and Burtraw, 1996). By employing a CGE model, this thesis builds on the RPS literature by assessing the economy-wide impacts of RPSs.

2.4.5 Comparing alternative policy approaches

The cost-benefit framework enables a comparison between alternative climate policies. The most common approach in the literature is on the basis of economic costs relative to climate benefits expressed as the cost of mitigating one ton of CO_2. By this measure, carbon pricing emerges as the most efficient climate policy (Stern, 2006; Stiglitz *et al.*, 2017), while other approaches are often referred to as "second-best". By equalizing marginal CO_2 costs across the economy, carbon prices ensure that CO_2 emissions are reduced where it is cheapest to do so. An important assumption in this analysis is that mitigation options are treated equally for each ton of CO_2 abated and compared purely on the basis of their economic cost. A recent study from this literature (Rausch and Mowers, 2014) compared carbon pricing to an RPS at the national level in the U.S. and showed that the latter results in larger welfare costs for the same amount of CO_2 abatement.

However, accounting for co-benefits may alter the conclusion of pure cost-per-ton comparisons. Boyce and Pastor (2013) found that different climate mitigation options have different air quality co-benefits because the ratio of air pollutants to CO_2 emissions varies considerably across CO_2 sources. They concluded that a policy that treats CO_2 emission reductions as equivalent, as does carbon pricing, is less than optimal.

For illustration, one could consider what would happen under a carbon price when co-benefits vary by mitigation option. A price on carbon imposes the same cost per ton of CO_2 on all sources, inducing companies to choose between abatement options purely based on relative economic costs. In a hypothetical case where upgrading a coal plant with CCS is cheaper than replacing the coal plant with renewable energy, carbon pricing would result in the implementation of the CCS option, even though the second alternative delivers greater air quality co-benefits for each ton of CO_2 abated. Indeed, if the difference between the implementation cost and the co-benefit (which can be considered the "net cost") is lower for the renewable energy option, then carbon pricing becomes less efficient than a policy that favors renewables over CCS. Conversely, carbon pricing may be more efficient if the mitigation options that prevail under it are the ones that also provide higher co-benefits.

Driscoll *et al.* (2015) provided a comparison between carbon pricing and a policy that combines demand-side energy efficiency improvements with an Emission Performance Standard. The carbon pricing scenario was found to deliver fewer projected avoided mortalities per ton of CO_2 abated than the latter scenario. In the former case, CO_2 abatement was partially provided by Carbon Capture and Storage, which allows fossil fuel power plants to continue to operate and emit air pollutants.

Thompson *et al*. (2014) presented a comparison between a carbon price and a clean energy standard (CES) in the power sector, with both scenarios reducing an equivalent amount of CO_2 emissions. Consistent with Driscoll *et al*. (2015), the second-best CES policy was found to confer a greater air quality benefit than carbon pricing ($302/t$CO_2$ compared to $170/t$CO_2$). However, it imposed a significantly higher cost ($255/t$CO_2$ compared to $17/t$CO_2$), resulting in a lower net benefit. Saari *et al*. (2015) found the same pattern for the costs and benefits of CES relative to carbon pricing, as did Thompson *et al*. (2016) for the case of regional policies in the Rust Belt region. These results suggest that carbon pricing is indeed more efficient (delivering greater net benefits) than a CES, but that the difference in efficiency decreases after air quality co-benefits are considered.

In contrast, Knittel and Sandler (2011) used a statistical analysis to show that higher-emission vehicles are more responsive to carbon pricing or gasoline taxation than cleaner vehicles. The authors concluded that these policies would deliver higher reductions in air pollutant emissions than fuel efficiency regulations, which reduce, rather than increase, marginal cost of an extra mile traveled. Therefore, their work suggested that a consideration of air quality co-benefits make fuel efficiency standards even less efficient in comparison to pricing policies.

It remains unclear, however, how RPSs compare to carbon pricing. This thesis builds on this literature by assessing the efficiency of RPSs relative to carbon pricing under a cost-benefit framework that accounts for health co-benefits.

Chapter 3

Integrated modeling framework

An examination of how emissions from electricity generation affect human health requires an integrated analysis of the chain of events that lead from the electricity system to eventual health outcomes. To follow this chain of impacts, I integrate the United States Regional Energy Policy (USREP) model with the Intervention Model for Air Pollution (InMAP). This section describes USREP (Section 3.1), the use of USREP to model Renewable Portfolio Standards (RPSs) (3.1.1), the estimation of resulting air pollutant emissions (3.1), and the application of InMAP to translate emissions to pollution concentrations (3.3) and premature mortalities (3.4). I then describe the estimation of the economic benefits of avoided premature mortalities (3.5), and the economic benefits of climate change mitigation (3.6). Finally, I detail the policy scenarios designed to explore the impacts of alternative policy options (3.7).

The modeling choices presented in this chapter represent the Base Case modeling assumptions chosen to estimate the impacts across policy scenarios. An uncertainty analysis that tests how results for each policy scenario change under alternative assumptions will be presented in Chapter 4 Section 4.2.

3.1 Computable General Equilibrium modeling with USREP

I use the Computable General Equilibrium (CGE) model USREP to estimate the economic effects of energy policy to 2030. These include changes in the power mix, fuel usage, and economic output, which help determine how policies affect air pollution, as well as simulate changes in overall economic consumption, which serve as a gauge for the cost of policy. This section introduces CGE modeling and discusses the rationale behind using CGE for modeling energy policy. It then describes USREP and the model updates implemented for this study.

The distinguishing characteristic of CGE models is their economy-wide scope. CGE modeling represents the consumption and production decisions by households and producers using standard economic utility and production functions respectively. These functions are parameterized using an Input-Output table, which represents the flows of all goods and services throughout the economy for a given baseline year, as well as elasticities of substitution between goods and services derived from econometric literature. The functions assume optimizing behavior in the form of welfare maximization and cost minimization by consumers and producers respectively. Supply and demand are matched across all markets through non-linear optimization until an economy-wide general equilibrium is reached.

An important feature is the representation of the circular flows in the economy based on the economic theory of general equilibrium formalized by Arrow & Debreu (1954). CGE modeling simulates the supply of products from producers to consumers (households) and, in turn, the supply of factors of production (labor and capital) from households back to producers. A flow of payments runs in reverse to this flow of goods and services. By representing this important economic feedback loop, CGE modeling captures not only the consumption and production decisions made throughout the economy but also the interactions between them.

By introducing changes to the underlying economic assumptions, researchers can test how the general equilibrium outcome changes as a result of new policy. A CGE model can thus estimate how a policy that affects one sector, such as electricity, would affect the rest of the economy, and in turn how these effects will feed back into the electricity sector.

This capability offers three main advantages relevant to the scope of this thesis. First, it allows the estimation of the economy-wide macroeconomic impacts, including costs, of energy policies, in contrast to sector-specific models such as electricity capacity expansion models. Second, CGE modeling makes it possible to assess how emissions of air pollutants from unregulated sectors respond to electricity-sector policies such as RPSs. Third, a CGE framework provides the capacity to model policies that span multiple sectors of the economy, such as carbon pricing, and compare these to sector-specific policies in a consistent manner using the same modeling framework. However, these advantages come with the disadvantage of representing individual economic sectors using top-down production functions, thereby forgoing certain important technological details (Tapia-Ahumada *et al.*, 2014). Later in this section, I discuss in more detail how electricity technologies are represented in USREP.

The model used in this thesis, USREP, implements a CGE approach, which was described by Yuan, *et al.* (2017). Here, I review some of the main features relevant to this work. USREP is recursive-dynamic CGE model, simulating the temporal evolution of the economy in sequential five-year periods based on the assumption that economic agents are myopic, making their decisions with information available in each five-year period. While, this thesis presents future projections of policy impacts, it must be underscored that dynamic CGE modeling does not provide forecasts of the future. Instead, it produces what-if scenarios of potential future states given a host of uncertain underlying assumptions.

The model aggregates the economy into 10 sectors listed in Table 2, including fuel production, electricity and energy-intensive industry, and transportation, thus allowing for the separate representation of major emission sources. The model contains 12 U.S. regions shown in Figure 4.

Most relevant for this thesis is the representation of the electricity sector. The present model version represents 12 electricity generating technologies listed in Table 2. As in Rausch and Karplus (2014), it differentiates between “legacy” technologies, prevailing at the time of the model’s baseline year (2006), as well as a number of new advanced technologies. The legacy technologies include coal, gas, oil, nuclear, and hydro plants, and the advanced technologies are combined cycle gas turbines, single-cycle gas turbine, wind, biomass, advanced nuclear, pulverized coal with CCS and NGCC with CCS. Additionally, as in Paltsev *et al.* (2005), the model includes two combined technologies representing wind with single-cycle gas turbine backup and wind with biomass backup. Following Yuan *et al.* (2017), all technologies are treated as perfect substitutes.

Sectors	Electricity technologies
Electricity generation	Legacy
Energy intensive industry	Coal
Other industry	Gas
Transportation	Oil
Services	Nuclear
Agriculture	Hydro
Coal production	Advanced
Gas production	Combined Cycle Gas Turbine
Refined oil production	Single-cycle Gas Turbine
Crude oil production	Wind
	Biomass
	Advanced Nuclear
	Pulverized Coal with Carbon Capture and Storage
	Combined Cycle Gas Turbine with Carbon Capture and Storage

Table 2: USREP sectors and power technologies

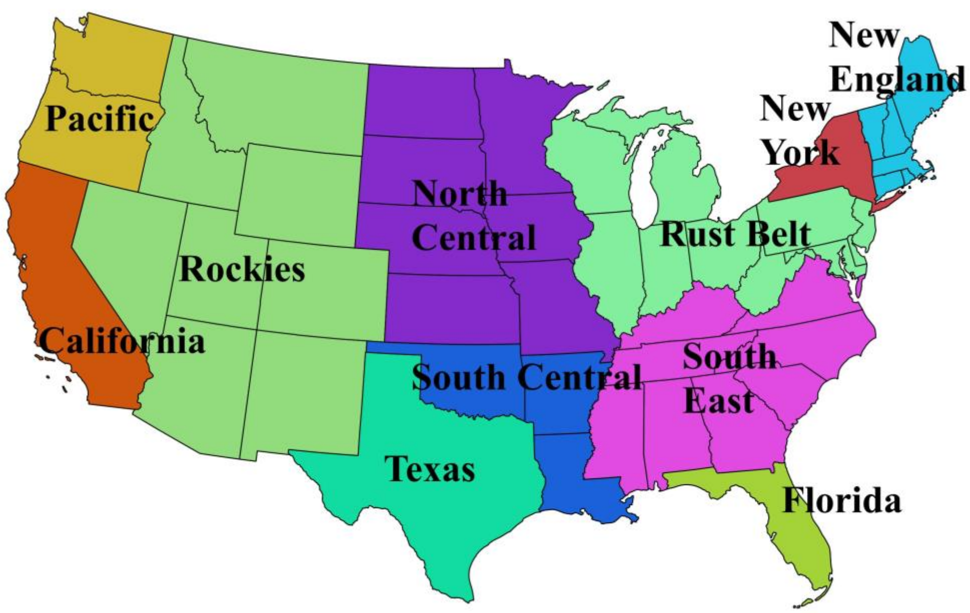

Figure 4: Regions represented in USREP

USREP models power generation in a top-down fashion, which is described by Rausch & Karplus (2014). Electricity production is modeled based on a nested Constant Elasticity of Supply production function. Paltsev *et al.* (2005) discussed the general approach. Constant Elasticity of Supply functions represent how, for each power generation technology, necessary inputs are used together to produce electricity. Once individual technologies are represented, the model chooses the cheapest mix of technologies based on total costs, reaching equilibrium between the total cost of generation and the price of electricity on an annual basis.

This top-down representation of the power sector introduces several modeling challenges. The annual representation of the electricity market does not permit a representation of intra-day operational characteristics of different technologies such as cycling costs and ramp rates, which determine how flexible a technology can be and can influence its ability to participate in intra-day market clearance. This limitation has implications for the impact of RPSs on power markets and emissions as discussed further in Chapter 4 Section 4.3. The annual resolution also creates challenges in representing the impact of the intermittency of renewable generation on the grid (Rausch and Mowers, 2014; Tapia-Ahumada et al., 2014). Finally, USREP does not represent intra-regional transmission constraints, which limits the amount of information that can be derived from the model regarding impacts at a particular location within a model region.

USREP represents the technological challenges of integrating new renewable energy into the grid by controlling the penetration of wind and other advanced technologies in two main ways. First, USREP constraints the penetration of each advanced technology once it has become cost-competitive to represent the gradual penetration that typically occurs in the case of new technologies as described by Paltsev *et al.* (2005). The approach involves the inclusion of an additional input “fixed factor” into the production function for each new technology. This fixed factor is represented as an initial endowment, which grows a function of output (for example, wind power generation) in the previous period. The penetration is determined by the initial endowment of this resource as well as the elasticity of substitution between it and the remaining wind generation inputs. For this thesis, I update USREP by parameterizing the growth of each technology’s fixed factor and its elasticity of substitution based on the latest available data derived by (Morris, Reilly and Chen, 2014). Second, wind penetration is controlled through wind supply curves described by Rausch and Karlpus (2014).

The penetration of new technologies is also determined in USREP by their competitiveness as measured by a mark-up parameter, which represents the cost of producing electricity with a given technology relative to the benchmark cost of electricity, which is assumed to be the cost of pulverized coal (Rausch and Karplus 2014). The cost of electricity represents the Levelized Cost of Energy (LCOE) and a transmission and distribution cost assumption that is universal across technologies.

For this thesis, I calculate LCOE’s and mark-ups for each advanced technology. I combine national recent technology cost data, including data on capacity factors and heat rates, from (Morris *et al.*, 2018) with fuel prices differentiated by USREP region from IMPLAN (2008), which are for the baseline year of the model, 2006, for consistency purposes.

An important assumption concerns the future of nuclear power capacity. While the model assumes that plants are available for their typical lifetime of 60 years, some plants may be retired prematurely due to factors not captured by the model. Average nuclear generation costs were higher in 2015 than they were in 2006, USREP's baseline year (Haratyk, 2017). This increase resulted from a rise in capital expenditures triggered by safety upgrades such as those following the Fukushima disaster. Higher fuel costs also played a role. In some cases, non-economic concerns also exert an influence on retirement decisions. For the Oyster Creek plant in New Jersey, safety concerns and associated regulatory compliance costs led to the decision to retire the facility (Haratyk, 2017).

It is thus necessary to input potential premature nuclear retirements exogenously. For this purpose, I conduct a review of potential nuclear retirements in the Rust Belt region. The resulting retirement assumptions reflect First Energy's announced retirement of Davis-Besse, and Perry plants in Ohio in 2020 and 2021 respectively, and Beaver Valley in Pennsylvania in 2021 (Walton, 2018), as well as Oyster Creek in New Jersey in 2018 (Exelon, 2018), Kewaunee in Wisconsin in 2013 (Wald, 2013), and Palisades in Michigan in 2022 (Balaskovitz, 2017).

3.1.1 Representing RPSs in USREP

I use USREP to represent an RPS as an additional constraint in the optimization structure, specifying that total renewable generation must equal a pre-determined percentage of the total electricity generated in the implementing region as described by (Morris, Reilly and Paltsev, 2010). Every unit of energy produced in the implementing region by any of the renewable energies in the model (wind, biomass or hydro) generates a Renewable Energy Credit (REC) that goes toward meet this constraint.

This is technically in contrast to RPS statutes in Rust Belt states where the renewable requirement is expressed as a percentage of energy sold (in other words, consumed) rather than generated. Utilities can meet these consumption requirements using RECs generated in another state, leading to a potential discrepancy between renewable power consumed and generated.

However, while RECs can be purchased from out-of-state, they are commonly sourced from generation within the same electricity market. States in the Rust Belt region have closely interconnected electricity systems. Most states in the region belong to the Pennsylvania-Jersey-Maryland (PJM) market. For some states, most of their territory lies within the Midcontinent Independent System Operator (MISO) market (Michigan, Wisconsin, Indiana and Illinois). Due to the interconnectedness of these power markets, this thesis assumes that a generation requirement is equivalent to a consumption requirement (equivalent to assuming that Rust Belt states can only use RECs from within the region).

