Health Co-Benefits of Sub-National Renewable Energy Policy in the U.S.

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—Ronald G. Prinn and John M. Reilly,
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Health co-benefits of sub-national renewable energy policy in the US

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Abstract

State and local policy-makers in the US have shown interest in transitioning electricity systems toward renewable energy sources and in mitigating harmful air pollution. However, the extent to which sub-national renewable energy policies can improve air quality remains unclear. To investigate this issue, we develop a systemic modeling framework that combines economic and air pollution models to assess the projected sub-national impacts of Renewable Portfolio Standards (RPSs) on air quality and human health, as well as on the economy and on climate change. We contribute to existing RPS cost-benefit literature by providing a comprehensive assessment of economic costs and estimating economy-wide changes in emissions and their impacts, using a general equilibrium modeling approach. This study is also the first to our knowledge to directly compare the health co-benefits of RPSs to those of carbon pricing. We estimate that existing RPSs in the 'Rust Belt' region generate a health co-benefit of \$94 per ton CO_2 reduced (\$2-477/tCO_2) in 2030, or 8¢ for each kWh of renewable energy deployed (0.2–40¢ kWh^{-1}) in 2015 dollars. Our central estimate is 34% larger than total policy costs. We estimate that the central marginal benefit of raising renewable energy requirements exceeds the marginal cost, suggesting that strengthening RPSs increases net societal benefits. We also calculate that carbon pricing delivers health co-benefits of \$211/tCO_2 in 2030, 63% greater than the health co-benefit of reducing the same amount of CO_2 through an RPS approach.

1. Introduction

Policies that address climate change can, as a co-benefit, improve air quality (Smith *et al* 2014). In the US, air pollution continues to harm human health despite improvements in air quality over the past decades (EPA 2018a). In 2016, ~93 000 premature deaths and ~1 600 000 years of life lost were attributed to ambient concentrations of fine particulate matter ($PM_{2.5}$) (Institute for Health Metrics and Evaluation 2017), the deadliest form of air pollution (Dockery *et al* 1993, World Health Organization 2006).

Air quality effects can form a large portion of the overall benefits of climate policy. A global summary of previous studies found that estimates of the air quality related health co-benefit of climate policy fell in the range of \$2-196/tCO_2 (Nemet *et al* 2010). Health co-benefits can thus be on the same order of magnitude as estimates for the social cost of carbon (SCC) of \$12-123/tCO_2 in 2020 (Interagency Working Group on Social Cost of Greenhouse Gases (IWG) 2016). Recent modeling work for the US and other regions has also found that health co-benefits alone can exceed the cost of climate policy (West *et al* 2013, Thompson *et al* 2014, EPA 2015, Shindell *et al* 2016, Thompson *et al* 2016).

Renewable energy policy is a particularly popular type of climate policy in the US (Leiserowitz *et al* 2018), frequently supported for reasons additional to climate change mitigation (Rabe 2006). Renewable Portfolio Standards (RPSs) are among the most prevalent types of renewable energy policies (Carley and

Chris 2012). An RPS requires electricity suppliers to source a given percent of electricity from eligible renewable power generating technologies. Such policies exist in 29 states and the District of Columbia, and in the European Union, China, India, and elsewhere (IRENA 2015).

Previous literature on the health co-benefits of US RPSs has focused on national-level effects (Eastin 2014, Mai *et al* 2016, Wiser *et al* 2016). State-level regulatory assessments have typically focused on the economic effects of RPSs (Heeter *et al* 2014). To our knowledge, only a small number of peer-reviewed studies have estimated state-level air quality impacts (Rouhani *et al* 2016, Hannum *et al* 2017). Rouhani *et al* (2016) estimated costs and benefits of an RPS in California using a bottom-up, partial equilibrium model (representing a sub-sector of economy with a large number of discrete technologies) for the power generation mix resulting from different RPS targets. The authors estimated health benefits using marginal benefits per unit of emission abatement from Siler-Evans *et al* (2013). Hannum *et al* (2017) used a top-down, computable general equilibrium (CGE) model (providing an economy-wide perspective taking into account market distortions and income effects) to estimate RPS costs in Colorado and the reduced-form air pollution model APEEP to estimate health benefits. Evaluating RPS impacts in other areas of the US continues to be relevant, especially in the absence of federal climate policy. Local effects 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).

