ENVIRONMENTAL AND ECONOMIC RESEARCH AND DEVELOPMENT PROGRAM

Quantifying the Air Quality Co-benefits of Lower-Carbon Electricity Production

Final Report
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List of Acronyms

2002-NEI	Model scenario reflecting year 2002 emissions, based on NEI
2008-BC	Model scenario reflecting year 2008 emissions, based on NEI + MyPower
2024-BAU	Model scenario reflecting year 2024 emissions, business as usual
2024-OFF	Model scenario reflecting year 2024 emissions with carbon offsets
2024-RPS	Model scenario reflecting year 2024 emissions with RPS
AQS	Air Quality System (EPA ground based measurements)
ASO4	Sulfate aerosol, a constituent of $PM_{2.5}$ (CMAQ variable name)
CAIR	Clean Air Interstate Rule
CAM	Clean Air Markets
CMAQ	EPA Community Multiscale Air Quality Model
CO_2	Carbon Dioxide
CONUS	Continental United States
EGU	Electricity Generating Unit
EPA	United States Environmental Protection Agency
ERL	*Environmental Research Letters* (journal)
GL	Great Lakes
GWh	gigawatt hour
HNO_3	Nitric acid
IMPROVE	Interagency Monitoring of Protected Visual Environments (source of ground based measurements)
IOP	Institute of Physics (publisher of *ERL*)
kWh	kilowatt hour
LADCO	Lake Michigan Air Directors Consortium
MDA8	Maximum Daily 8-hour ozone mixing ratio
MFB	Mean Fractional Bias
MFE	Mean Fractional Error
MMBtu	one million British thermal units
NAAQS	National Ambient Air Quality Standards
NARR	North American Regional Reanalysis
NEEDS	National Electric Energy Data System
NEI	National Emissions Inventory (from EPA)
NO_2	Nitrogen Dioxide
NO_x	Nitrogen Oxides ($NO + NO_2$)
NO_3^-	Nitrate aerosol, a constituent of $PM_{2.5}$
$PM_{2.5}$	Particulate Matter less than 2.5 micro-meters in diameter
O_3	Ozone
OH	Hydroxyl radical
PI	Principal Investigator
RPS	Renewable Portfolio Standard
SIP	State Implementation Plan
SMOKE	Sparse Matrix Operating Kernel Emissions Model
SO_2	Sulfur Dioxide
SO_4^{2-}	Sulfate aerosol, a constituent of $PM_{2.5}$
STN	Speciation Trends Network (source of ground-based measurement)

WDNR	Wisconsin Department of Natural Resources
WRF	Weather Research and Forecasting Model
WRF-ARW	WRF-Advanced Research WRF (Advanced Research version)

Executive Summary

With Focus on Energy support, we have quantified the air quality co-benefits from energy strategies identified by the 2007 Wisconsin Governor's Global Warming Task Force[1]. Many of the policy scenarios examined by the Task Force for their carbon mitigation value would simultaneously reduce health-relevant air pollution. By taking emissions contributing to ozone and particulate matter into account, the full costs and benefits of energy policies may be more accurately quantified. In 2007, then Wisconsin governor Jim Doyle formed a Task Force on Global Warming to recommend strategies for reducing the state's anthropogenic greenhouse gas emissions. The Task Force recommended a range of policies including an increased Renewable Portfolio Standard (RPS), energy efficiency measures, and the purchase of carbon offset credits. If all of the Task Force carbon reduction policies are implemented in-state (no carbon offsets), the state's current RPS of 10% by 2015 would increase to 24% by 2024, and demand reduction policies would decrease overall electricity generation by an estimated 18% compared to business-as-usual. The demand reduction measures considered include energy efficiency programs, new residential and commercial building codes, a state appliance efficiency standard, and a residential rental lighting standard. We quantify the emissions and air quality impacts of these proposed demand reduction and RPS policies, and compare results with current (2002 and 2008) and alternative future (2024) scenarios.

To characterize the air quality response to emissions changes, we use the state-of-the-art atmospheric chemistry model from the United States Environmental Protection Agency (EPA). This model, the Community Multiscale Air Quality (CMAQ) model, was used to evaluate how proposed energy policies can aid Wisconsin counties in achieving compliance with the National Ambient Air Quality Standards (NAAQS). To date, only a handful of studies to date have estimated future plant-by-plant emission changes and resulting air quality impacts. Our work here is the first to simulate plant-by-plant electricity generation under future policy and technology changes, develop associated emission scenarios, and quantify air quality impacts.