This assumption is not unfounded as available data suggests that, in practice, the vast majority of RECs used by Rust Belt states originate within the region. In 2015, 92 percent of the main non-solar RPS requirement in Ohio was met with generation within this region (PUCO, 2016). Indiana contributed 21 percent, indicating that REC trading occurs between PJM and MISO territories. In Pennsylvania, the non-solar share of generation within the Rust Belt was 74 percent in 2015 (PA Public Utility Commission, 2016). For New Jersey, this figure was 97 percent in

2016 (NJ Board of Public Utilities, 2016). In Michigan more than 99 percent of the total RPS was met with generation within the region (Michigan Public Service Commission, 2018). In Maryland the same share equaled 91percent in 2015 (Public Service Commission of Maryland, 2017). Therefore, the representation of RPS as a generation requirement in USREP are approximately accurate.

3.2 Modeling emissions

The next step of the modeling framework is to estimate future emissions of air pollutants resulting from changes in the economic system derived from USREP. In this section, I describe the historical emissions data used and the steps taken to estimate 2030 emissions by scenario.

This thesis uses the most recent historical emissions data by source from the EPA's 2014 National Emissions Inventory (NEI) (EPA, 2017a). I use an aggregated version of this data (aggregated across time, space, and type of chemical species), prepared by (Tessum *et al.*, 2018). The dataset contains annual emissions of SO_2, NO_x, Volatile Organic Compounds (VOCs), NH_3, and primary $PM_{2.5}$ per source. The two types of sources featured are point sources, representing facilities, and area sources. They are classified according to EPA's Source Classification Code (EPA-SCC). The dataset is highly resolved with sources representing 6,506 unique EPA-SCC types. Overall, the dataset contains 14,924,260 emission sources in the U.S.

To estimate 2030 emissions, I scale historical emissions to 2030 using 2014-2030 projections estimated by USREP (as shown in the equation below). For this purpose, I match each emission source with a corresponding USREP output along two dimensions: source type and location. First, to pair source types, I map the EPA-SCC categories to one of 21 USREP variables using a mapping developed by Thompson *et al.* (2014). The authors matched 471 EPA-SCC codes to the following USREP variables: the fuel consumption of (coal, oil, and gas) in three sector categories (electricity, energy intensive industry, and other), the estimated value output (production multiplied by price) for 11 different sectors, and residential CO_2 emissions. I introduce a change to the mapping used in Thompson *et al.* (2014) by matching residential transportation related EPA-SCC codes to CO_2 emissions from private transportation calculated by USREP (rather than matching them to transportation output) to more accurately represent changes in this sector. I then match the 471 high-level EPA-SCCs in the Thompson *et al.* (2014) dataset to the 6,506 EPA-SCCs in the Tessum dataset by subsuming the latter EPA-SCCs into the higher-level categories, within which they fall. For example, two different anthracite coal-burning technologies are matched to their higher-level category that covers all anthracite coal plants. Second, to match location, I assign regional values to each source by overlaying the emissions data over a map of USREP regions using Geographic Information System (GIS) Python modules.

Next, I calculate emissions in 2030 by scaling 2014 emissions based on USREP outputs using the following formula:

$$E_{SCC,r,2030} = E_{SCC,r,2014} * \frac{V_{i,r,2030}}{V_{i,r,2014}}$$

Where: $E_{SCC,r,2030}$ denotes 2030 emissions originating from a source categorized by EPA-*SCC*, in USREP region r. $V_{i,r,2030}$ denotes value of the corresponding USREP variable i in region r.

3.3 Modeling concentrations

To estimate how emissions of air pollutants lead to concentrations of air pollution, I use InMAP. This section introduces InMAP, briefly compares it to other approaches, and specifies key modeling choices made in this thesis.

InMAP is a recently developed peer-reviewed Reduced Complexity Model (RCM). Tessum, Hill, and Marshall (2017) detail the modeling structure. Unlike other RCMs such as APEEP (Muller and Mendelsohn, 2006) and EASIUR (Heo, Adams and Gao, 2016), InMAP simulates the formation of secondary $PM_{2.5}$ and the long-range transport of pollution particles in a way similar to state-of-the-science Chemical Transport Models (CTMs). However, InMAP differs significantly from a CTM in its representation of atmospheric chemistry and physics. Rather than estimating all chemical properties (such as the oxidation rates for the transformation of SO_2 into SO_4 $PM_{2.5}$ particles) and meteorological conditions (such as wind vectors), it uses exogenous assumptions derived in advance from CTM simulations. Moreover, InMAP implements a linear representation of the chemical transformation of emissions into secondary $PM_{2.5}$. This is an important limitation, as these relationships are in reality non-linear. Thus, InMAP is suited for estimating marginal changes in concentrations, rather than total concentration values.

The advantage of InMAP relative to CTMs is the reduction of computational requirements. InMAP brings computation time from days to hours for a single simulation of the contiguous U.S. Thus, it allows for a greater number of policy scenarios to be run and for its more practical integration with economic models such as USREP.

Despite its simplified structure, InMAP results are comparable to outputs from the CTM WRF-CHEM. Tessum *et al.* (2017) show that InMAP recreates WRF-CHEM projections for changes in total $PM_{2.5}$ with $R^2 = 0.92$.

Relative to another commonly used RCM (EASIUR), InMAP provides the ability to estimate not only the impacts of emissions, but also where they occur. Therefore, InMAP offers the advantage of estimating impacts specific to a particular political jurisdiction. This capability is particularly relevant for the purposes of exploring impacts of sub-national policy on a jurisdiction's own air quality. Without accounting for long-range transport, it is possible to significantly overestimate how a jurisdiction is impacted by emission sources within its borders. For instance, Goodkind *et al.* (2017) estimated that around 30 percent of $PM_{2.5}$ mortality related damages occur more than 128 km from emission sources. The concentration of coal plants along the Ohio River, which acts as a border between Ohio and Kentucky, exemplifies the issue.

Another distinguishing feature of InMAP is its use of a variable-resolution grid where grid cell size varies throughout the domain. I specify 8 nests of varying grid sizes, with the largest grid equal to 288 km^2 and the smallest equal to 1 km^2. While InMAP offers the ability for grid sizes to be determined endogenously depending on population exposure, I run the model statically, keeping the grid specification constant across scenarios to enable comparability between outputs from different scenarios. In this case, grid cells are exogenously kept smaller in urban areas and larger in rural and remote areas (Tessum, Hill and Marshall, 2017).

An important assumption when projecting future concentrations concerns the modeling of changes in meteorology. In this thesis, the approach is to keep meteorology constant, following the methodology employed by Thompson *et al.* (2014). However, future change in meteorology triggered by climate change can influence the distribution and formation of $PM_{2.5}$, leading to a climate penalty as discussed in Chapter 2. The use of historical meteorological conditions in this thesis may lead to an underestimate of air quality co-benefits of climate mitigation. The magnitude is likely moderate as Garcia-Menendez *et al.* (2015) estimated climate penalty related co-benefits to be a fraction of emission abatement related co-benefits found in other studies using comparable VSL-based approaches (Thompson *et al.,* 2014, West *et al.*, 2013).

3.4 Modeling premature mortalities

This section discusses the approach and assumptions used to estimate premature mortalities resulting from pollution concentrations.

Premature mortalities are estimated within InMAP. I apply the standard Cox Proportional Hazards model as a Concentration Response Function (CRF). The number of deaths is a function of $PM_{2.5}$ concentrations, the number of people exposed, baseline mortality rates, and a concentration-response coefficient. I use CRF coefficients estimated by Krewski *et al.,* (2009) who found that for every $10\mu g/m^3$ increase in concentrations, premature deaths increase by 7.8 percent.

To make future projections, I scale population and baseline mortality rates to 2030. InMAP uses demographic-specific and spatially-resolved population and baseline mortality rates for 2013. I scale these using demographic-specific but U.S.-wide population projections (U.S. Census Bureau, 2012). It bears noting that these population growth assumptions are not necessarily consistent with population growth in USREP, which is computed endogenously.

I treat the lives lost due to changes in $PM_{2.5}$ concentrations as occurring in the same year as the change in the concentrations. This is likely to result in a small overestimate of 2030 impacts in comparison to accounting for delayed effects and discounting them back to 2030 (Barrett *et al.*, 2015).

Following the estimation of premature mortalities in InMAP, I downscale the spatial resolution of the results to the state level. I intersect InMAP's variable-grid of mortality estimates with a U.S. state layer. Where state boundaries cross InMAP grids, I divide the grid among states and apportion premature mortalities in proportion to area.

3.5 Estimating economic benefits of avoided mortalities

The monetary benefit of avoided premature mortalities is quantified here using the Value of Statistical Life (VSL). I use a range of 2006 VSL estimates published by the EPA from $0.85 to

$19.8 million (EPA, 2014). The EPA's central estimate of $7.4 million is used for the central (Base Case) estimates presented in this thesis.

For the projection of future benefits of avoided mortalities, I account for the income effect of higher future incomes on VSL estimates using the following equation.

$$VSL_{s,2030} = VSL_{s,2006} * \frac{GDP_{s,2030}}{GDP_{s,2006}}^{0.4}$$

Where: *s* is an index of policy scenarios. The scenario-specific GDP estimates are derived from USREP and represent GDP in the Rust Belt region. Finally, the formula uses a 0.4 income elasticity, based on the recommended central value in EPA's Benefits Mapping and Analysis Program-Community Edition model (RTI International, 2015). No discounting is applied to the overall benefit of avoided mortality because the value represents benefits incurred in 2030.

While health risks include both mortality and morbidity effects, the former constitute the vast majority of monetary health effect estimates (Burtraw *et al.*, 2003; Thompson *et al.*, 2014), and are thus the focus of this thesis.

3.6 Estimating economic benefits of climate change mitigation

USREP estimates economy-wide CO_2 emissions, which are used to estimate CO_2 reductions resulting from climate policies. I apply the Social Cost of Carbon (SCC) to estimate the monetary benefit of these CO_2 reductions. For this, I use the SCC recommended by the Interagency Working Group (IWG) on Social Cost of Greenhouse Gases (IWG, 2016) of $50/t$CO_2$ in 2030 (i.e. long-term climate mitigation benefits have been discounted to 2030), for consistency with mortality cost estimates.

This thesis leaves out considerations regarding non-CO_2 Greenhouse gases (GHGs). The carbon pricing policy scenario (described in more detail below) is assumed to only apply to CO_2 emissions, consistent with the approach taken by most existing carbon pricing policies. Considerations of non-CO_2 GHGs may, however, be an important component of cost benefit analyses. The higher potency and shorter atmospheric lifetime of gases such as methane may justify the use of "Social Cost of Methane" and other respective metrics (Marten and Newbold, 2012).

3.7 Policy scenarios

This section describes the policy scenarios that I design to test the costs and benefits of RPSs and carbon pricing in the Rust Belt region. Table 3 lists the policy scenarios, describes how they are specified in USREP, and distinguishes them from the modeling cases, which represent different modeling assumptions used to estimate results across all policy scenarios. The Base Case modeling assumptions are the ones presented in this chapter, while the following Chapter 4 will present results under the alternative modeling choices listed in Table 3.

<table>
<tr><th>Policy scenarios</th><th>Specifications</th><th>Modeling cases</th></tr>
<tr><td>No RPS</td><td>Renewable requirement in Rust Belt frozen at 2015 level of 6%</td><td rowspan="5">- Base Case assumptions (presented in Chapter 3)

- Alternative concentration-response assumptions (Chapter 4 Section 4.2.1)

- Alternative Value of Statistical Life assumptions (Chapter 4 Section 4.2.2)

- Alternative wind power costs (Chapter 4 Section 4.2.3)

- Alternative Social Cost of Carbon assumptions (Chapter 4 Section 4.2.4)</td></tr>
<tr><td>BAU</td><td>Renewable requirement in line with average of existing RPSs in the Rust Belt weighted by electricity sales reaching 13% in 2030</td></tr>
<tr><td>RPS +50%</td><td>Rust Belt renewable requirement reaching 20% in 2030</td></tr>
<tr><td>RPS +100%</td><td>Rust Belt renewable requirement reaching 26% in 2030</td></tr>
<tr><td>CO_2 price</td><td>Economy-wide cap-and-trade in the Rust Belt capping emissions sufficiently to achieve CO_2 emissions equivalent to RPS +100%. Renewable requirement in the Rust Belt equivalent to the BAU scenario.</td></tr>
</table>

Table 3: Policy scenarios and modeling cases tested in this study.

First, I define a Business as Usual (*BAU*) scenario to represent the current state of RPS statutes. To parameterize the model, I review the renewable requirements of existing RPS statutes up to 2030 using data from the Database of State Incentives for Renewables & Efficiency (N.C. Clean Energy Technology Center, 2018a). As described in Chapter 2, most statutes divide the overall renewable requirement into tiers and carve-outs that allow different technology types. For this study, it is important to account for carve-outs that specifically require solar or other distributed generation because these technologies are not represented in USREP. I reflect this by subtracting any such carve-out requirements from the overall renewable requirement. As a last step, I calculate weighted average RPSs by USREP region using state power sales in 2016 (EIA, 2017b).

To test the impact of existing RPSs, I include a counterfactual *No RPS* scenario. In this scenario all RPSs in the Rust Belt region are assumed to be repealed as of 2015. I represent this by freezing the renewable requirement in the model at the 2015 level. This is done to reflect the assumption that renewable plants built prior to 2015 will continue to operate, as their low marginal costs may encourage continued participation in the market, even if the RPS is repealed, which would not otherwise be reflected in USREP.

Next, I test two scenarios that represent a strengthening of Rust Belt RPSs, by 50% (*RPS +50%*) and 100% (*RPS +100%*). Figure 5 illustrates these RPS scenarios. The average RPS in the Rust

Belt weighted by 2016 electricity sales (EIA, 2017) in the *BAU* reaches 13% in 2030. The *RPS +50%* and *RPS +100%* scenarios reach approximately 20% and 26% respectively.

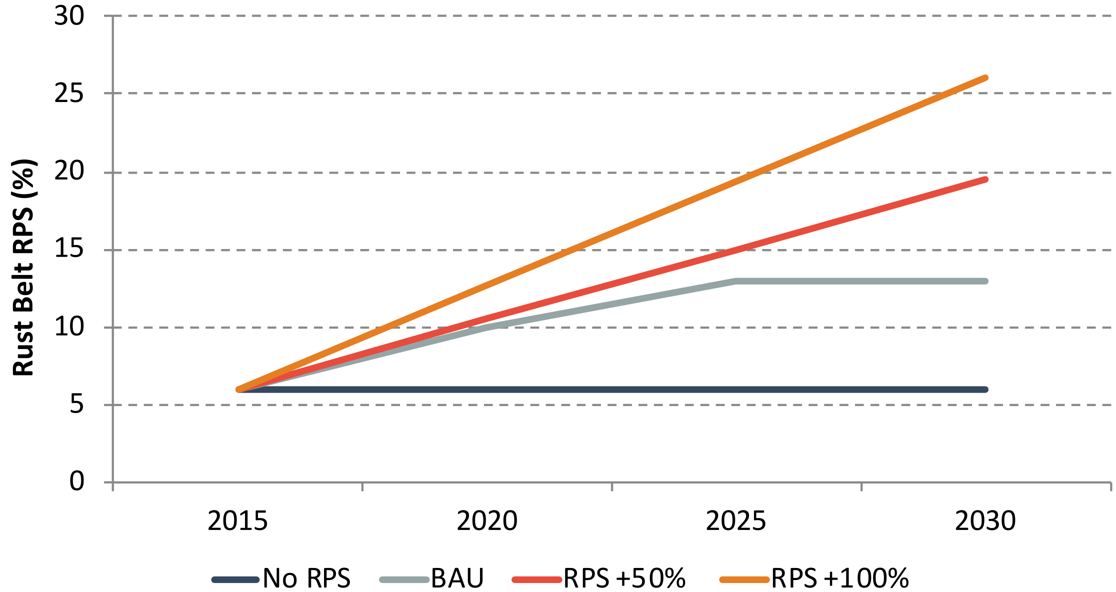

Figure 5: Renewable generation requirement by RPS policy scenario in the Rust Belt

An alternative policy option to raising the RPS may be the implementation of a carbon price in the Rust Belt. To explore the differences between these two options, I include a *CO_2 price* scenario that models an economy-wide cap-and-trade system in the Rust Belt region. This scenario introduces a constraint on CO_2 emissions and a market for CO_2 permit trading, allowing emissions to be reduced in the sectors where it is most cost-effective to do so. This scenario includes *BAU*-level RPSs in the Rust Belt region and adds a cap on CO_2 emissions beginning in 2020, which is stringent enough to result in the same amount of cumulative CO_2 as the *RPS +100%* scenario.