A challenge concerning RPS evaluation is the quantification of economic impacts. Modeling studies that estimate the impacts of RPSs have commonly employed partial equilibrium electricity system models (Mai *et al* 2016, Rouhani *et al* 2016, Wiser *et al* 2016). While electricity system models offer detailed bottom-up representation of power markets, they generally preclude considerations of the ripple effects and feedbacks that such policies can cause beyond the electricity sector. An alternative approach is the use of CGE modeling (Thompson *et al* 2014, Saari *et al* 2015, Hannum *et al* 2017). Such models represent the whole economy and capture feedbacks between producers and consumers based on the economic theory of general equilibrium formalized by Arrow and Debreu (1954). While CGE models usually represent energy sector technologies in less detail relative to bottom-up approaches, CGE models enable researchers to estimate the economy-wide costs of climate policy and assess how sector-specific policies influence emissions from unregulated sectors. The US Environmental Protection Agency (EPA) has stated that a general equilibrium approach may be preferable when a policy can be expected to impact a wide number of sectors (EPA 2014). Previous literature has argued that CGE-based methods are particularly appropriate for

analyzing climate policy (Bhattacharyya 1996, Sue Wing 2009). To our knowledge, Hannum *et al* (2017) represents the only sub-national RPS study to quantify future health co-benefits and total economic costs using a general equilibrium approach.

Decision making can also benefit from an understanding of how RPSs compare to alternative policies. Economists often recommend carbon pricing as the most cost-effective climate mitigation policy (Pigou 1932, Stern 2006, High-Level Commission on Carbon Prices *et al* 2017). Rausch and Mowers (2014) estimated that a carbon price reduces CO_2 emissions at 25% of the cost of 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 compared to other policies (Knittel and Sandler 2011, Boyce and Pastor 2013, Thompson *et al* 2014, Driscoll *et al* 2015).

Here, we assess future $PM_{2.5}$ related health co-benefits of RPSs in the 'Rust-Belt' region, comprised of Pennsylvania, Ohio, Wisconsin, Michigan, Illinois, Indiana, West Virginia, New Jersey, Maryland, and Delaware. We further estimate the total economic costs of this region's RPSs, quantified as the loss of household consumption, a common economic measure for societal policy costs (Paltsev and Carpos 2013), by using a general equilibrium approach that captures the ripple effects of RPSs beyond the electricity sector. This study also represents, to our knowledge, the first direct comparison of the health co-benefits of RPSs and carbon pricing.

2. Methods

We link a series of models to estimate how climate policy influences the economy, emissions, $PM_{2.5}$ concentrations, human health, and climate. We integrate the MIT US Regional Energy Policy (USREP) model, a CGE model for the US economy, with a reduced-form air pollution model, the Intervention Model for Air Pollution (InMAP). We use USREP to simulate the 2030 economic impacts and CO_2 effects of climate policy. We estimate resulting air pollutant emissions by scaling a base-year emissions inventory to account for changes in the economy simulated by USREP. We then use InMAP to translate emissions to pollution concentrations and premature mortalities. Finally, we estimate the economic benefits of avoided deaths and climate change mitigation, quantified using the Value of Statistical Life (VSL) and the SCC. We use these models to evaluate five scenarios designed to explore the impacts of alternative policy options.

The USREP model, which was described in detail in Rausch *et al* (2010) and Yuan *et al* (2017), contains 12 regions and aggregates economic activity into 10 economic sectors. Power generating technologies are parameterized based on cost data from the US Energy Information Administration (EIA 2017a), compiled

by Morris *et al* (2019). Electric vehicles are modeled as in Chen *et al* (2017). RPS policies are represented in the model using the approach described by Morris, Reilly, and Paltsev (2010).