We find that the impact of air emissions from electricity generation depends critically on the spatial distribution of power plants and electricity dispatch decisions. These emissions interact with weather patterns and land cover, so any evaluation of the air quality impacts of electricity policies must account for the spatially heterogeneous changes in associated emissions.

To quantify the air quality impacts of the estimated emission reductions, we calculate the difference in ambient concentrations of sulfate, nitrate, and ozone between current and future scenarios. Sulfate, a component of fine particulate matter ($PM_{2.5}$) exhibits the largest monthly mean response to emission reductions during the July 2003 study period. Emissions reductions from the 2008 to the 2024 (including RPS and demand reduction) show moderate monthly mean sulfate decreases (0.05-0.2 $\mu g/m^3$, 5-9%) throughout much of the state and extending over Lake Michigan into western Michigan. The largest calculated sulfate decreases (0.2-0.3 $\mu g/m^3$, 9-15%) occur in southeastern and south-central Wisconsin, consistent with spatial patterns in emission reductions. Most relevant to air quality management is the impact of proposed emission reductions on peak events. At Milwaukee, the largest urban area in Wisconsin, modeled sulfate

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[1] http://dnr.wi.gov/air/aq/global/climatechange/GTFGW.html

decreases are largest when sulfate concentrations are between 2 and 4 μg/m^3 and total $PM_{2.5}$ concentrations exceed 15 μg/m^3 (the annual $PM_{2.5}$ NAAQS limit) but not 35 μg/m^3 (the daily $PM_{2.5}$ NAAQS). This limited example suggests that proposed changes would impact what we call here "secondary episodes" (days with concentrations above the annual NAAQS limit, but not above the daily limit). Overall, we find that sulfate PM on "moderate" air pollution days can decrease up to 15% with proposed energy policies.

Using this methodology for Wisconsin, power sector carbon reduction policies are found to significantly reduce statewide emissions of NO_x and SO_2 compared to the business-as-usual scenario, with smaller reductions for policies utilizing external carbon offsets. The most significant findings from this work deal with the strong potential to reduce summertime sulfate, contributing to high-PM episodes, with increased efficiency and renewable energy generation. Although the single-month study here is appropriate as proof-of-concept, our results point to issues for further analysis, all of which are currently under investigation. We find an important impact on peak air pollution events, relevant for regulatory compliance. However, a single month is not adequate to quantify peak events, since there may only be a couple events in a single month at each location. Thus, we are extending this analysis to cover both winter and summer, with three simulated months in each season. Also, our model evaluation showed errors in CMAQ simulation of summertime nitrate, which may be corrected with a newer version of CMAQ (v. 4.7, versus v. 4.6 presented here).

Beyond our focus on Wisconsin, our team has explored more generally the challenges and opportunities for integrated decision-making on energy systems to achieve both climate and air quality goals. We have shown that opportunities exist for synergistic emission control policies and "win-win" energy policy solutions through an analysis published in the high-ranking journal *Environmental Research Letters (ERL)*. The published manuscript is attached to this report as Appendix A, and has been cited over 30 times (according to Google Scholar, June 2012).

In a review of previous studies we found a range of estimates for the air quality "co-benefits" of climate change mitigation of \$2-196/tCO2 with a mean of \$49/tCO2, and the highest co-benefits found in developing countries. These values are of a similar order of magnitude to abatement cost estimates, but they are only rarely included in integrated assessments of climate policy. If these air quality benefits were taken into account, it would be expected to strongly affect climate policy assessment. Climate policy design, cost, and incentives for international cooperation, among other aspects, would be expected to change. Because policy debates are framed in terms of cost minimization, policy makers are unlikely to value air quality co-benefits unless they can be compared on an equivalent basis with the benefits of avoided climatic damages. While air quality co-benefits have been prominently portrayed as a hedge against uncertainty in the benefits of climate change abatement, this assessment finds that full inclusion of co-benefits depends on—rather than substitutes for—better valuation of climate damages.

In summary, we have conducted a detailed scientific assessment of air quality co-benefits from climate policies for Wisconsin, and we have conducted a detailed policy assessment of air quality co-benefits for broader climate decision-making. Both point to the significance of considering multi-pollutant impacts of energy strategies, rather than considering carbon reductions alone.