In addition to the policies being tested, it is pertinent to consider whether USREP represents other energy policies that currently exist in the Rust Belt. Across all policy scenarios, USREP does not feature other climate or air pollution policies (with the exception of the U.S. Corporate Average Fuel Economy standard). As a result, this thesis does not explore how the policies modeled here may interact with other policies.

However, a potentially important interaction can exist between climate policies and existing air pollution cap-and-trade systems (Groosman, Muller and O'Neill-Toy, 2011). Under the Clean Air Act (CAA), the EPA instituted the Acid Rain Program cap-and-trade for SO_2 and the Cross-State Air Pollution Rule (CSAPR) cap-and-trade markets for SO_2 and NO_x emissions as discussed in Chapter 2. The existence of these programs creates the possibility that climate policy would lead to a waterbed effect in air pollution emissions: reducing emissions in one place makes more permits available, allowing other sources to increase their emissions (just as a pushing a waterbed in one corner causes it to rise elsewhere) (Burtraw *et al.*, 2017). This waterbed effect can negate the effect of climate policy on air pollutant emissions already capped by existing CAA programs. This may lead this thesis to overestimate health co-benefits of climate policy. However, the potential for this overestimation is likely to be limited due to the surplus availability of permits in the CSAPR and Acid Rain programs. The pollution caps for SO_2 and NO_x under CSAPR exceed emissions by a factor of between two and five depending on the permit type (EPA, 2018a). In the Acid Rain program, the total amount of permits available (issued and banked by companies from previous years) exceed annual emissions by a factor of 28. These permit surpluses indicate that the pollution caps of these programs are non-binding. In cases of non-binding caps, the possibility of the waterbed effect is lowered (if a surplus of permits is already present, making more permits available is less likely to increase emissions) (Whitmore, 2016). Whether the permit surpluses in the CSAPR and Acid Rain programs persist in the future is uncertain. SO_2 permits auctions in 2018 cleared at only $0.06 (EPA, 2018b), suggesting that market participants may not expect significant shortages in the future. However, NO_x permits traded in 2016 at prices far above zero, pointing to potential expected scarcity (EPA, 2018a).

Chapter 4

Results and discussion

This chapter first presents Base Case results (Section 4.1) derived from using the Base Case modeling assumptions discussed earlier in Chapter 3 for the five policy scenarios (detailed earlier in Chapter 3 Section 3.7). The results begin with the power mix effects (Section 4.1.1), economic output effects (4.1.2) and CO_2 emission effects (4.1.3) of different policy scenarios. The section then presents air pollution results, following the chain of policy impacts from effects on emissions (4.1.4) to concentrations (4.1.5) and mortalities (4.1.6). The presentation of Base Case results concludes with a comparison of the costs and benefits of different policy scenarios (4.1.7).

In addition to the Base Case results, this chapter presents results for each policy scenario under alternative modeling assumptions (Section 4.2) regarding the Concentration-Response Function (CRF) and Value of Statistical Life (VSL) for measuring health co-benefits, the assumed wind power costs, which influence the economic impact of renewable energy policy, and the Social Cost of Carbon (SCC) for measuring climate mitigation benefits.

Finally, the chapter ends with a discussion of how the presented results compare to previous findings in the literature (Section 4.3).

4.1 Base case results

4.1.1 Power mix effects

A relevant place to begin exploring the impacts of Renewable Portfolio Standards (RPSs) is the power mix, as their effects on the shares of different power generating technologies help determine their eventual impacts on emissions, climate change, and human health.

Figure 6 illustrates the 2030 power mix in the Rust Belt region estimated by the United States Regional Energy Policy (USREP) model for each scenario, as well as the current power mix modeled by USREP for 2015. The current (2015) power mix estimated by USREP is reasonably consistent with the latest historical data for this region for 2016 (EIA, 2017a), with individual technologies differing by no more than 4 percentage points in their respective shares.

In the absence of climate policy (the *No RPS* scenario), the model projects a small decline in coal's share of the power mix from 37% in 2015 to 33% in 2030. The share of gas generation grows from 27% to 30%. Concurrently, the total amount of power generation in the Rust Belt region declines by 3% due to assumed energy efficiency improvements.

The addition of the existing RPSs (the *BAU* scenario) results in two main responses in the Rust Belt's power mix in 2030 relative to the *No RPS* scenario: a change in the relative power mix shares of different technologies and a decrease in total power generation. First, an increase in the renewable share from 6% to 13% in line with existing RPSs is accompanied by a decline in coal's share of the power mix in 2030 from 33% to 29%. The share of gas power also declines due to RPS policy from 30% to 26%. These changes are associated with decreases in generation of 46 and 48 TWh (-15% and -18%) respectively.

Among renewable technologies, wind is the main contributor to meeting existing RPS targets as represented by the *BAU* scenario. In the *BAU* scenario, the wind share is 11% compared to 5% in the *No RPS* scenario. The hydro share is the same across scenarios while biomass does not enter the power mix due to its relatively higher costs in USREP compared to wind. Solar does not feature in the results, as it is not represented in the model.

The second response occurring in USREP in the *BAU* scenario is a reduction in total power generation of 3.8% relative to the *No RPS* scenario. This is driven by a 3% increase in the 2030 price of electricity faced by consumers in the Rust Belt resulting from *BAU* RPSs. The higher electricity price triggers lower overall consumption of power (-3%) in the Rust Belt. The decrease in generation (production) caused by the RPS is greater than the decrease in consumption as the RPS lowers net export of electricity from the Rust Belt to other regions (by 12% in the *BAU* relative to *No RPS*).

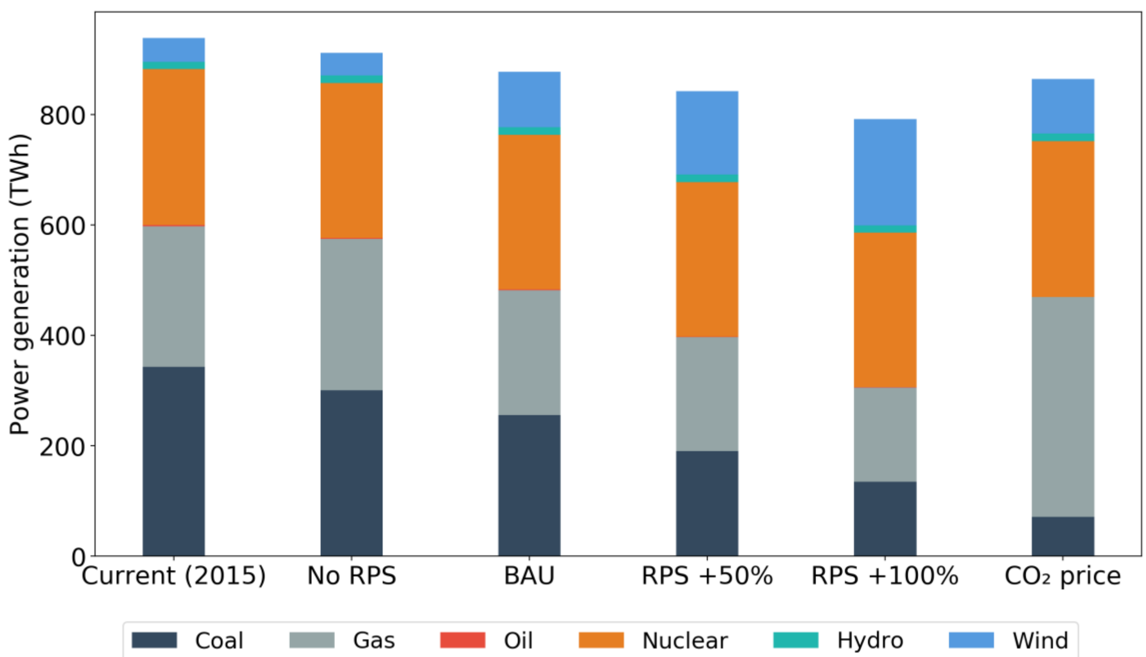

Figure 6: Power mix in 2030 by scenario and current (2015) power mix for the Rust Belt

The RPS effects on the power mix intensify as the stringency of the RPS renewable requirement increases. Raising the RPS requirement by 50% (corresponding to a renewable requirement of approximately 20%) leads to a further reduction in the share of coal share to 23% (from 29%

under *BAU*), a decline of 65 TWh in terms of generation. The share of gas sees a smaller reduction to 25% (from 26%), with generation down 20 TWh. Total power generation declines by 4.1% relative to *BAU*. The *RPS +100%* scenario (requiring 26% renewable energy) results in a coal share of 17% (-56 TWh), gas share of 22% (-35 TWh) and a 6% reduction in power generation relative to *RPS +50%*.

The *CO_2 price* scenario leads to qualitatively different changes in the power mix compared to the effects of the RPS. As the CO_2 price increases marginal cost for CO_2 emitting technologies based on their CO_2 emission intensity, it increases the competitiveness of gas relative to coal, leading to fuel switching. The renewable share does not rise above its value in the *BAU*, as the model finds it cheaper to reduce emissions by switching from coal to gas, as well as through reductions in other sectors (further discussed below).

4.1.2 Economic output effects

To further explore the differences in economic impacts of an RPS relative to carbon pricing, I compare the impacts of the *RPS +100%* scenario and the *CO_2 price* scenario relative to the *BAU* scenario on the economic output of each USREP sector (Figure 7). As illustrated, the predominant effect of carbon pricing is to decrease coal extraction output (-50%) while raising natural gas extraction output (+9%), consistent with the changes carbon pricing elicits in the power mix discussed above.

While the RPS also results in significant reductions in coal output (-40%), this scenario also affects other sectors of the economy such as electricity (-10%) and natural gas (-2%). Additionally, the RPS scenario has a small impact on a number of other sectors due to a larger impact on the electricity price (+10% from *BAU* compared to +2% under CO_2 pricing), which propagates throughout the economy. For example, the *RPS +100%* scenario lowers non-energy-intensive manufacturing ("other industry") output (-0.6%), agriculture output (-0.7%), and transportation output (-0.4%) while carbon pricing has a negligible impact (<0.1%) on these sectors. The RPS scenario also raises output in the oil sectors somewhat as it incentives greater usage of internal combustion engine vehicles relative to electric vehicles as discussed further below.

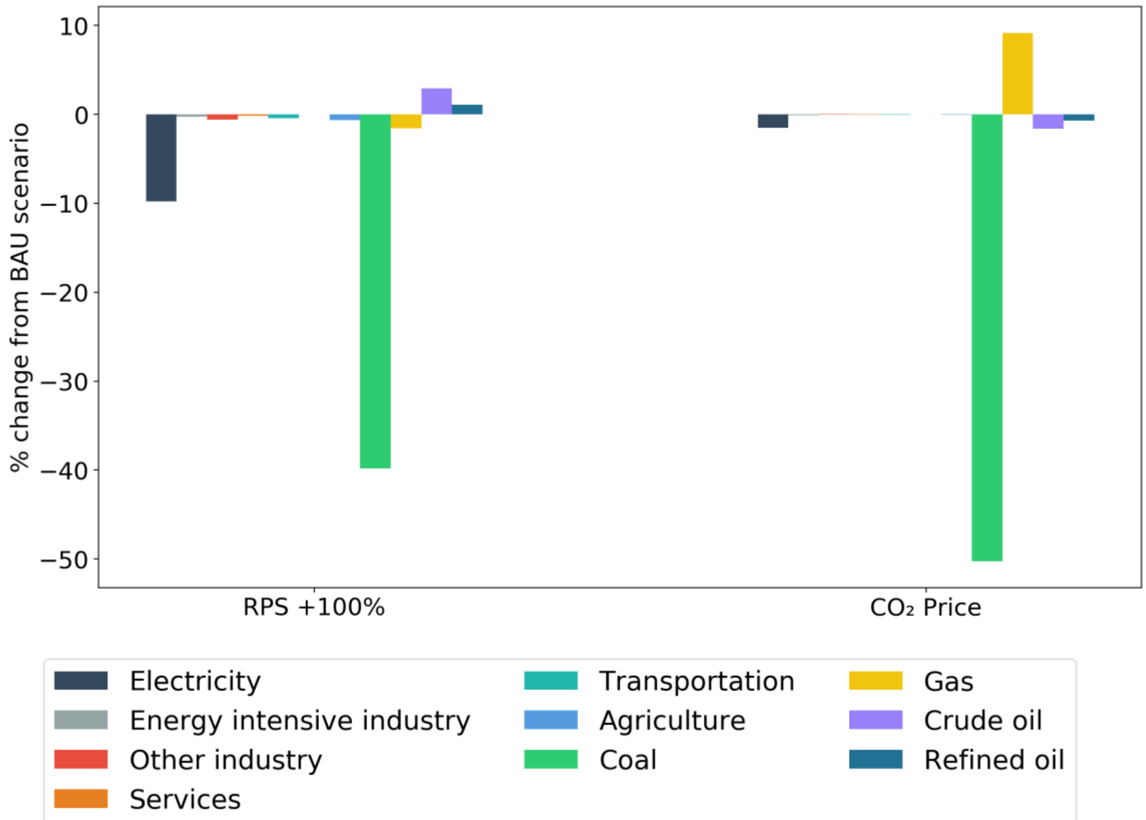

Figure 7: Changes in output by sector relative to *BAU* scenario for the Rust Belt

4.1.3 CO_2 emission effects

Figure 8 illustrates CO_2 emissions by sector estimated by USREP for each scenario and compares them to current (2015) emissions. Modest reductions occur between 2015 and 2030 in the absence of policy (*No RPS* scenario) mainly due to an estimated decline of coal generation and energy efficiency improvements assumed by USREP. The *BAU* scenario results in the abatement of 50 Mt CO_2 compared to the *No RPS* scenario. Most of the CO_2 reductions are driven by the decline in generation from coal-fired power plants. Abatement increases with RPS stringency. The two additional RPS scenarios reduce 112, and 168 Mt CO_2 respectively relative to *No RPS*.

By design, the *CO_2 price* achieves the same CO_2 reduction as the *RPS +100%* scenario. As the *CO_2 price* scenario already includes the *BAU*-level RPS, the cap-and-trade system implemented is specified to be stringent enough to deliver reductions of 118 Mt CO_2 in 2030 compared to the *BAU* (equivalent to the *RPS +100%* scenario). The CO_2 price generated by the model to achieve these reductions is relatively modest at $3.40/t$CO_2$ in 2030.

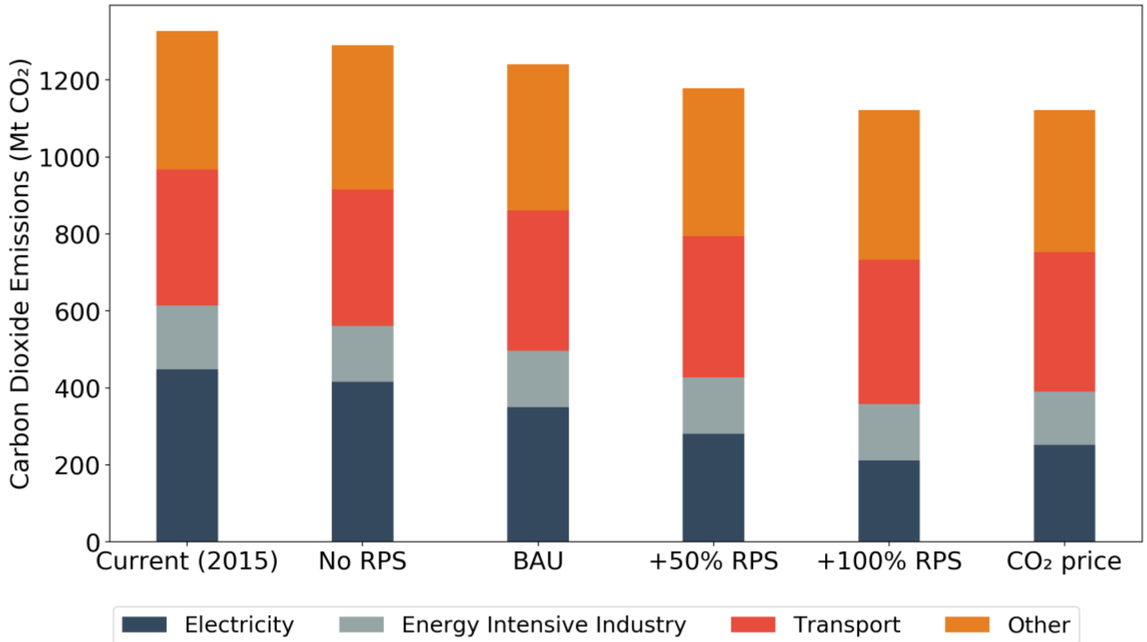

Figure 8: CO_2 emissions in 2030 by scenario and current (2015) emissions for the Rust Belt

A comparison across sectors shows that electricity is the largest source of Rust Belt emissions in 2030 in the *No RPS* scenario. This sector is also the main source of CO_2 abatement.