Air pollutant emissions in 2030 are estimated by scaling 2014 emissions from the US National Emissions Inventory (NEI) (EPA 2017) based on region-specific changes in economic variables in the period from 2014 to 2030 estimated by USREP, following the approach of Thompson *et al* (2014). First, 2014 emissions are aggregated across pollutant species, time, and space to match the specifications of InMAP (Tessum *et al* 2019). Next, we match the EPA Source Classification Codes used to categorize individual emission sources to relevant economic variables estimated by USREP. Unlike Thompson *et al* (2014), who matched private transportation air pollutant emissions to transportation sector output estimated by USREP, we match private transportation emissions to USREP's estimate of CO_2 emissions of transportation to more accurately represent changes in this sector.
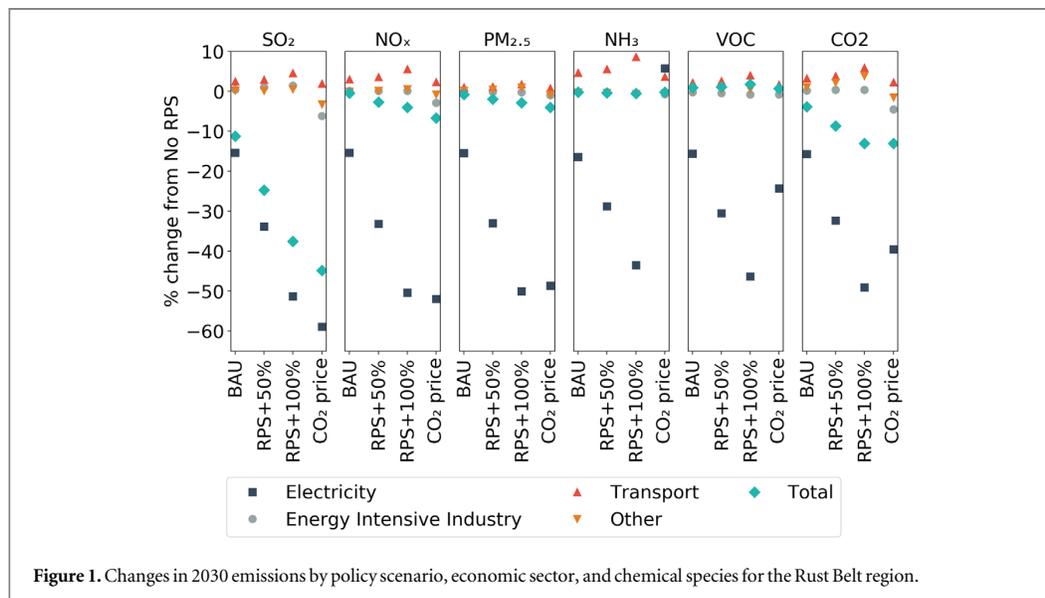
The estimated 2030 emissions are entered into InMAP to estimate 2030 concentrations of $PM_{2.5}$. InMAP simulates the formation of secondary $PM_{2.5}$ and long-range transport of pollution particles using spatially-resolved annual-average physical and chemical information derived from a state-of-the-science Chemical Transport Model (WRF-Chem). InMAP makes simplifying assumptions regarding atmospheric chemistry. For example, it contains a linear representation of the chemical transformation of emissions into secondary $PM_{2.5}$. The model was described in detail by Tessum, Hill, and Marshall (2017). InMAP is run statically at a varying spatial-resolution containing up to eight nesting levels, with the largest grid size equal to 288 km^2 and the smallest equal to 1 km^2. We use 2005 historical meteorology from Tessum, Hill, and Marshall (2015).

InMAP is also used to estimate premature mortalities. We estimate a concentration-response coefficient for the impact of $PM_{2.5}$ concentrations on early deaths by pooling coefficients estimated by Krewski (2009) and Lepeule *et al* (2012) using random effects pooling as described by EPA (2018b). To estimate premature deaths in 2030, we scale population and mortality data to 2030 using US-wide and demographic-specific population projections (US Census Bureau 2012). We further downscale the spatial resolution of InMAP results to the state level to allow the estimation of results specific to political jurisdictions. To do so, we intersect InMAP's variable-resolution grid of mortality estimates with state boundaries. Where state boundaries cross InMAP grid cells, we divide the grid among states and apportion premature mortalities in proportion to area. We treat all lives lost due to 2030 $PM_{2.5}$ concentrations as occurring in 2030. This assumption results in a small overestimate of 2030 co-benefits, as we do not discount premature mortalities occurring later than 2030. As discussed in more

detailed in the supplementary material, a discount rate of 3% and a cessation lag structure used in regulatory analyzes (EPA-SAB 2004) results in an 11% reduction in the dollar value of health co-benefits.