Study Motivation

In the past decade, many U.S. states have developed electricity generation and consumption policies aimed at reducing carbon dioxide (CO_2) emissions. Currently, 29 states have non-voluntary renewable portfolio standards (RPS) for electricity, and 20 states have energy efficiency resource standards [1]. As states move away from carbon-intensive generation, emissions of health-damaging pollutants such as nitrogen oxides (NO_x) and sulfur dioxide (SO_2) are typically reduced, resulting in improvements to local and regional air quality [2, 3]. Consequently, the U.S. EPA has recently released guidance on how to include carbon reduction policies into air quality improvement plans (State Implementation Plans, or SIPs) [4]. In addition, air quality improvements from lower carbon generation would eliminate some of the need for expensive new pollution control equipment, offsetting a portion of the cost of carbon reduction policies [5-7]. Thus, taking air quality into account is critical to achieving the broader environmental goals of lower carbon electricity policies in the most cost-effective manner.

The impact of air emissions from electricity generation depends fundamentally on the spatial distribution of power plants and electricity dispatch decisions. These emissions interact with weather patterns and land cover, so any evaluation of the air quality impacts of electricity policies must account for the spatially heterogeneous changes in associated emissions. Here, we present an analysis of the changes in fine particulate matter ($PM_{2.5}$) and ozone (O_3) associated with proposed energy efficiency and renewable energy measures in Wisconsin.

We simulate the state's electricity system and its potential response to policies using the MyPower electricity model, which calculates plant-by-plant reductions in NO_x and SO_2 emissions. Then, we use CMAQ, a leading mathematical model of atmospheric chemistry and meteorology to calculate how these emission changes would impact air quality in Wisconsin and neighboring states. Would reducing air pollution from power plants yield day-in, day-out improvements? Might it actually reduce the number of "bad air" days in Wisconsin counties, possibly bringing new areas into attainment with federal regulations? The CMAQ air quality model developed and used by the U.S. EPA for regulatory analysis and policy design, and it is well suited to our study goals.

Beyond addressing questions relevant to energy and environmental decision-making in Wisconsin, we have been active in communicating our results, publishing work in high-rated academic journals, and connecting Focus on Energy funding with student training and career development.

Why Quantify Air Quality Co-Benefits?

The issues motivating our study connect public policy, electricity generation, and air quality. As such, our study team includes an expert in climate and energy policy, Dr. Greg Nemet, a leading electricity systems modeler, Dr. Paul Meier, and an expert in atmospheric chemistry and air quality modeling, Dr. Tracey Holloway (project P.I.).

While we focus our scientific analysis on Wisconsin, the study context extends well beyond our home state. In fact, worldwide countries, states, and cities are wrestling with the challenge of reducing carbon emissions. At stake is cost: how much will it cost to reduce carbon, and what are the economic benefits of such policies? Part of the benefit, of course, is avoided climate change, including slowing sea level rise and reducing the risk of extreme weather events. Benefits beyond climate, including air quality, are often referred to as "co-benefits." The value of these co-benefits is often significant, and can inform the design of no-regrets policies that meet multiple social, environmental, and economic goals.

It is well known that many strategies for reducing greenhouse gas emissions also decrease emissions of health-damaging air pollutants and precursor species, including particulate matter, NO_x and SO_2. For example, natural gas emits carbon per energy-unit produced than coal, and it also emits less nitrogen and almost no sulfur. Solar and wind energy emit essentially zero. While power plants are spending millions to reduce NO_x and SO_2 under current air quality rules, these same goals could be met with energy conservation, renewable energy investment, or switching to lower-carbon fossil fuels – potential win-win opportunities.

As a first phase of our analysis, Dr. Nemet led an analysis into the way these air quality co-benefits have been treated in past assessments of carbon reduction costs and benefits. Based on this study, our team identified barriers and opportunities for incorporating air quality co-benefits into future climate policy assessments. In a survey of previous studies we found a range of estimates for the air quality co-benefits of climate change mitigation of \$2-196/tCO_2 with a mean of \$49/tCO_2. These values are similar to carbon abatement cost estimates, but these co-benefits are only rarely included in integrated assessments of climate policy.

Because policy debates are framed in terms of cost minimization, policy makers are unlikely to fully value air quality co-benefits unless they can be compared on an equivalent basis with the benefits of avoided climatic damages. Although an evaluation of climate damages for Wisconsin was beyond the scope of our scientific study, the importance of air quality assessment is clear.