Relative changes in emissions by sector are highlighted further in Figure 9, which compares the effects of all policy scenarios on each sector relative to the *No RPS* scenario. As an electricity sector policy, the RPS results in a greater amount of CO_2 abatement in the electricity sector. However, an emissions leakage effect is present whereby emission abatement in the electricity sector is partially offset by emission increases in the “other” and transportation sectors. The emission increase in the “other” category occurs due to higher emissions from residential sources mainly as a result of higher natural gas and oil usage, as higher electricity prices triggered by the RPS induce households to switch from electric heating toward gas and oil burners.

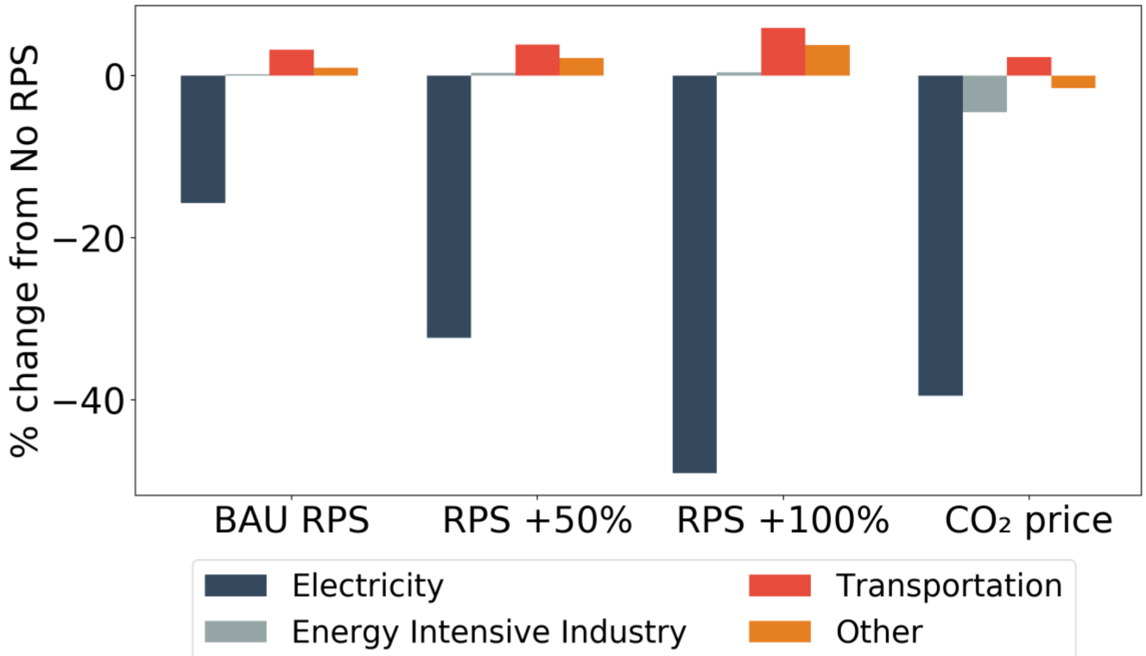

Figure 9: Changes in CO_2 emissions by sector and scenario in 2030 in the Rust Belt

Transportation emissions go up as higher electricity prices incentivize households to increase usage of internal combustion engine vehicles relative to electric vehicles. The share of vehicle miles traveled by electric vehicles falls from 9% in the *No RPS* scenario to 4% in the *BAU* scenario, while total vehicle miles traveled are virtually unchanged. Emissions form energy intensive industry remain relatively unchanged in the RPS scenarios as somewhat higher consumption of coal (equal to 0.7% in the *BAU* relative to the *No RPS*, resulting from lower coal demand from the electricity sector) is offset by lower consumption of oil (a change of 0.5%, triggered by somewhat higher oil prices resulting from increased residential demand), and somewhat lower sectoral output (-0.1%).

The cap-and-trade system implemented in the *CO_2 price* scenario partially offsets the emission leakage from electricity to other sectors due to its economy-wide scope. Emissions from transportation increase somewhat compared to *No RPS* but by less than in the *BAU* scenario. The increase in emissions from this sector occurs as the *CO_2 price* scenario includes a *BAU*-level RPS, which is the cause behind the rise in emissions. In the “other” sectors, the CO_2 price fully offsets the emission leakage discussed above. In this scenario, emissions from these sectors decrease due to lower coal consumption in non-energy-intensive manufacturing but reductions are somewhat offset by a higher output of natural gas extraction, which rises by 7% relative to *No RPS*.

Most reductions in *CO_2 price* scenario occur in the power system, as fuel switching from coal to gas in the power sector is a relatively inexpensive CO_2 abatement option relative to alternatives in other parts of the economy. Modest CO_2 abatement occurs in the energy intensive industry sector due to lower coal usage (-10% lower coal-related emissions in this sector compared to *No RPS*).

The policies modeled also trigger changes in CO_2 emissions outside of the Rust Belt region. As displayed in Figure 10, the RPS scenarios increase electricity emissions in other areas. This emission leakage results mainly from greater coal consumption for power generation. Transportation emissions go down outside of the Rust Belt in the *BAU* scenario relative to *No RPS* as vehicle miles traveled by electric vehicles rise 8%.

Overall, CO_2 emission changes in non-Rust-Belt regions equate to a small decrease in *BAU* of -2Mt CO_2 and increases of 7 and 13 Mt CO_2 in *RPS +50%* and *RPS +100%* relative to *No RPS*. The emission leakage effects in the *RPS +50%* and *RPS +100%* mean that 6% and 8% of the respective emission reductions in the Rust Belt are offset by increases in emissions elsewhere.

In contrast, the *CO_2 price* scenario results in less emission leakage outside of the Rust Belt compared to the equivalent *RPS +100%* scenario. Overall, an increase of 5 Mt CO_2 is present in non-Rust-Belt regions under CO_2 pricing (corresponding to 3% of Rust Belt emission reductions). Due to this leakage effect, on a national scale, the CO_2 scenario reduces 8 Mt CO_2 more than the *RPS +100%* scenario.

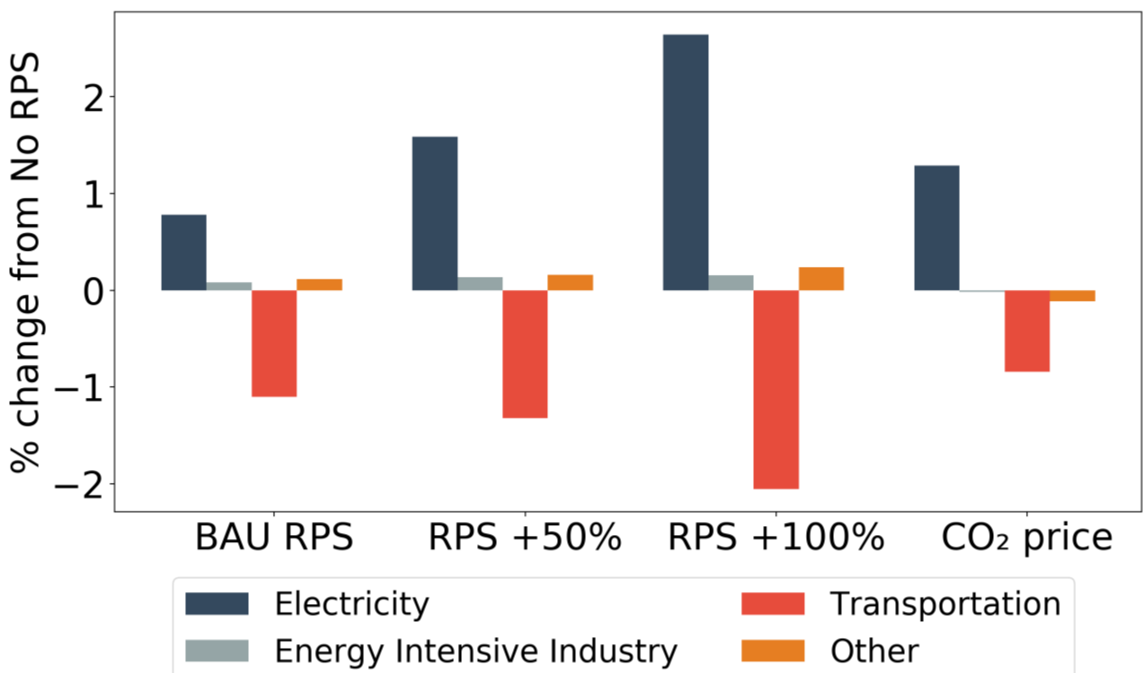

Figure 10: Changes in CO_2 emissions by sector and scenario in 2030 in areas other than the Rust Belt

4.1.4 Air pollutant emission effects

Emissions of primary air pollutants vary across pollutant types, economic sectors, and scenarios. Figure 11 displays 2030 emissions by scenario and sector for each pollutant category in the Intervention Model for Air Pollution (InMAP) and compares them to current (2015) emissions by sector. Among the selected sectors, the electricity production is the main source of SO_2 emissions, while transportation contributes the highest share to NO_x emissions. "Other" sectors are the predominant source of primary $PM_{2.5}$, NH_3 and Volatile Organic Compounds (VOCs), in large part due to land use activities and biogenic sources.

From 2015 to 2030, emissions of SO_2 decline by 14% in the *No RPS* scenario due to a decline in coal use in the power sector. Emissions of NO_x rise by 3% from 2015 to 2030 in the *No RPS* scenario as economic growth and an increase in gas consumption for electricity offset lower coal usage. Economic growth also causes emissions of the remaining pollutants to rise by 2030.

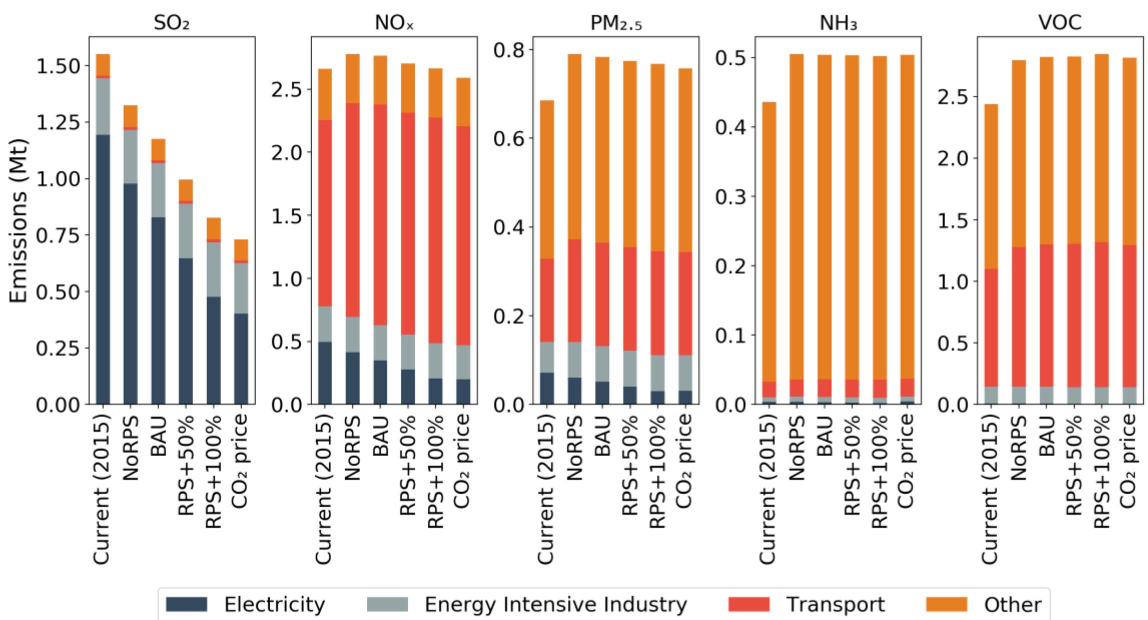

Figure 11: Air pollutant emissions in 2030 by pollutant type and scenario and current (2015) emissions for the Rust Belt region

The modeled policy scenarios cause a reduction in SO_2 and NO_x primarily. Changes in emissions by policy scenario relative to the *No RPS* scenario are displayed more clearly in Figure 12. SO_2 emissions decrease by 11% in the *BAU* scenario. This is mainly driven by the 15% decline in coal generation, due to coal's relatively high SO_2 emission intensity. RPS-driven reductions scale with RPS stringency for all pollutants.

The *CO_2 price* scenario results in greater reductions of SO_2, NO_x, primary $PM_{2.5}$ than the comparable *RPS +100%* scenario, consistent with the larger reduction of coal use for electricity generation.

Total emissions of NH_3 remain relatively unchanged across scenarios, while VOC emissions exhibit small increases. These changes are explained further below.

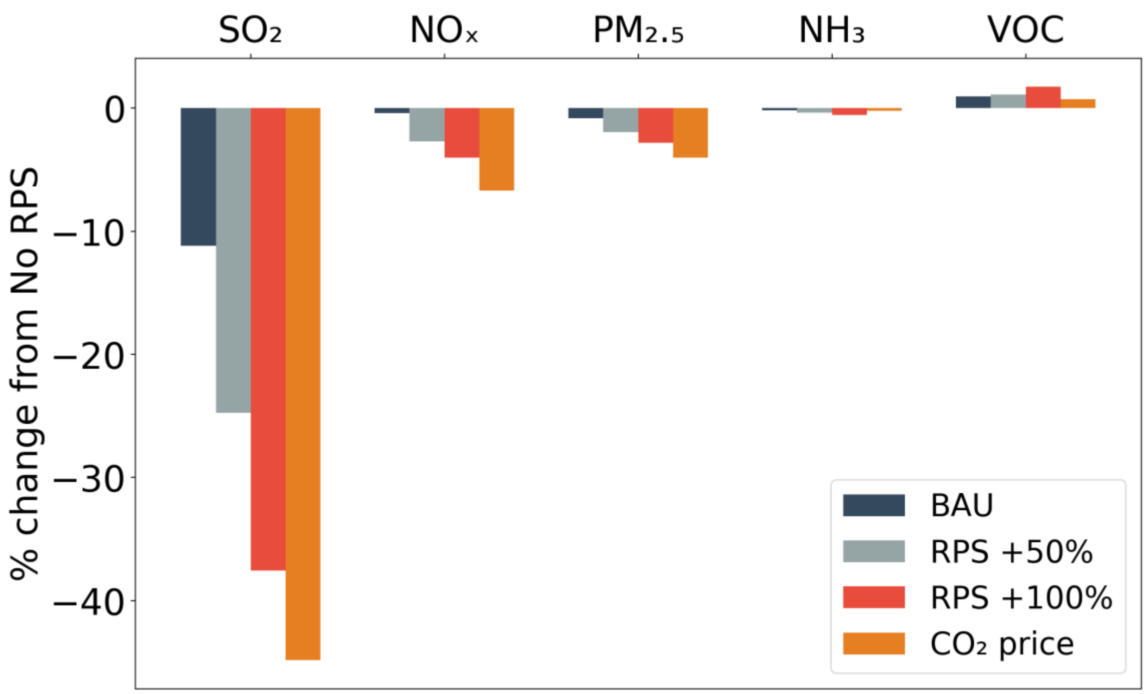

Figure 12: Changes in air pollutant emissions by scenario in 2030 for the Rust Belt region

Breaking down the emission changes by sector shows that the power sector is the main source of emission reductions across all RPS scenarios (Figures 13). The lower consumption of coal and gas in the power sector leads to reductions of all air pollutant types in this sector. The energy intensive industry sector emissions show virtually no change across RPS scenarios due to the countervailing forces of somewhat lower coal consumption and somewhat higher oil consumption discussed previously. Emissions from transportation rise in the RPS scenarios as electric vehicles contribute a lower portion to total vehicle miles traveled. Other sectors exhibit little change in emissions as higher residential emissions (resulting from the greater usage of gas and oil discussed above) are offset by lower emissions in other sectors resulting from lower economic activity caused by higher electricity prices.