The economic co-benefit of avoided premature mortalities is quantified using the VSL, consistent with regulatory analyzes (EPA 2015). We use a range of VSL estimates published by the EPA, equal to $1–23 million in 2015 dollars (EPA 2014). The EPA's central estimate, equal to $8.6 million in 2015 dollars, is used for the central results of this study. We scale VSL estimates by changes in GDP from 2015 to 2030 occurring in each policy scenario, using an income elasticity of 0.4 based on the recommended central value in EPA's Benefits Mapping and Analysis Program-Community Edition model (RTI International 2015). Finally, we estimate climate change mitigation benefits using CO_2 emission changes estimated by USREP and the EPA's central SCC estimate of $56.6/$tCO_2$ in 2030 (2015 dollars) (Interagency Working Group on Social Cost of Greenhouse Gases 2016). All monetary impacts presented in this paper are expressed in 2015 dollars.

To evaluate alternative policy options, we design five policy scenarios: business-as-usual (BAU), RPS + 50%, RPS + 100%, No RPS, and CO_2 price. The BAU scenario reflects current RPS statutes. It simulates a regional RPS for the Rust Belt region, with a renewable requirement equal to the average of the renewable requirements of the existing RPSs in individual Rust Belt states (NC Clean Energy Technology Center 2018), weighted by 2016 electricity sales (EIA 2017c). We subtract any RPS requirements specific to solar or distributed generation (known as 'carve-outs') from the total renewable requirement, as these technologies are not represented in our economic model. These carve-outs represent 5% of the total weighted average renewable requirement in the Rust Belt region (N.C. Clean Energy Technology Center 2018). The estimated RPS requirement for the Rust Belt equals 6% in 2015 and 13% in 2030. Two additional scenarios (RPS + 50% and RPS + 100%) test the impacts of strengthening the region's RPSs. These scenarios reflect a gradual increase in the renewable requirement over time to reach a 2030 value that is 50% and 100% larger respectively than the 2030 requirement under BAU. Additionally, we include a counterfactual No RPS scenario. In this scenario, all RPSs in the region are assumed to be repealed as of 2015. Finally, we define a CO_2 price scenario to represent the impact of implementing a carbon price as an alternative to strengthening RPSs. The CO_2 price scenario implements a cap-and-trade system in the Rust Belt in 2020. The cap is specified to be stringent enough to achieve the same amount of cumulative CO_2 reductions as the RPS + 100% scenario. The CO_2 price scenario includes a BAU-level RPS, so that it represents the impacts of a CO_2 price in addition to existing RPS policy. For each of these five scenarios, we present our central results as well as two sensitivity cases that

Figure 1. Changes in 2030 emissions by policy scenario, economic sector, and chemical species for the Rust Belt region.

change the capital costs of wind turbines by +/− 15% (labeled high cost and low cost).

3. Results

3.1. Emissions

The three RPS scenarios reduce total SO_2 emissions in the Rust Belt by 11%–38% relative to the No RPS scenario, with emission reductions being directly proportional to RPS stringency. The impact on other pollutant species is smaller, with RPS scenarios abating total NO_x emissions by 0.4%–4.0%, primary $PM_{2.5}$ emissions by 0.8%–2.8%, NH_3 emissions by 0.2%–0.6% and VOC emissions by 1%–1.7%. As illustrated in figure 1, the majority of emission impacts occur in the electricity sector, which contributes 70%, 13%, 7%, 1%, and 0% to total emissions of SO_2, NO_X, primary $PM_{2.5}$, NH_3, and VOC respectively in the BAU scenario. These changes take place as RPS policy causes renewable generation deployment to displace coal- and gas-based generation from the power mix. The percentage of renewable generation estimated by USREP in 2030 is 6%, 13%, 20%, and 26% in the No RPS, BAU, RPS + 50% and RPS + 100% scenarios, respectively. The share of electricity produced by coal in 2030 is 33%, 29%, 23%, and 17%, respectively. This is equivalent to reductions of 46, 111, and 167 TWh in the BAU, RPS + 50% and RPS + 100% scenarios relative to No RPS. The 2030 gas share changes from 30% in the No RPS scenario to 26%, 25%, 22% (58, 78, 113 TWh) in the three RPS scenarios, respectively. The amount of energy provided by nuclear and oil, which compromise the remainder of the energy mix (respectively contributing 32% and 0.2% under BAU), is relatively unchanged across scenarios. With regard to CO_2 emissions, the three RPS scenarios reduce 2030 emissions in the Rust Belt by 50, 112, 168 Mt CO_2

compared to No RPS (equivalent to 4%, 9%, and 13% respectively).