We published this work in open-access journal *Environmental Research Letters* in 2010, and it is included here as an appendix, with the publishers permission[2]. In the two years since its publication, the paper has been cited an impressive 29 times (according to Google Scholar, March 2012).

A Multi-Pollutant Approach to Quantifying Emissions Reductions

Having shown the importance of air quality co-benefits for meaningful evaluation of energy policies in the Nemet et al. 2010 *ERL* paper, we launched the second phase of our study: quantifying how proposed energy policies in Wisconsin would impact health-damaging pollutants across the Upper Midwest.

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[2] Personal correspondence with Dr. Jill Membrey, IOP Publishing, March 22 2012 *"As one of the co-authors of the article, Professor Holloway is able to ... us[e] all or part of the article in compilations or other scholarly publications of the authors' own work. These rights cover non-commercial purposes, and the conditions of use require the display of citation information and the IOP's copyright notice, along with a link back to the on-line abstract in the journal on IOPscience."*

We focused on power sector emissions of NO_x and SO_2, which impose direct human health impacts when inhaled. Both are subject to concentration standards through the U.S. EPA criteria pollutant standards (NAAQS). Beyond direct standards, these pollutants are also regulated in large part as precursors to $PM_{2.5}$ and ground-level ozone. A significant portion of NO_x and SO_2 emissions in the U.S., over 30% and 80% respectively, originate from the electricity sector [8], and reducing these emissions has been the focus of $PM_{2.5}$ and ozone improvement efforts in the Eastern U.S. [5, 9-11]. Emissions of NO_x and SO_2 contribute to sulfate (SO_4^{2-}) and nitrate (NO_3^-), two important $PM_{2.5}$ constituents, as well as ozone. Sulfate is primarily formed through the atmospheric oxidation of SO_2, which occurs in both gaseous (via the hydroxyl radical, OH) and aqueous (dominated by hydrogen peroxide, H_2O_2, with some oxidation via ozone) phases [12]. Atmospheric NO_x concentrations contribute both to the formation of nitrate, which is commonly formed from nitric acid (HNO_3, the principal sink of NO_x) [13], and tropospheric ozone, which is formed through a complex chain of reactions in the presence of sunlight [14]. Sulfate and nitrate are major components of $PM_{2.5}$ in the Midwest: sulfate exhibits average contributions to $PM_{2.5}$ of about 16% in the winter (Dec.-Feb.) and 29% in the summer (Jun.-Aug.), whereas nitrate exhibits the opposite seasonal cycle with average contributions of about 27% in the winter and only 7% in the summer [8, 15].

Quantifying the air quality co-benefits from lower-carbon electricity generation requires accounting for spatial heterogeneity in emission changes under technology and policy scenarios. Commonly, air quality modeling studies estimate changes in anthropogenic emissions with uniform emission growth/reduction factors applied to the entire power sector [16-19]. However, because the capacity, technology, and operating costs of individual power plants vary greatly, any policy affecting the electricity sector will inevitably be implemented with a high degree of spatial variability. Several studies have considered changes in electricity emissions on a plant-by-plant basis, highlighting how air quality impacts depend on the spatial distribution of emissions changes or uncertainty [9, 10, 20, 21]. Only four studies to date have estimated future plant-by-plant emission changes and the resulting air quality impacts [2, 22-24]. However, none of these studies simulate plant-by-plant electricity generation under future policy or technology changes as a means of developing future emission scenarios. We are the first to evaluate plant-by-plant emission changes as a response to policies and technology, and quantify associated air quality impacts.

Evaluation of climate policies typically requires the use of an electricity-sector model to estimate the costs and feasibility of increased renewable energy at the utility, state, or national scale [25-28]. To date, however, these electricity models have not been used in a research capacity to generate spatially explicit future emission scenarios for air quality evaluation with chemical transport models.

Led by co-investigator Dr. Paul Meier, we employ the MyPower electricity model [29] to simulate the spatially heterogeneous electricity system of Wisconsin and its potential response to CO_2-reduction policies. Dr. Meier is the developer of MyPower, which uses publicly available data on power plant characteristics and operating cost to calculate how specific policies would impact individual power plants and their air emissions.