Relative to an RPS (the *RPS +100%* scenario), CO_2 pricing results in higher electricity sector abatement of SO_2 (Figure 13) due to lower coal use. However, it also leads to smaller reductions of primary $PM_{2.5}$, NH_3, and VOC from power stations due to greater use of gas. However, this effect is offset by the fact that carbon pricing decreases energy intensive industry emissions of SO_2 and NO_x due to the previously mentioned lower coal consumption in the sector.

In transportation, emissions of air pollutants increase somewhat under the *CO_2 price* but less than under the *BAU* scenario, in line with the changes observed in CO_2 emissions discussed previously. The reductions of primary $PM_{2.5}$, NH_3 and VOCs in energy intensive industry and transportation under carbon pricing outweigh the smaller reductions of these pollutants in the electricity sector, leading to the larger overall reductions in these pollutants under carbon pricing relative to the *RPS +100%* displayed earlier in Figure 12.

CO_2 pricing also abates “other” sector SO_2 emissions by inducing lower consumption of coal in non-energy-intensive manufacturing. NO_x and primary $PM_{2.5}$ emissions in the “other” sectors remain relatively unchanged as lower coal usage is offset by higher emissions from natural gas extraction.

Figure 13 also helps explain the changes in total NH_3 and VOC emissions illustrated earlier in Figure 12. Total NH_3 emissions remain relatively unchanged as the chosen policies have a relatively low effect on sectors in the “other” category where most emissions originate. The rising VOC emissions under RPS scenarios result from higher transportation sector emissions.

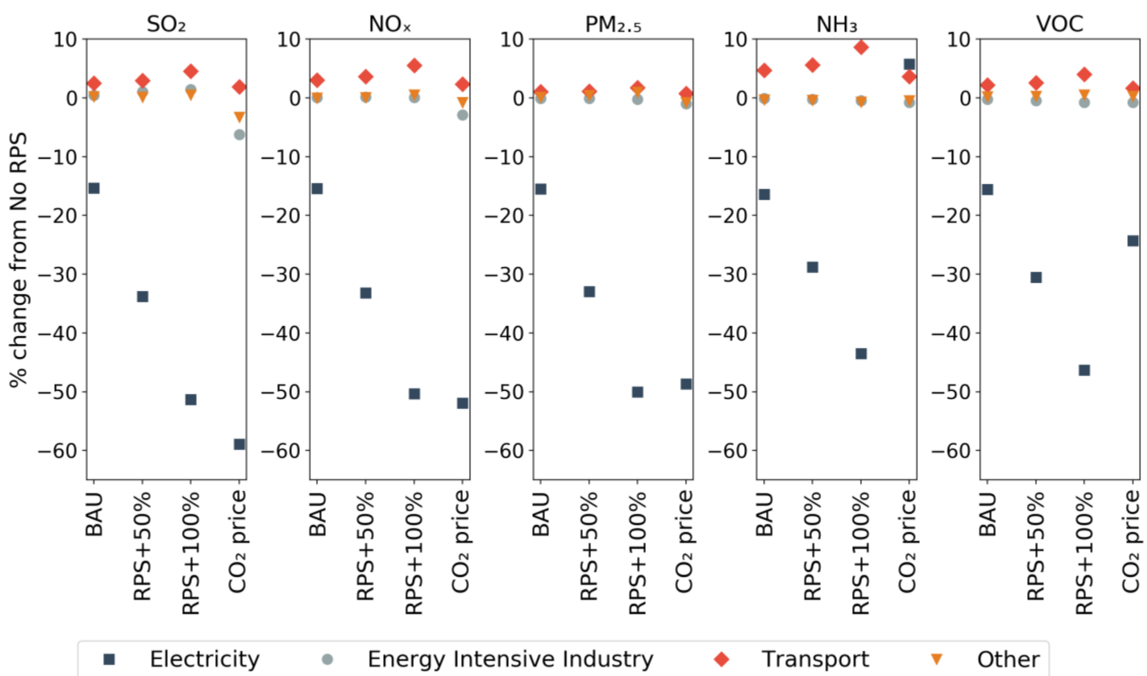

Figure 13: Changes in air pollutant emissions by sector, scenario, and pollutant type in 2030 for the Rust Belt region

In areas outside of the Rust Belt, air pollutant emissions also change as a result of policies in the Rust Belt. Figure 14 shows non-Rust Belt changes by air pollutant in each scenario compared to *No RPS*. SO_2 emissions change the most in all scenarios, by up to approximately 2%, resulting from the decrease in electricity exports from the Rust Belt and the greater coal consumption outside of the Rust Belt resulting from the decrease in coal prices caused by climate policy in the Rust Belt. In RPS scenarios, NO_X and VOC emissions decline somewhat, which can be largely attributed to a higher number of vehicle miles traveled by electric vehicles.

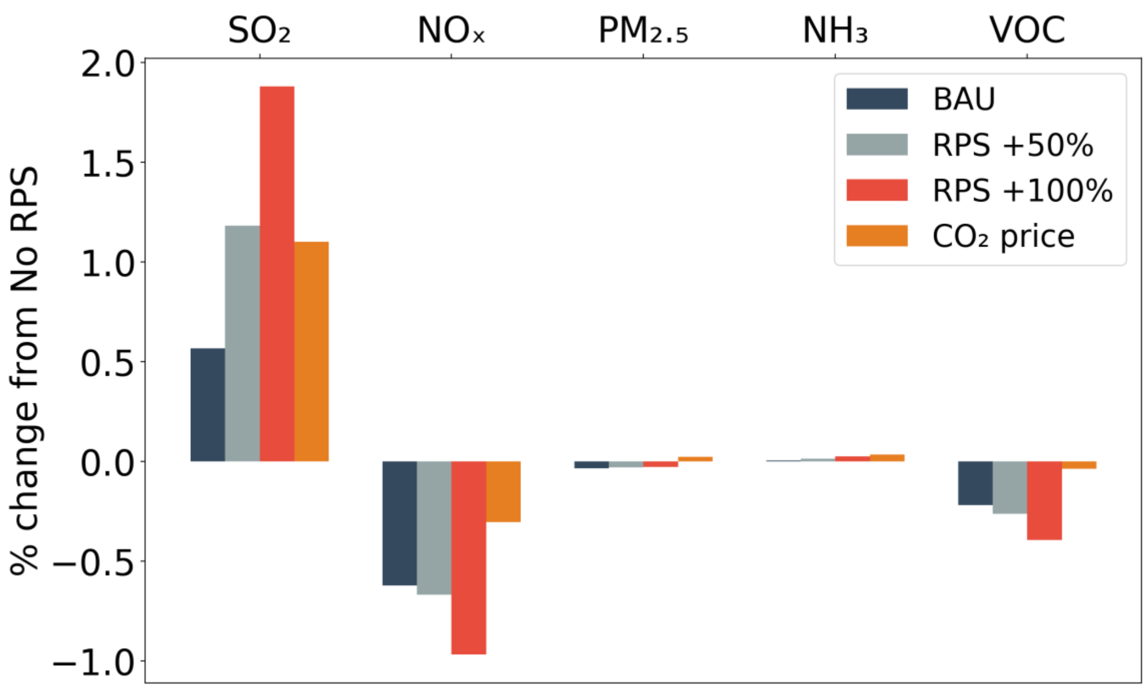

Figure 14: Changes in air pollutant emissions by sector and scenario in 2030 in areas other than the Rust Belt

The spatial distribution of SO_2 and NO_X emission changes is displayed in Figure 15. SO_2 emission decreases are consistent with the location of coal plants along the Ohio river and the Chicago and Detroit areas. Small increases occur in areas outside of the Rust Belt as discussed previously. NO_X emissions also decline along the Ohio river. However, increases take place in urban areas such as Chicago and Detroit as transportation sector NO_X emission increases outweigh decreasing NO_X emissions from electricity production. Areas outside of the Rust Belt show lower emissions consistent with the decrease in transportation sector NO_X emissions shown previously in Figure 14.

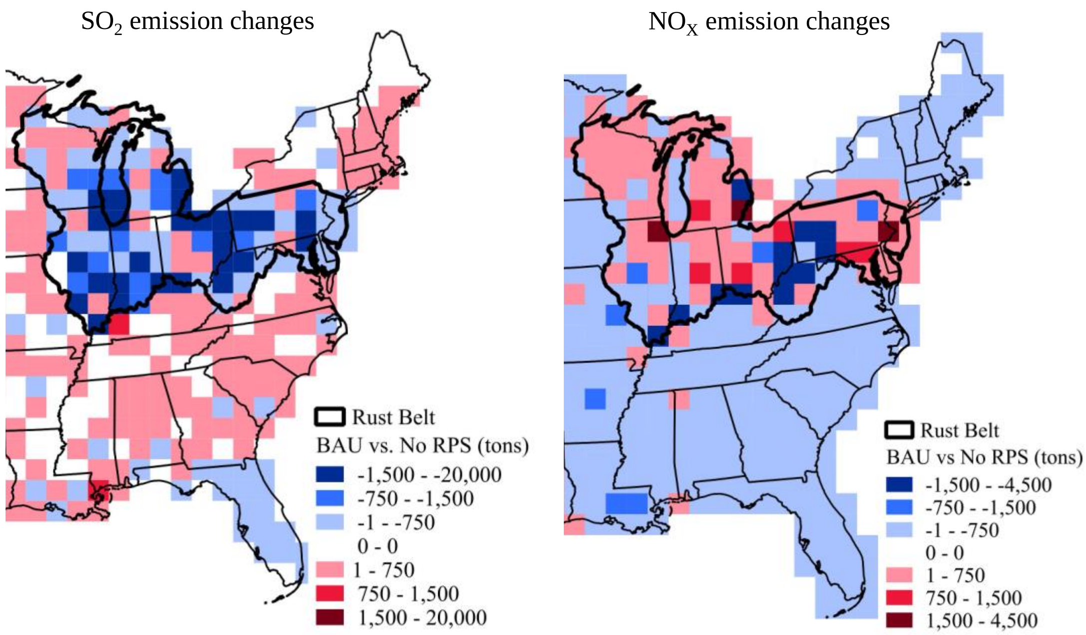

Figure 15: Spatial distribution of SO_2 and NO_X emission changes in the *BAU* scenario relative to *No RPS*

4.1.5 $PM_{2.5}$ concentration effects

The effect of *BAU* emission changes on 2030 concentrations of $PM_{2.5}$ relative to *No RPS* simulated by InMAP is illustrated in Figure 16. $PM_{2.5}$ concentrations exhibit decreases of up to 0.4 μg/m^3. Most of the changes occur within the Rust Belt where the effect is strongest in Maryland, Delaware, Pennsylvania, Indiana, Ohio, and West Virginia. Average population-weighted concentration changes in these states range from -0.14 μg/m^3 in Maryland to -0.1 μg/m^3 in West Virginia. States located downwind of the Rust Belt region also experience improved air quality. Downwind benefits occur in particular for Virginia and New Jersey.

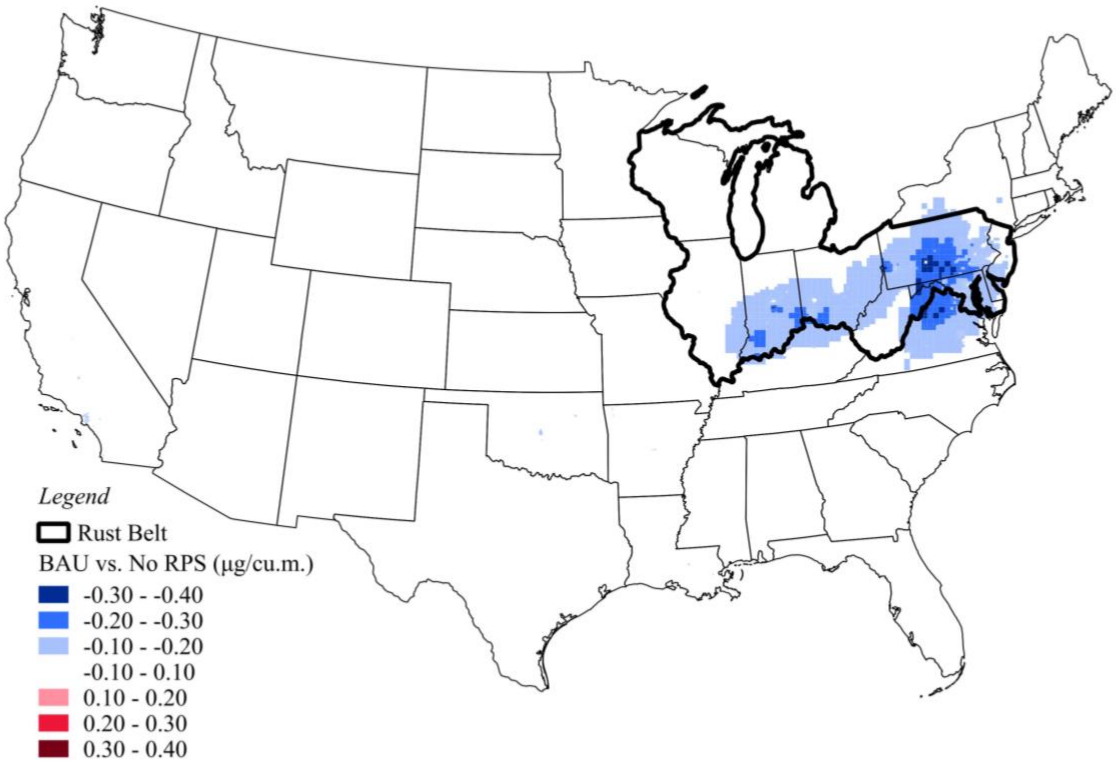

Figure 16: Changes in 2030 $PM_{2.5}$ concentrations (µg/m³) in *BAU* scenario relative to *No RPS*

Under the more stringent climate policies represented by the *RPS +100%* and the *CO_2 price* scenarios, $PM_{2.5}$ concentrations decrease by up to 2 µg/m³ (Figure 17). Population-weighted average concentrations change the most in Maryland, Delaware and Pennsylvania by approximately -0.5 µg/m³ in the *RPS +100%* scenario and between -0.7 and -0.8 µg/m³ in the *CO_2 price* scenario.

Among downwind states outside of the Rust Belt, the largest change under the *RPS +100%* scenario occurs in Virginia (-0.4 µg/m³), followed by New York (-0.2 µg/m³). The *CO_2 price* scenario has similar effects, reducing these concentrations by -0.5 and -0.2 µg/m³ respectively.

The two scenarios show a similar geographic pattern of concentration changes, with carbon pricing leading to somewhat stronger concentration changes in line with its larger effect on SO_2, NO_X, and primary $PM_{2.5}$ emissions.

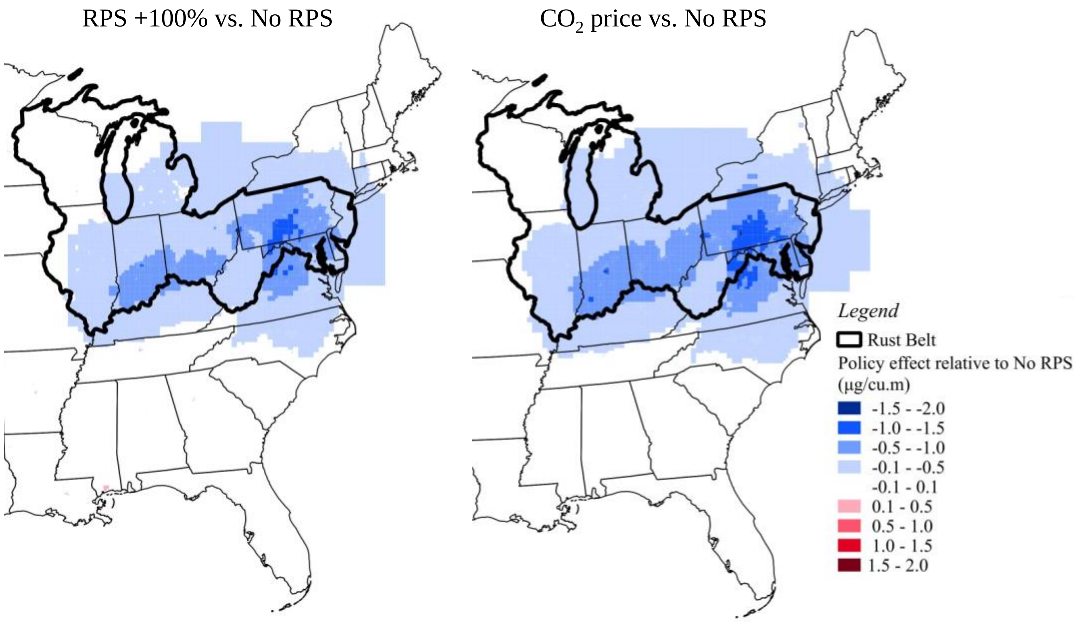

Figure 17: Comparison of 2030 $PM_{2.5}$ concentrations changes in RPS and carbon pricing scenarios

4.1.6 Mortality effects

Premature mortality changes in 2030 by scenario relative to *No RPS* estimated by InMAP are shown in Figure 18. As a percent of total premature mortalities attributed to $PM_{2.5}$ by InMAP, these changes represent reductions that range from 1% for the *BAU* scenario and 6% for the *CO_2 price*. Consistent with changes in emissions and concentrations, mortality reductions are greater in the *CO_2 price* scenario relative to the comparable *RPS +100%* scenario.