RPSs are also estimated to lead to an emission leakage effect: a rise in transportation sector emissions that partially offsets reductions in the electricity sector. In the BAU scenario, emissions of SO_2 and NO_x in that sector rise by 3% while primary $PM_{2.5}$ emissions increase by 1% relative to No RPS. This occurs as higher electricity prices caused by RPS policies incentivize households to increase usage of internal combustion engine vehicles relative to electric vehicles. In the BAU scenario, the share of vehicle miles traveled by electric vehicles in 2030 is 4% (compared with 9% in the No RPS), while total vehicle miles traveled are virtually the same. This difference is driven by a 3% increase in the 2030 price of electricity faced by consumers in the Rust Belt under BAU relative to No RPS. This strong response in vehicle miles traveled to power price changes occurs because electric vehicles happen to be on the cusp of being competitive against internal combustion engine vehicles in our scenario. As a result, small changes in costs have a relatively large effect on the uptake of electric vehicles. Thus, the magnitude of this result is not generalizable outside of our scenarios.

The CO_2 price scenario, by design, achieves the same emission reductions as the RPS + 100% scenario. The reductions required to be achieved by the modeled regional cap-and-trade system are 118 Mt. The CO_2 price generated by the model to achieve these reductions is relatively modest at \$4/tCO_2 in 2030. This scenario exerts qualitatively different effects on the economy. In the electricity sector, the CO_2 price increases the marginal cost of CO_2 emitting technologies based on their CO_2 emission intensity, bolstering the competitiveness of gas relative to coal, thus leading to fuel switching. This scenario results in a 2030 coal
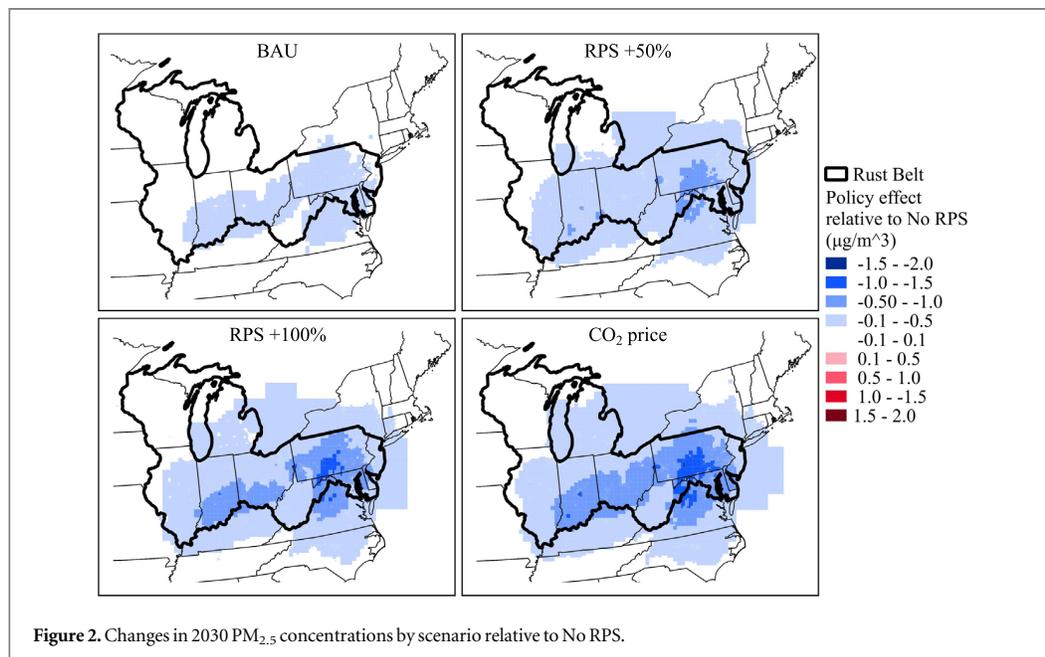
Figure 2. Changes in 2030 PM$_{2.5}$ concentrations by scenario relative to No RPS.

share of 8%, and an increased gas share of 46%. The renewable share remains unchanged from the BAU scenario because the CO_2 price achieves the required CO_2 reduction through cheaper abatement options, with coal-to-gas switching playing a predominant role. As a result of the lower amount of coal generation, carbon pricing reduces electricity sector emissions of SO_2 and NO_x to a greater degree than the comparable RPS + 100% scenario (figure 1). However, the greater use of gas under carbon pricing results in higher emissions of PM$_{2.5}$, NH_3, and VOCs in the electricity sector compared to RPS + 100%. The CO_2 price scenario lowers emissions in other sectors due to its economy-wide scope. For example, it lowers coal consumption in energy intensive industry. It also partially offsets the increase in transportation sector emissions caused by the BAU RPS.