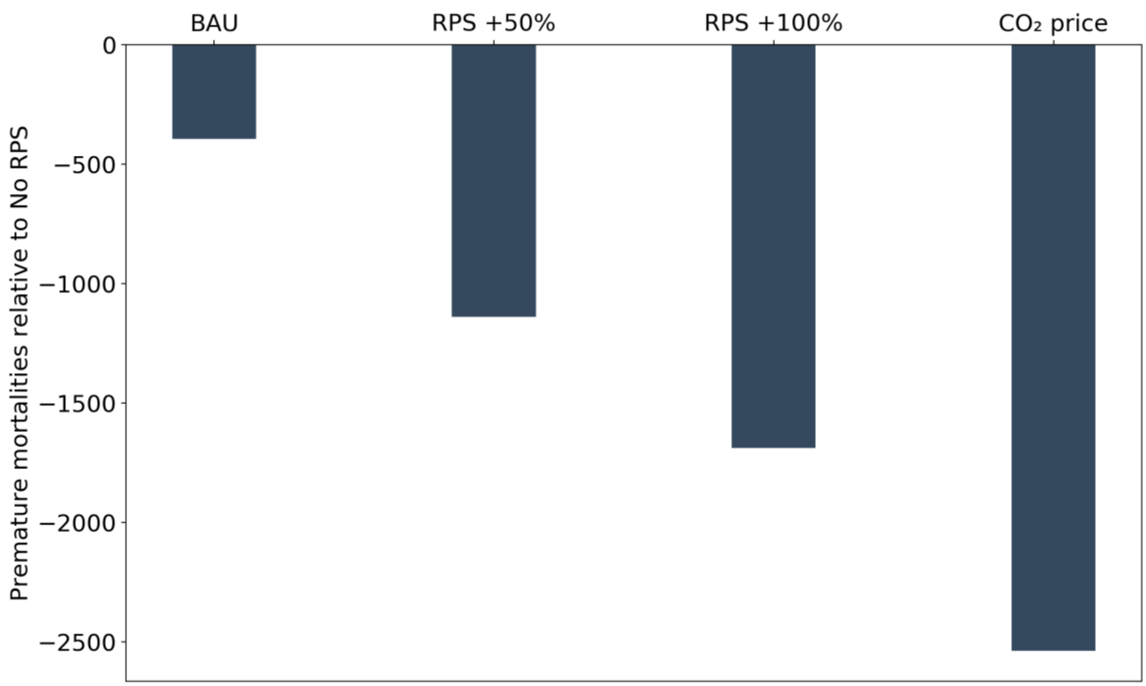

Figure 18: Changes in 2030 premature mortalities in the Rust Belt by scenario relative to *No RPS*

The distribution of avoided premature mortalities across states in the *BAU* scenario is displayed in Figure 19. Pennsylvania is where the *BAU* scenario avoids the most mortalities compared to *No RPS*. This is driven both by the changes in concentrations discussed above as well as by the relatively large state population compared to states with similar concentration changes such as Maryland, Delaware and Indiana. The difference between Pennsylvania and Ohio, which have comparable populations, can be explained by the slightly larger reductions in $PM_{2.5}$ concentrations occurring in Pennsylvania. Among non-Rust-Belt downwind states, New York exhibits a relatively high number of avoided mortalities despite a relatively small change in population-weighted average $PM_{2.5}$ concentrations (-0.06 μg/m^3), due to the state's large population.

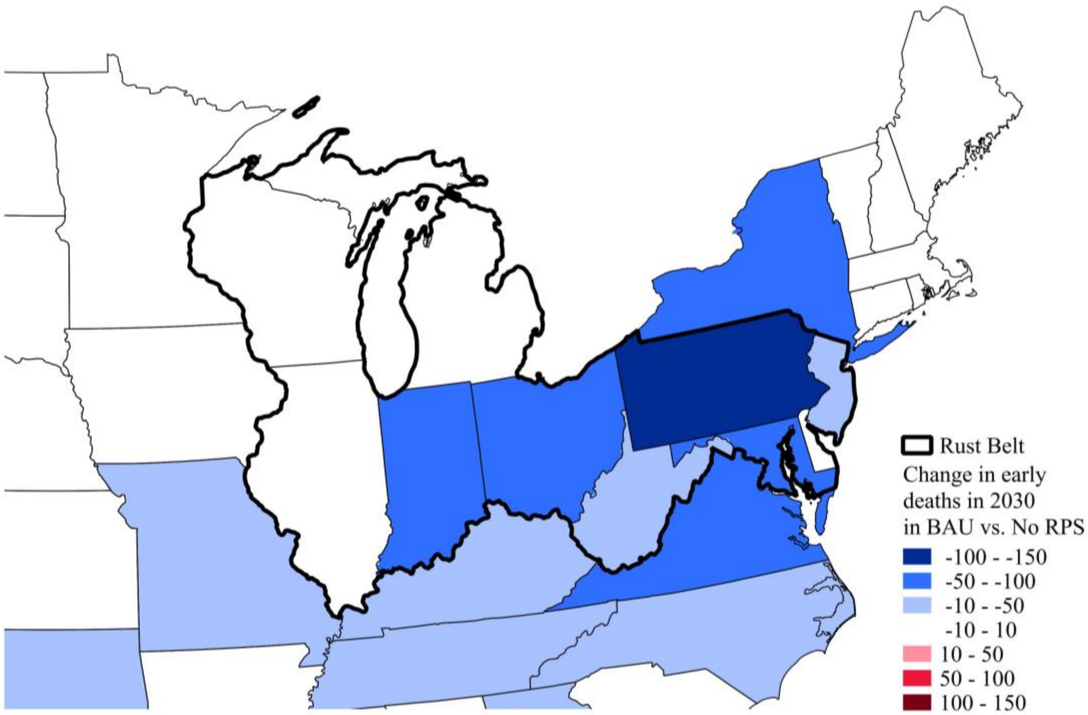

Figure 19: Spatial distribution of changes in 2030 premature deaths under *BAU* relative to *No RPS* by state

4.1.7 Overall costs and benefits

Figure 20 displays total 2030 economic costs and compares them to the CO_2 abatement and avoided mortality benefits. The economic cost of each policy is quantified as a decline in household consumption relative to the *No RPS* scenario. This is equivalent to the economic welfare measure of Hicksian Equivalent Variation, which represents the amount of income needed to compensate consumers for welfare losses suffered as a result of a policy (Paltsev *et al.* 2005). As discussed in Chapter 3, the air quality and climate benefits are estimated using assumptions for the VSL (using a 2006 VSL of \$7.4 million scaled to 2030 based on GDP projections for each scenario as described in Section 3.5) and SCC (using a 2030 SCC of \$50/tCO_2 as described in Section 3.6). Costs and benefits for each scenario are represented as the additional costs or benefits in each scenario relative to the *No RPS*.

The *BAU* scenario shows that, in comparison to costs, climate benefits are somewhat lower while health benefits are somewhat higher. Notably, air quality co-benefits alone exceed policy costs across policy scenarios. As the stringency of the RPS increases, the gap between total benefits (climate and air quality related) and costs expands. The difference between benefits and costs rises from \$2.8 billion in the *BAU* to \$10 in the *RPS +50%* scenario and \$15 billion in the *RPS +100%* scenario. This pattern is driven by the difference between marginal air quality co-benefits and marginal costs. The marginal air quality co-benefits (the incremental co-benefit incurred by moving from the *No RPS* to the *BAU*, from the *BAU* to the *RPS +50%*, and from the *RPS +50%* to the *RPS +100%* scenario) are larger than the marginal costs across all RPS scenarios tested. In other words, the air quality co-benefits rise more rapidly than policy costs as RPS stringency is increased.

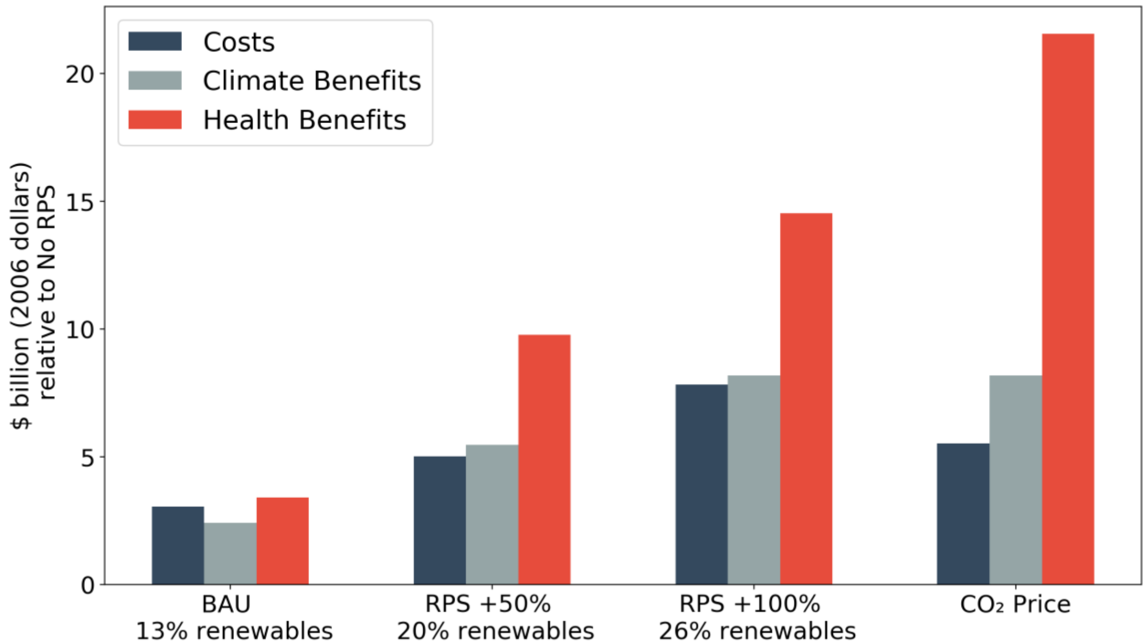

Figure 20: Costs and benefits in 2030 by scenario relative to *No RPS*

The *CO_2 price* leads to higher health benefits than the comparable *RPS +100%* scenario. The majority of the difference is due to the increase in transportation sector emissions in the RPS scenario discussed previously. Another factor driving the higher benefits in the *CO_2 price* scenario is the greater reduction of coal generation, which outweighs the higher level of gas burning in the *CO_2 price* scenario due to the relatively high air pollutant emissions intensity of coal relative to gas. Economic costs under the *CO_2 price* are 29% lower than the *RPS +100%* scenario costs as the economy-wide carbon price incentivizes the least-cost CO_2 abatement options while the *RPS +100%* scenario predetermines the way in which CO_2 emissions are reduced by imposing a renewable energy requirement.

The economic costs of the modeled climate policies are relatively small in percentage terms. In the *BAU* case, the overall household consumption loss is equivalent to a decrease of 0.1% relative to *No RPS*. Under the more stringent *RPS +50%* and *RPS +100%* scenarios, the consumption loss is equivalent to a drop of 0.1% and 0.2% relative to *No RPS*. The *CO_2 price* results in a consumption loss of 0.15%.

The estimated 2030 health benefits for the three *BAU*, *RPS +50%*, and *RPS +100%* scenarios correspond to climate co-benefits of \$68, \$87, and \$86 for each ton of CO_2 abated respectively. The *CO_2 price* results in a health co-benefit of \$128/tCO_2. Measured in reference to renewable generation, the three RPS scenarios result in 2030 benefits of \$0.06, \$0.09, and \$0.1 per kWh of new renewable generation.

4.2 Uncertainty analysis

4.2.1 Uncertainty in the Concentration Response Function

One of the main uncertainties in this study concerns the choice of CRF for estimating the impact of concentration changes on premature mortalities. Figure 21 compares mortality change effects by scenario for the CRF derived by Krewski *et al.* (2009) (representing a 7.8% increase in premature mortalities for each 10 $\mu g/m^3$ increase in concentrations), which was used for the Base Case results presented above, with the CRF in Lepeule *et al.* (2012) (representing an increase in premature mortalities of 14%). Using the latter CRF results in an approximately 80% increase in the estimated changes in premature mortalities.

Additionally, Figure 21 displays the 95% Confidence Interval estimated for each CRF. The upper and lower confidence limits imply an uncertainty around the estimated changes of approximately 45-50% for both CRFs.

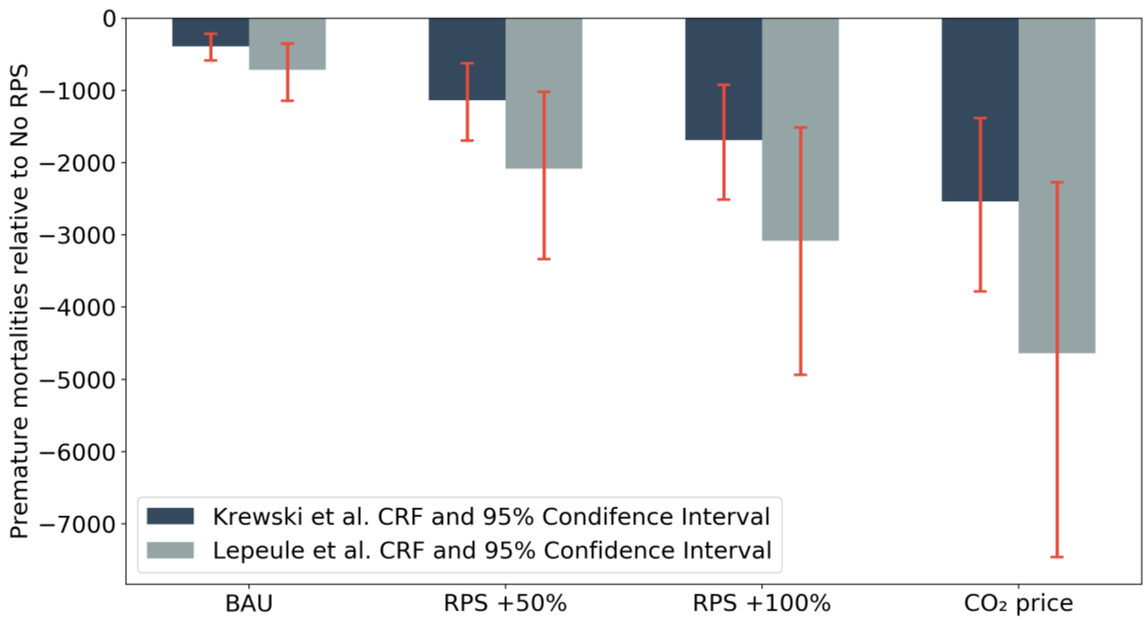

Figure 21: Changes in premature mortalities by scenario relative to *No RPS* (central estimates and 95% Confidence Intervals) for two different CRF assumptions

4.2.2 Uncertainty in the Value of Statistical Life

Another key assumption is the choice of VSL. Figure 22 shows estimated policy benefits relative to *No RPS* including error bars based on the full distribution of 2006 VSL values reported by the EPA (2014). The 2006 VSL estimates compiled by the EPA cover a wide range from \$0.85 to 19.8 million (with a central estimate of \$7.4, which is behind the Base Case results of this thesis). In accordance with this wide range, the estimated health benefits vary dramatically from the central estimate from approximately -90% to +270%.

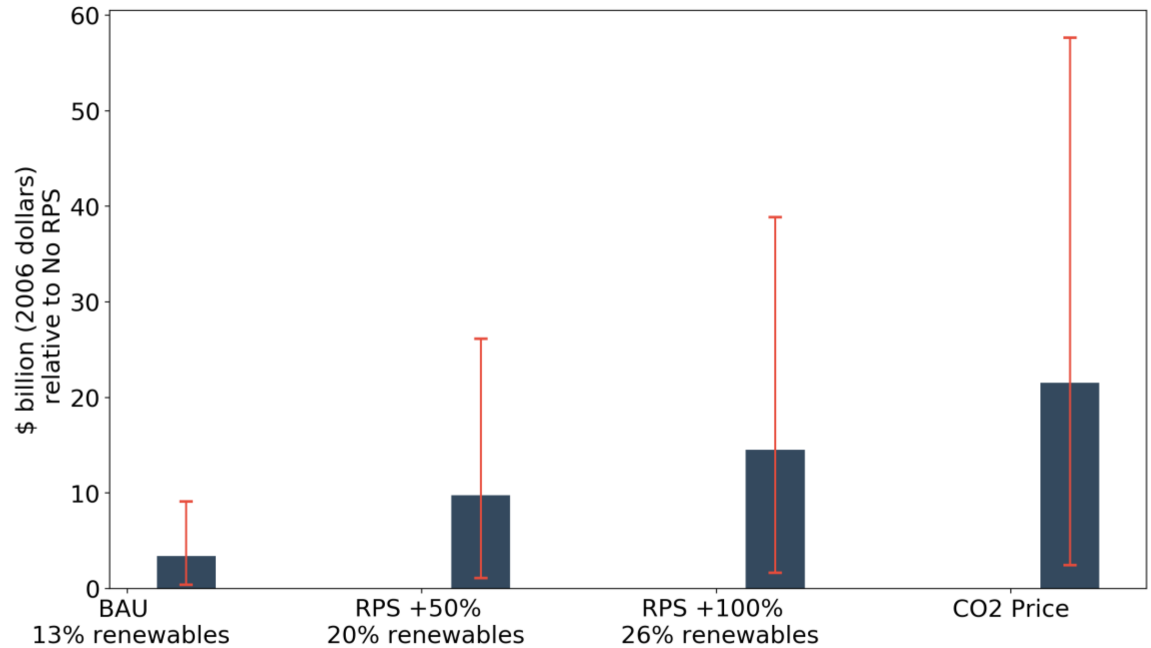

Figure 22: Health benefits by scenario relative to *No RPS* including VSL uncertainty

4.2.3 Uncertainty in wind power costs

Uncertainty also surrounds the cost of different power generating technologies. In particular, the cost of renewables may be subject to change in the future. NREL (2017) projected the capital cost of wind turbines to change by -20% to +16% from 2015 to 2030 depending on technology and wind resource categories. To test the sensitivity of these results to alternative wind plant costs, this thesis presents results for two additional cases: a Low Cost case (15% lower capital cost) and a High Cost case (15% higher capital cost).

Figure 23 presents the sensitivity of net benefits (health benefit – cost) to the alternative cases across policy scenarios. In the High Cost case, the increase in wind costs raises the total economic cost by between 46% in the *BAU* and 13% in the *CO_2 price* scenario. The Low Cost case lowers overall cost by 3% for the *BAU* to 15% for the *RPS +100%*. With regard to the relative costs of carbon pricing and RPSs, the CO_2 price scenario is 24% cheaper than the *RPS +100%* in the Low Cost case (as opposed to 29% under Base Case assumptions). These alternative cases also slightly influence the magnitude of health benefits. Overall, the net benefit of the *BAU* scenario relative to *No RPS* rises from $0.4 billion to $1.4 (+400% approximately) in the Low Cost scenario and decreases from $0.36 to -$0.9 billion (-240% approximately) in the High Cost scenario, suggesting a considerable sensitivity of these results to wind power cost changes.

However, the uncertainty stemming from the choice of CRF and VSL, as represented by the error bars, outweighs the uncertainty in wind power costs. The error bars shown combine the full range of VSL alternatives mentioned previously with the 95% Confidence Interval of the CRF estimated by Krewski *et al.* (2009).

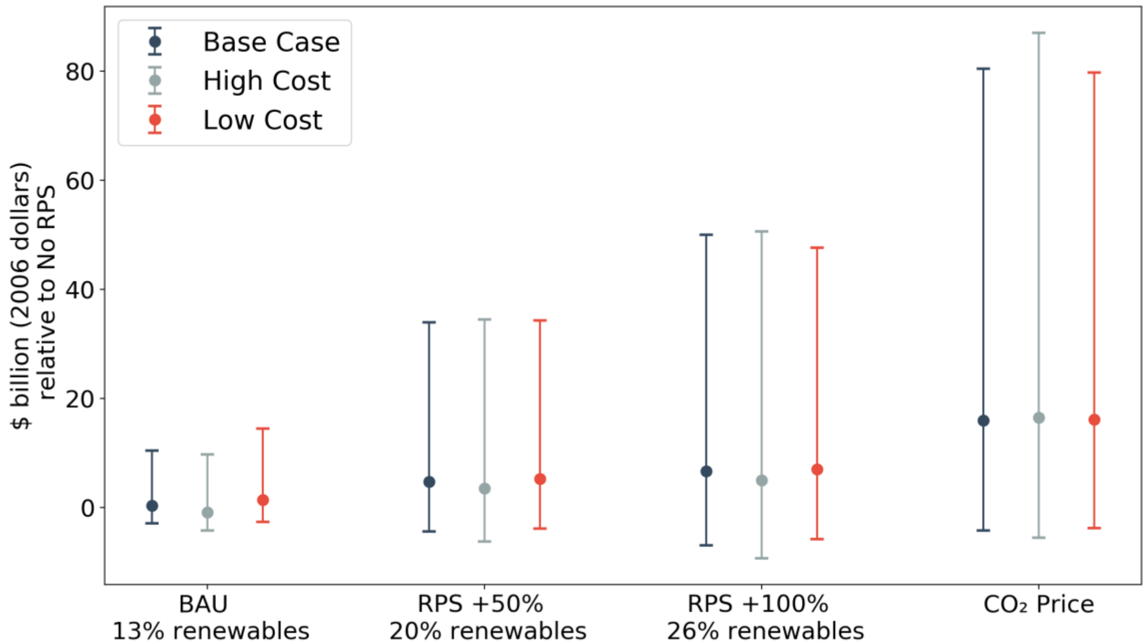

Figure 23: Net health benefits (health benefits – costs) by scenario and alternative wind power cost cases. Error bars denote combined VSL and CRF uncertainty**.** The CRF is based on Krewski *et al.* (2009)

4.2.4 Uncertainty in the Social Cost of Carbon

To quantify uncertainty related to the quantification of climate benefits, I consider alternative assumptions for the discount rate used to translate long-term climate benefits into benefits in 2030. In addition to the central value of \$50/tCO_2 (representing a discount rate of 3%) used in the Base Case results, I use alternative SCCs assuming 5% and 2.5% discount rates of \$16/tCO_2 and \$73/tCO_2, as recommended by the Interagency Working Group on Social Cost of Greenhouse Gases (IWG, 2016). Finally, I use the IWG's recommended "High Impact" SCC of \$152/tCO_2, which represents the 95th percentile of the SCC probability distribution. This value is meant to represent the magnitude of damages calculated by Integrated Assessment Model studies in the case of low-probability high-impact climate outcomes as discussed in Chapter 2.

Figure 24 presents the sensitivity of the estimated climate mitigation benefits to these alternative SCC assumptions. The estimated benefits across scenarios vary approximately by a factor of three, in proportion to the variation in the assumed SCC values.

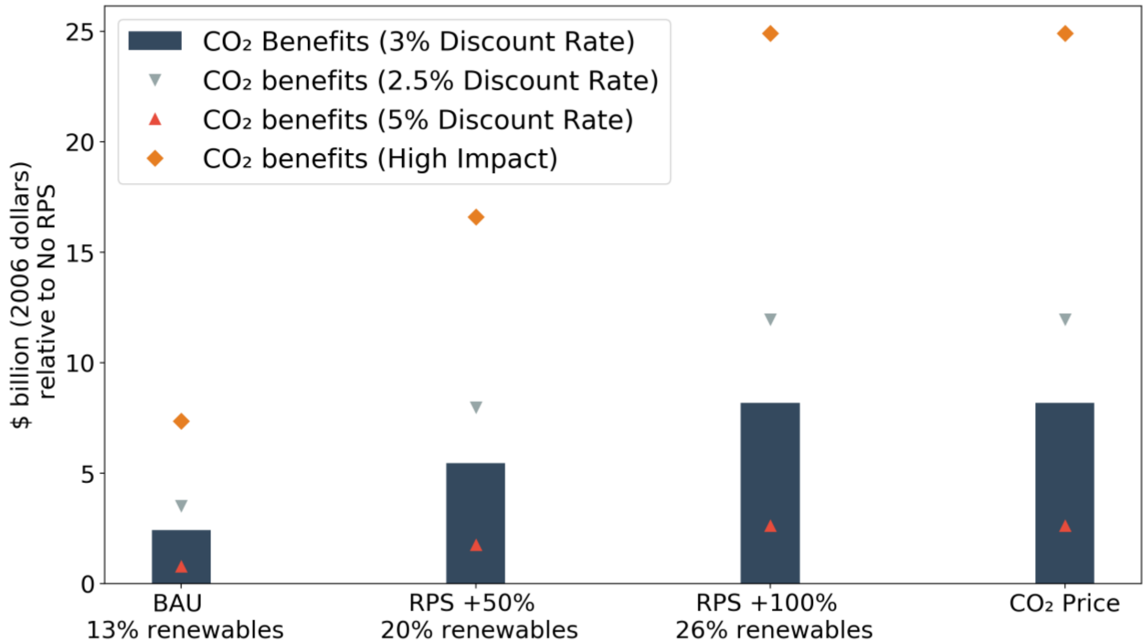

Figure 24: Climate mitigation benefits by scenario including alternative SCC assumptions

4.3 Discussion

This section compares key results from this thesis to the findings of previous studies. It discusses the impact of an RPS on the power mix, the estimated health benefit, and the estimated economic costs.

4.3.1 RPS impacts on the power mix

The RPS-driven power mix changes estimated by USREP represent a plausible picture of potential impacts. Even though USREP does not represent intra-day power market dynamics due to its annual resolution, the RPS power mix impacts presented above are comparable to those derived from modeling that simulates intra-day power market operations (Buonocore *et al.*, 2016; Mai *et al.*, 2016).

An important factor in modeling renewable energy policy is the intra-day variation of renewable energy generation, which may require more frequent cycling of coal and gas power plants (Kumar *et al.*, 2012; Perez-Arriaga and Batlle, 2012; Van Den Bergh, Delarue and D'haeseleer, 2013). Coal plants generally have higher cycling costs compared to gas plants (Kumar *et al.*, 2012), suggesting that, all else being equal, increasing penetration of variable renewable energy may favor gas over coal.

Capturing these dynamics, Mai *et al.* (2017) estimated that existing RPSs across the U.S. result in an almost equal displacement of coal and natural gas generation in 2030, which is in line with the Base Case results derived for the Rust Belt from USREP. With regard to renewable generation, the authors also found that hydro, biomass and geothermal play a small role in meeting RPS requirements, as also implied by the base case results presented here. However, Mai *et al.* (2017) estimated that solar contributes approximately as much as wind toward RPS requirements in 2030, which stands in contrast to results derived by USREP, which does not represent solar.

The Base Case results of this thesis are also comparable to findings by Buonocore *et al.* (2016). Using results from a detailed power dispatch model, the authors similarly estimated that the addition of wind and solar power displaces both coal and gas from the power mix in the Rust Belt region. The study found that wind power displaces mainly coal power when installed in the Chicago area, Cincinnati area, or Northern Ohio and mainly gas power when installed in Eastern Pennsylvania or Southern New Jersey. Solar power was estimated to result in more displacement of gas relative to coal for most modeled areas in the Rust Belt.

It also bears mentioning that more frequent cycling of fossil fuel power plants resulting the penetration of variable renewable energy may also impact the emission factors of these power plants. However, the exclusion of this effect in this thesis is not expected to meaningfully bias the results (Gross *et al.*, 2006; Göransson and Johnsson, 2009; Fripp, 2011).

4.3.2 Health benefits

The central estimate of health benefits for existing Rust Belt RPSs (the *BAU* scenario) of $0.06/kWh compares closely to the $0.05/kWh estimated by both the prospective modeling by Mai *et al.* (2017) and the retrospective analysis by Wiser *et al.* (2017). While direct comparisons are made difficult by differences in air pollution modeling methodologies and different timeframes for which impacts were estimated, the somewhat higher air quality benefit estimated here is consistent with the relatively high emission intensity of the Rust Belt power sector discussed in Chapter 2 Section 2.2.

On a per ton of CO_2 abated basis, the estimated central co-benefits fall in the range of co-benefit estimates reported by Nemet, Holloway and Meier (2010) of \$2–196/tCO_2. The results of this thesis are also in the same order of magnitude as the \$148/tCO_2 and \$80/tCO_2 estimates by Thompson *et al.* (2016) for a regional Clean Energy Standard (CES) and a regional cap-and-trade in the same part of the U.S. (the study covered New England states in addition to the states contained in this thesis' Rust Belt region). The central carbon pricing co-benefit estimated in this thesis of \$128/tCO_2 is higher in line with the fact Thompson *et al.* (2016) include New England states, where fossil fuels play a smaller role in the power mix (EIA, 2017a).

In comparison, Thompson *et al.* (2014) estimated somewhat higher co-benefits for a Clean Energy Standard and a carbon price at the national level of \$254/tCO_2 and \$140/tCO_2. The somewhat higher carbon pricing co-benefit is in part due inclusion of ozone-related benefits and morbidity effects by Thompson *et al.* (2014), which are excluded in this thesis. Another factor explaining the difference is this thesis' use of a lower CRF coefficient for the calculation of the mortality impact of $PM_{2.5}$ concentrations.

Despite the differences in assumptions, the disparity between the results of this thesis and those found in Thompson *et al.* (2014) and Thompson *et al.* (2016) is reasonable. This comparison is consistent with Tessum, Hill and Marshall (2017) in suggesting that results obtained from a Reduced Complexity Model (RCMs), used in this thesis, and state-of-the-art Chemical Transport Models (CTMs), used in the studies discussed, are comparable. The relative congruence of these results may be explained by the fact that the climate policies modeled in this thesis are relatively modest, allowing the linear assumptions within RCMs, which the models use to simplify non-linear atmospheric chemistry, to capture marginal changes with relative accuracy. An important area for future work would be a direct comparison between RCMs and CTMs tested under the same policy scenarios and assumptions as discussed in Chapter 5.

The results of this thesis are also congruent with Thompson *et al.* (2014) with regard to the impact of alternative climate policy stringency levels on health co-benefits. This thesis shows that increasing RPS stringency by 50% from the *BAU* scenario to the *RPS +50%* scenario increases health co-benefits by 27% from \$68/tCO_2 to \$87/tCO_2. Similarly, Thompson *et al.* (2014) showed that doubling the stringency of the modeled cap-and-trade increased health co-benefits by approximately 40%. The implication of these results is that health co-benefits can exhibit considerable sensitivity to the chosen policy stringency level.

Additionally, this thesis' results, similarly to the findings of Thompson *et al.* (2014) suggest that at certain relatively modest policy stringency levels, health co-benefits may exhibit increasing, rather than diminishing, returns in response to an increase in CO_2 reductions. The marginal health co-benefit of the *RPS +50%* scenario (measured by dividing the incremental increase in health co-benefits by the incremental increase in CO_2 abatement) is estimated here to be \$102/tCO_2, higher than the \$68/tCO_2 marginal co-benefit of the *BAU* scenario, which is consistent with the findings of Thompson et al. (2014) for a national cap-and-trade. The returns to scale may change, however, at higher policy stringency levels. The marginal health co-benefit of the *RPS +100%* scenario is estimated to be \$84/tCO_2, exhibiting diminishing returns.