3.2. PM$_{2.5}$ concentrations and mortalities

The effect of our policy scenarios on PM$_{2.5}$ concentrations relative to No RPS mostly occur in the Rust Belt region (figure 2). The relative reductions are largest in Maryland, Delaware, Pennsylvania, Indiana, Ohio, and West Virginia. In the BAU scenario, average population-weighted concentration changes in these states range from $-0.14\ \mu\mathrm{g\,m}^{-3}$ (-1.5%) in Maryland to $-0.10\ \mu\mathrm{g\,m}^{-3}$ (-2.4%) in West Virginia.
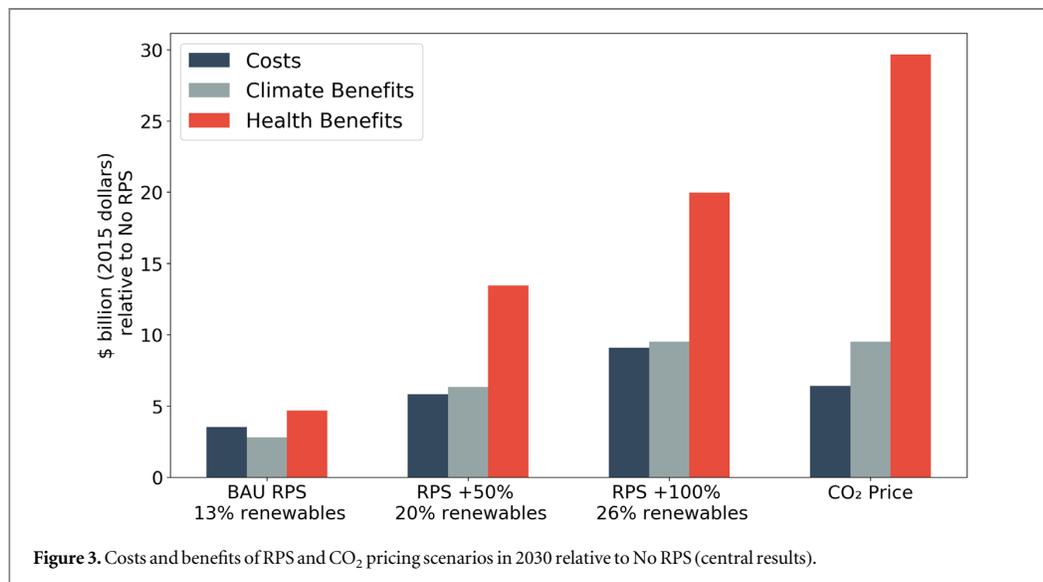
Concentrations of PM$_{2.5}$ are even lower under the more stringent climate policies. We observe the largest reductions in the CO_2 price scenario. Maryland experiences the greatest decrease in population-weighted concentrations of $0.76\ \mu\mathrm{g\,m}^{-3}$ (-8.2%) relative to No RPS. The smallest reduction occurs in Wisconsin and equals $0.06\ \mu\mathrm{g\,m}^{-3}$ (-0.9%). Concentrations also decline in downwind states such

Virginia (up to $-0.5\ \mu\mathrm{g\,m}^{-3}$, or -7.1%), followed by New York (up to $-0.2\ \mu\mathrm{g\,m}^{-3}$, -1.9%). The location of air quality improvements partially reflects the distribution of coal plants along the Ohio river. These improvements in air quality are estimated to result in 467, 1350, 1999, and 3006 avoided annual premature mortalities in the Rust Belt in the three RPS scenarios and the CO_2 price scenario relative to No RPS (equivalent to 0.9%, 2.5%, 3.7%, and 5.5% reductions in mortalities respectively).

3.3. Costs and benefits

The health co-benefits of existing RPSs in the Rust Belt exceed both the total policy costs and estimated climate benefits according to our central results (figure 3). The combined uncertainty in the concentration-response coefficient and the VSL leads to a large range of health co-benefit values spanning three orders of magnitude (table 1). Uncertainty in the concentration-response coefficient is based on the coefficient's 95% confidence interval. VSL uncertainty accounts for all values published in EPA (2014). The VSL uncertainty is responsible for more than half of the combined uncertainty reported in table 1 (see supplementary document, available online at stacks.iop.org/ERL/14/085012/mmedia).