4.3.3 Economic costs

This thesis estimates RPS costs to be higher than estimates by electricity system studies (Mai *et al.* 2017) due to the inclusion of economy-wide costs. Total RPS-driven consumption losses equate to costs of around \$0.05/kWh across all three RPS scenarios compared to ±\$0.0075/kWh estimated by Mai *et al.* (2017) where the authors calculated only electricity system costs. This large discrepancy demonstrates the value of employing general equilibrium models to the study of the economic impacts of renewable energy policy.

Energy policies can impact large sections of the economy through their impact on the electricity price and the prices of different fuels such as coal and gas, which influences the amount of fuel consumption in other sectors such as manufacturing or residential heating. Their effects are not only far-reaching but can also run in opposite directions. Renewable energy policies can, on the one hand, increase the price of electricity for residential consumers but, on the other, lower the price of natural gas paid by the same consumers (by reducing natural gas consumption in the electricity sector). The use of general equilibrium modeling allows the estimation of the final net effects on household consumption.

On the basis of consumption losses, this thesis finds that an RPS is more expensive than a CO_2 price (implemented here as a cap-and-trade), in line with previous findings in the literature. The consumption loss in the *RPS +100%* scenario compared to the *BAU* was estimated to be approximately twice as large as the consumption loss in the *CO_2 price* scenario relative to *BAU*. In comparison Rausch and Mowers (2014) estimated that a national RPS is four times costlier than a cap-and-trade policy. The difference can be in part explained by the recent decline in wind power costs captured in the cost data used for this thesis.

The finding of this thesis that health benefits tend to exceed policy costs is congruent with previous literature comparing economic costs with health co-benefits (West *et al.*, 2013; Thompson *et al.*, 2014, 2016; Shindell, Lee and Faluvegi, 2016).

Chapter 5

Conclusions

Renewable energy policy is on the agenda of lawmakers across the United States and beyond as decision makers navigate challenges such as climate change and air pollution as well as opportunities such as the increasing cost-competitiveness of renewable energy technologies. A particular challenge for lawmakers in the U.S. is the design of Renewable Portfolio Standards (RPSs), exemplified by the large number of bills proposing to strengthen, weaken, or otherwise modify existing statutes as discussed in Chapter 2. To aid decision making in this area, this thesis has sought out to quantify the air quality co-benefits of RPSs under several stringency levels. The focus of this assessment has been on the Rust Belt, where air quality effects are likely to be particularly relevant due to the severity of air pollution in this region. This thesis has compared these co-benefits to what may be considered the primary benefits of these climate policies: their contribution toward mitigating climate change, as well as their economic costs. It has also compared RPSs to carbon pricing to improve understanding of the relative costs and benefits of alternative climate policy choices.

This chapter discusses the implications for policy making that can be drawn from the results presented in Chapter 4. Finally, I discuss the limitations of this thesis and avenues for future work.

5.1 Policy Implications

5.1.1 Implications for the costs and benefits of Renewable Portfolio Standards

The results of this thesis suggest that the air quality health co-benefits of RPSs in the Rust Belt can be substantial, large enough to warrant their evaluation and consideration along with other policy impacts in regulatory assessments and cost-benefit analyses. In relative terms, the 2030 air quality co-benefits of the modeled RPSs were estimated here to be on par with or larger than the 2030 climate benefits of these policies (Chapter 4 Section 4.1.7). Air quality co-benefits may a particularly salient for climate policy making as they occur locally, thus directly benefitting the implementing jurisdiction. Additionally, these co-benefits materialize in the near term, relative to the more long-term climate mitigation benefits of climate policy (Nemet, Holloway and Meier, 2010).

The assessed air quality co-benefits were also shown to be generally higher than policy costs (Sections 4.1.7 and 4.2), suggesting that air quality improvements alone may justify RPS implementation. This result, however, is sensitive to uncertainty in several key modeling parameters. In particular, the choice of the assumed Value of Statistical Life (VSL) and the statistical uncertainty associated with the Concentration Response Function (CRF) were the largest source of uncertainty among the assumptions tested.

The sensitivity of the results underscores the importance of quantifying uncertainty. Representing sensitivity to individual assumptions is likely to be particularly pertinent for modeling choices embedded with value judgments such as the choice of VSL or the discount rate used in the calculation of the present value of future climate change benefits. For such assumptions, quantifying their associated uncertainty separately from other assumptions allows audiences to draw their own conclusions on the basis of unique individual values.

This thesis also has implications for one of the main design elements of RPS policies: the stringency of the renewable energy requirement. As the 2030 RPS requirement is increased successively from 6% (an outcome assuming that all Rust Belt RPSs are discontinued as of 2015) to 13% (representing the 2030 level of stringency of existing RPSs), 20%, and 26%, air quality co-benefits increase more rapidly than policy costs under the central, Base Case assumptions of this thesis (Sections 4.1.7 and 4.2). As a result, the net benefits (climate and air quality benefits minus costs) increase as the stringency of the RPS rises. This result indicates that, all else equal, increasing the Rust Belt RPSs from their current levels may lead to greater economic efficiency (higher net-benefits).

It must be acknowledged that by focusing on economic efficiency, this cost-benefit analysis has excluded considerations of equity. However, the air quality effects of renewable energy policy may also have important implications for relative policy impacts on different segments of the population. Non-whites and people in poverty have been estimated to be more severely impacted by $PM_{2.5}$ concentrations than other demographic groups (Mikati *et al.*, 2018). Air pollution mitigation may therefore benefit these groups more than others. Equity effects will critically depend on how policies impact individual fossil fuel power plants, which could be addressed in future research.

5.1.2 Implications for climate policy instrument choice

To guide choices between alternative policy instruments, economists often compare policies on the basis of efficiency (total benefits minus total costs). As discussed in Chapter 2, a common approach for evaluating climate policies is by comparing their economic costs in relation to climate benefits on the basis of dollars-per-ton of CO_2 abated. The results presented above make a case for the inclusion of air quality co-benefits in comparisons between climate policies.

In particular, this thesis showed that a carbon price resulted in greater air quality co-benefits in 2030 than an equivalent RPS for each ton of CO_2 abated (Sections 4.1.7 and 4.2). This result was driven by differences in the way each policy achieved CO_2 reductions. The RPS reduced CO_2 emissions primarily by switching from coal- and gas-fired power generation toward renewables. In contrast, the carbon price achieved most CO_2 abatement by triggering a switch from coal to gas, leading to less coal-fired power generation than the equivalent RPS scenario. Since coal use is the main determinant of air pollution due to its relatively high emission intensity per unit of energy (as discussed in Chapter 2), the CO_2 pricing scenario resulted in greater air quality improvements per ton of CO_2 reduced. This result suggests that, in the timeframe up to 2030, carbon pricing may be more efficient relative to an RPS than previously thought based on dollar-per-ton comparisons that do not consider air quality co-benefits.

However, this thesis does not provide an answer to the question of which of the two policies is more efficient overall. How the total net benefits of these policies compare will be determined by a number of additional factors not assessed in this thesis (further discussed in Section 5.2). In particular, the greater reliance on natural gas under carbon pricing could result additional environmental impacts associated with gas extraction through hydraulic fracturing and gas pipeline leakage of methane emissions. Additionally, a transition toward gas-burning power plants in the short term could lead to a high-carbon lock-in, whereby the newly built gas infrastructure may make it more difficult for states to achieve more stringent long-term climate goals (such as targets to reduce CO_2 emissions by 80 percent by 2050 in Massachusetts, New Jersey, and California) (Erickson *et al.*, 2015)

Equity considerations can also alter the relative merits of each policy. Experience with cap-and-trade systems has raised concerns due to the potential for hot spots: the concentration of air pollution impacts in areas with high-emission-intensity plants that are costly to shut down or upgrade; though specific cap-and-trade designs can minimize such risks (Farber, 2012; Schmalensee and Stavins, 2017).

Ultimately, the choice between alternative policy options may depend to a large extent on the political acceptability that each policy instrument garners in a given jurisdiction (as discussed in Chapter 2 Section 2.2.2). Political acceptability will in part be determined by the political economy characteristics of alternative policies as discussed below.

5.1.3 Implications for the political economy of climate policy

For the purposes of policy making, an important consideration can be the distribution of policy impacts across stakeholders as it can determine the support or opposition stakeholders may mount during the policy making process. Policies can be distinguished by whether the distribution of their impacts is diffuse (exerting a small influence on a large number of stakeholders) or concentrated (exerting a large influence on a small number of stakeholders). The theory of collective action (Olson, 1982) suggests that the smaller the group of impacted stakeholders, the more likely it is for them to organize for or against a proposed policy. The lobbying efforts of a concentrated group of stakeholders can then lead to regulatory capture (Stigler, 1971), bolstering or constraining the political feasibility of policy proposals.

The results presented in Chapter 4 suggest that the costs of RPSs are more diffuse compared to carbon pricing. Across power sector technologies, RPSs increased renewable generation at the expense of both coal and gas. In contrast, carbon pricing affected only coal generation, while having a beneficial impact on gas generation and a neutral impact on renewable generation (Section 4.1.1). In terms of economic output, the *RPS +100%* scenario exerted a wider negative impact across electricity, natural gas and a number of other sectors such as transportation, services, and agriculture, while the impact of the *CO_2 price* was more concentrated in the coal sector (Chapter 4 Section 4.1.2). The RPS scenario also had a greater impact on consumers as measured by consumption losses (Section 4.1.7). These results are congruent with the observation by Rabe (2018) that the costs of RPSs are more disguised compared to the costs of carbon pricing.

The benefits of climate change and air pollution mitigation are relatively diffuse for both RPSs and carbon pricing. Across policy types, these benefits accrue to large portions of the general public. The spatial distribution of the reductions in $PM_{2.5}$ were shown to be relatively similar for the *RPS +100%* and the *CO_2 price* scenarios (Chapter 4 Section 4.1.5).

However, this thesis also showed that RPSs and carbon pricing can confer relatively concentrated economic benefits on specific industries. The *RPS +100%* scenario doubled 2030 renewable generation from *BAU* (Section 4.1.1). Under the CO_2 price scenario, natural gas power generation rose by 76% and gas extraction increased by 9% in 2030 (Sections 4.1.1 and 4.1.2). This partially explains why oil and gas companies have urged governments to implement carbon pricing (UN, 2015).

This carbon pricing result is not necessarily generalizable as the natural gas industry may not always be incentivized to support this policy. The benefit that the gas industry extracted from the modeled *CO_2 price* scenario was the increase in market share relative to coal. Yet, if coal becomes phased out of the power mix in the future, carbon pricing would cease to have this competitiveness advantage for natural gas. The policy would function merely as an additional expense, potentially provoking future opposition from owners of natural gas infrastructure. This prospect may dampen the willingness (both future and current) of this industry to support carbon pricing.

In summary, the results of this thesis have demonstrated that RPSs impose costs that are relatively diffuse in comparison to carbon pricing (and confer benefits that may be more concentrated). Viewed from the lens of collective action, these results partially explain why carbon pricing has been less politically successful than RPSs. However, this thesis has shown that implementing an RPS instead of a CO_2 price can come at the expense of air quality (as implementing a carbon price was estimated to result in better air quality than strengthening existing RPSs).

One partial solution for policy makers seeking to enhance the political tractability of carbon pricing, suggested by the theory of collective action, is to direct the revenues raised from the sale of CO_2 permits toward certain concentrated constituencies by, for example, funding clean energy research or deployment subsidies. Another approach is the use of revenues for the direct compensation of affected industries (Jenkins and Karplus, 2016) or the free allocation of CO_2 permits under cap-and-trade, a common design feature for such systems currently in existence (Grubb, Hourcade and Neuhoff, 2014).

5.1.4 Implications of economic spill-over effects on policy evaluation

This thesis also showed that RPS policies result in wide-ranging effects, which have implications for both their total benefits and total costs. While the modeled RPS policies reduced electricity sector emissions of CO_2 and primary air pollutants as expected, they led to increases in transportation sector emissions by increasing electricity prices and discouraging electric vehicle adoption (Sections 4.1.3 and 4.1.4). While this leakage effect was relatively small for the policies modeled, it could be expected to intensify under more stringent RPS policies. Emissions leakage has implications both for the efficiency of RPS policy (partially offsetting overall benefits) and for equity (altering the spatial distribution of emissions by, for example, resulting in higher NO_x emissions in cities).

The potential existence of such emission leakage effects makes the case for an economy-wide approach to climate policy. Policy making can address emission leakage by complementing RPSs with other policies such an economy-wide carbon price. As shown in Sections 4.1.3 and 4.1.4, adding an economy-wide carbon price to existing RPS policies in the Rust Belt (the *CO_2 price* scenario) partially offsets the emission leakage in the transportation sector.

Assessments of spill-over effects also have implications for the evaluation of policy costs. As presented in Section 4.3.3, the total economic costs of RPSs in the Rust Belt were estimated to be an order of magnitude higher than estimates in previous literature based on electricity system modeling. The economy-wide effects of RPS policies demonstrate the additional value general equilibrium modeling can contribute to renewable energy policy assessments performed using sector-specific electricity system models.

5.2 Future Work

While the use of general equilibrium modeling offers the advantage of capturing economic spill-over effects of RPS policies, it introduces the disadvantage of representing the electricity sector in a top-down fashion, thus omitting operational details such as intra-day power dispatch decision-making (as discussed in Chapter 4 Section 4.3.1). Recent modeling work has demonstrated the possibility of leveraging the advantages of both approaches through hybrid approaches that iteratively combine both types of models (Rausch and Mowers, 2014; Tapia-Ahumada *et al.*, 2014). Though general equilibrium modeling provides important exploratory insight regarding climate policy impacts, the use of hybrid approaches to represent the effect of variable renewable energy on intra-day power markets will be a valuable extension.

Another uncertainty not quantified in this thesis is the structural uncertainty associated with the choice of air pollution model for simulating the chemical transformation and transport of air pollutants in the atmosphere. A key area for future work will be the application of state-of-the art Chemical Transport Models (CTMs) alongside the type of Reduced Complexity Model (RCM) used in this thesis. As RCMs use linear relationships to simplify the real-world non-linear chemical relationships that determine pollution formation, it will be important to test how their results compare to results derived from CTMs. Future efforts to assess climate policies can benefit from an understanding of which types of policies can be robustly tested using RCMs and which types of policies require the use of the more computationally expensive CTMs. The policy scenarios presented in this thesis offer a suitable set of test cases, as they have been designed to represent realistic policy options being considered by states in the Rust Belt region.

While this thesis has attempted to quantify economic costs in a comprehensive manner (by using an economy-wide model to estimate the total consumption losses resulting from policy), it has omitted a number of benefits that may be conferred by renewable energy policy, which could be addressed by future research. These include potential abatement of ground-level ozone (discussed in Chapter 2 and modeled by Thompson *et al.*, 2014), mitigation of climate penalty effects on $PM_{2.5}$ concentrations (discussed in Chapter 2 and modeled by Garcia-Menendez, Saari, Monier, and Selin, 2015), reduction in morbidity cases resulting from lower $PM_{2.5}$ concentrations (discussed in Chapter 2 and modeled by Thompson *et al.*, 2014), decrease in water consumption (Mai *et al.*, 2016), or the mitigation of environmental impacts from coal mining (OTA, 1979) and natural gas extraction (EPA, 2016). It bears mentioning that also not quantified in this work are potential negative environmental impacts of renewable technologies. These may include CO_2 emissions associated with renewable technology manufacturing, which does not meaningfully alter the relative climate benefit of wind or solar power compared to coal or gas (EIA, 2015), land use impacts (Denholm *et al.*, 2000), or impacts on wildlife (AWWI, 2014).

Future work may also apply improvements to several additional modeling choices. With regard to the calculation of air pollution related mortalities, recent literature has shown that the mortality response is non-linear with respect to the level of $PM_{2.5}$ concentrations (Vodonos, Awad and Schwartz, 2018), showing a stronger response at low concentrations and vice versa. The use of non-linear concentration response would represent an improvement to the linear assumption used in this thesis. Another future extension would be the inclusion of the Social Cost of Methane (Marten and Newbold, 2012) in estimating the climate effects of different climate policies to explore the implications of the future role of natural gas in the power mix. Finally, the use of economic models containing a higher spatial resolution would allow the study of climate policy impacts at the state, rather than regional, level to further guide specific policy choices facing state-level lawmakers.

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