The health co-benefits of the BAU, RPS + 50%, and RPS + 100% scenarios correspond to co-benefits of $94, $120, $119 per ton of CO_2 reduced respectively. These estimates are equivalent to health co-benefits of 8¢, 12¢, and 13¢ per kWh of new renewable generation. In comparison, the economic costs of the three RPS scenarios correspond to 6¢, 5¢, and 6¢ per kWh respectively. In percentage terms, the economic costs represent a decrease in macroeconomic

Figure 3. Costs and benefits of RPS and CO_2 pricing scenarios in 2030 relative to No RPS (central results).

Table 1. Costs and benefits in 2030 by policy scenario (billion 2015 dollars). Climate benefit uncertainty includes uncertainty in the discount rate and marginal damages of climate change. The air quality uncertainty includes the 95% confidence interval for the concentration-response coefficient and the full range of values for the Value of Statistical Life reported in EPA (2014).

Policy scenarios	Climate benefits	Health co-benefits	Costs
BAU	$2.8 ($0.9–8.6)	$4.7 ($0.1–23.7)	$3.5
RPS + 50%	$6.4 ($2.0–19.3)	$13.5 ($0.3–68.3)	$5.8
RPS + 100%	$9.5 ($3.0–29.0)	$20.0 ($0.4–101.4)	$9.1
CO_2 price	$9.5 ($3.0–29.0)	$29.7 ($0.7–151.0)	$6.4
BAU (low cost)	$2.9 ($0.9–8.7)	$6.0 ($0.1–30.5)	$3.4
RPS + 50% (low cost)	$6.0 ($1.9–18.4)	$13.4 ($0.3–68.1)	$5.2
RPS + 100% (low cost)	$8.9 ($2.9–27.1)	$18.7 ($0.4–95.1)	$7.7
CO_2 price (low cost)	$8.9 ($2.9–27.1)	$29.3 ($0.6–149.2)	$5.9
BAU (high cost)	$2.9 ($0.9–8.8)	$4.9 ($0.1–24.8)	$5.2
RPS + 50% (high cost)	$6.6 ($2.1–20.2)	$14.3 ($0.3–72.4)	$8.0
RPS + 100% (high cost)	$9.9 ($3.2–30.2)	$21.0 ($0.5–106.6)	$11.9
CO_2 price (high cost)	$9.9 ($3.2–30.2)	$32.4 ($0.7–165.3)	$8.1

consumption of 0.1%, 0.1%, and 0.2% in the three RPS scenarios relative to No RPS.

Monetized benefits of CO_2 reductions (referred to here as 'climate benefits') are also comparable to policy costs and may substantially exceed them depending on the assumed SCC (table 1). We quantify the uncertainty in climate benefits using the four alternative SCC assumptions provided by IWG (2016). The high end of the uncertainty range reflects the 95th percentile of the SCC probability distribution, recommended by the IWG as a way to represent the marginal impact of low-probability, high-impact damages caused by climate change. The low end represents the use of a 5% discount rate (relative to the 3% rate used for the central SCC value).

Carbon pricing results in greater health co-benefits than the comparable RPS + 100% scenario. Since the CO_2 price scenario includes a BAU-level RPS, we estimate the co-benefit of carbon pricing based on the additional health benefits relative to the BAU, resulting in an estimated health co-benefit of $211/tCO_2

(the equivalent estimate for the RPS + 100% scenario equals $129/tCO_2$). The health co-benefit of the CO_2 price is higher partially due to its stronger effect on coal-fired generation. It is also due to the increase in transportation sector emissions occurring under RPSs, which offsets their overall health co-benefits. In addition, carbon pricing results in lower cost by incentivizing the least-cost CO_2 abatement options. Relative to the BAU, the additional costs of the RPS + 100% scenario are twice as large as the costs of carbon pricing.

We test the impact of the emission leakage in the transportation sector under RPSs by recalculating health co-benefits assuming private transportation emissions remain the same as in the No RPS scenario, thus eliminating the effect of RPSs on private transportation emissions. Under this experiment, health co-benefits in the Rust Belt were 35%–79% higher depending on the RPS scenario (the BAU scenario exhibited the largest increase). This emission leakage effect is sensitive to the extent to which RPSs increase electricity prices, which is

the underlying cause behind the changes in emissions from transportation as discussed previously. Electricity system modeling by Mai *et al* (2016) estimates that existing RPSs lead to smaller changes in 2030 power prices between +1% and −0.4% depending on region and underlying assumptions.

4. Discussion and conclusions

Health co-benefits may alone justify the implementation of RPSs or carbon pricing as our central estimates show. This result is consistent with previous literature, which found that the health co-benefits of climate policy (including RPSs and other instruments) tends to exceed policy costs (West *et al* 2013, EPA 2015, Mai *et al* 2016, Shindell *et al* 2016, Thompson *et al* 2014, 2016, Wiser *et al* 2016). Our estimated health co-benefits of 8¢/kWh are greater than the national average of 1.2–4.2¢/kWh estimated by Mai *et al* (2016), consistent with the greater share of coal generation in the Rust Belt region (EIA 2017b).

We further estimate that increasing the renewable requirement of existing RPSs in the Rust Belt region would increase net societal benefits. As RPS stringency is raised, health co-benefits increase more than costs. The marginal health co-benefits (the incremental co-benefit incurred from the No RPS to the BAU scenario, and so on) are larger than the marginal costs across all RPS scenarios tested.

Our results also demonstrate that there can be meaningful differences between the health co-benefits of alternative climate policies. We find that, to 2030, carbon pricing is more efficient (greater net benefits) relative to an RPS than suggested by cost-per-ton-reduced comparisons that do not consider health co-benefits (e.g. Rausch and Mowers (2014)). Regardless of efficiency, however, RPS policies have been more politically popular, leading to their more frequent implementation (Rabe 2018). Additionally, while carbon pricing results in higher health co-benefits in 2030, the relative merits of different climate policies would differ in an assessment that includes the full environmental externalities of natural gas extraction (EPA 2016), the Social Cost of Methane (Marten *et al* 2012) or the implications that increasing natural gas consumption may have for long-term policy targets aiming to achieve deep reductions in CO_2 emissions (Erickson *et al* 2015). Our paper has also not addressed non-air quality related hazards associated with renewable technologies (e.g. Moura Carneiro, Barbosa Rocha, and Costa Rocha 2013).

Several limitations of this work are worth noting. First, we do not attempt to causally attribute the estimated benefits to RPS policies as we do not capture other renewable energy policies that may induce deployment. Instead, the results of this study are indicative of the effects of renewable technology deployment consistent with the requirements of modeled RPS scenarios.

Second, the use of general equilibrium modeling introduces the disadvantage of representing the electricity sector in a top-down fashion, thus omitting details including intra-day power dispatch based on operational limits such as power plant ramping flexibility. We thus do not explicitly represent certain challenges of renewable integration such as the occurrence of negative spot electricity prices. Renewable integration challenges are expected to be less severe at the modest penetration levels modeled in this paper with the highest modeled renewable share at 26%. Recent work has demonstrated the possibility of leveraging the advantages of both general equilibrium and electricity sector modeling through hybrid approaches that iteratively combine both types of models (Rausch and Mowers 2014, Tapia-Ahumada *et al* 2015). Third, our scenarios do not model air pollution policy in the US such as the emission trading systems for SO_2 and NO_x emissions under the Cross-State Air Pollution Rule (CSAPR). This may cause our results to overestimate the effects of climate policies on air pollution if reductions in air pollutant emissions from one source cause the transfer of emission permits, allowing another source to increase emissions, offsetting the original reductions (Groosman *et al* 2011). This effect is likely to be limited, however, as emission sources already have access to a surplus number of permits under CSAPR, particularly for SO_2 (EPA 2018a).

An important area for future work will be to quantify the uncertainty in health effects associated with the choice of air pollution model. While the health co-benefit results presented here compare closely to estimates derived from chemical transport models (Thompson *et al* 2014, 2016), we do not quantify uncertainty related to model choice. Subsequent research could apply state-of-the art chemical transport models alongside the type of reduced-form model used in this work to a variety of relevant policies to help understand which atmospheric modeling methodologies are best suited to which types of policy evaluations.

Acknowledgments

Data availability statement

The data that support the findings of this study are available from the corresponding author upon reasonable request. The data are not publicly available for legal and/or ethical reasons.

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