Total annual electricity production is estimated for each electricity-generating unit (EGU) using a least-cost dispatch routine that satisfies forecasted electricity demand represented with seasonal load duration curves. Dispatch ranking is ordered by increasing marginal cost as determined by each plant's thermal efficiency, fuel price, variable operation and maintenance cost, production credits, and operational constraints. All EGUs in Wisconsin are included as reported in the U.S. EPA National Electric Energy Data System (NEEDS) version 3.02 [31], as well as related information from the EPA's implementation of the Integrated Planning Model [32] and historical generation from U.S. Energy Information Administration historical generation data [33]. Modeling of future power generation is based on scenarios developed as part of the Task Force modeling [34] and includes changes in fuel prices, planned additions of new power plants, retirement of existing power plants, and other parameters.

Unit-specific emissions for emitting EGUs are calculated as a function of electricity generation (kWh), heat rate (MMBtu/kWh), and emission factor (tons/MMBtu), thereby generating a bottom-up emissions inventory for Wisconsin EGUs that is unique to this study. Here, we consider emissions of CO_2, NO_x, and SO_2. While the power sector is also an important source of primary $PM_{2.5}$ emissions, plant-specific emission factors are not readily available or estimated, and therefore primary $PM_{2.5}$ emissions are not included in our analysis. EGU heat rates and emission factors were derived from NEEDS and the U.S. EPA 2009 Clean Air Markets (CAM) database [35]. Post-combustion pollution control technologies, such as selective catalytic reduction for NO_x and wet/dry scrubbing for SO_2, are also considered in emissions calculations. All pollution controls currently installed (as of 2009) are incorporated in our calculations because emission factors are based on 2009 historical emission rates. Additional pollution controls are simulated in a "business as usual" scenario as needed to meet the state-specific emissions reduction targets set by the Clean Air Interstate Rule (CAIR, more recently named the Transport Rule): new NO_x controls are assumed to be added to 8 of the highest-emitting plants (17 total EGUs), and new SO_2 controls are assumed to be added to 4 plants (6 total EGUs), with unit selection based on personal consultation with the Lake Michigan Air Directors Consortium (LADCO) and the Wisconsin Department of Natural Resources (WDNR). These same pollution controls are consistent among the three 2024 scenarios described below.

Wisconsin is selected as our study region, and all emission reduction policies are applied only to power plants in the state. In 2007, then Wisconsin governor Jim Doyle formed a Task Force on Global Warming to recommend strategies for reducing the state's anthropogenic greenhouse gas emissions. The Task Force recommended a range of policies including an increased Renewable Portfolio Standard (RPS), energy efficiency measures, and the purchase of carbon offset credits [30]. If all of the Task Force carbon reduction policies are implemented in-state (no carbon offsets), the state's current RPS of 10% by 2015 [1] would increase to 24% by 2024, and demand reduction policies would decrease overall electricity generation by an estimated 18% compared to business-as-usual. The demand reduction measures considered include energy efficiency programs, new residential and commercial building codes, a state appliance efficiency standard, and a residential rental lighting standard. This Wisconsin-focused policy portfolio, which we refer to as "2024-RPS," receives the most detail in our analysis of emissions and air quality impacts. For comparison, we also consider a carbon-offset scenario that achieves the same CO_2 emission reductions (with the same demand reduction programs) as the 2024-RPS scenario, but allows 35% of carbon reductions beyond demand-driven changes to be met by purchasing carbon offsets from an external market. This scenario is referred to as "2024-OFF" (Carbon Offsets).

The two future policy scenarios, 2024-RPS and 2024-OFF, are compared with three additional cases:

1. A 2002 scenario using National Emissions Inventory (NEI) emissions for Wisconsin EGUs (2002-NEI), used for CMAQ model evaluation and quality control. The NEI is the most widely used source of emissions data for the U.S., and these emissions (2002) are from the same time period as modeled meteorology (2003) to support comparison of modeled air quality with ground-based observations.

2. A 2008 Base Case scenario with MyPower-simulated Wisconsin EGU emissions (2008-BC), used to evaluate the MyPower against 2008 reported emissions and to compare with the future scenarios as a consistent baseline

3. A 2024 Business As Usual scenario (2024-BAU). This scenario is calculated by MyPower based on a 1.9% annual electricity demand increase from 2008 to 2024, trends in fuel prices, planned addition of new generation capacity, planned retirement of old capacity, and changes in pollution control efficiency. The 2024-BAU trends and changes are consistently incorporated into the 2024-RPS and 2024-OFF.

Because the power system is interconnected across the Eastern U.S. and Canada, policy modeling encounters an unavoidable "leakage" issue: policies in Wisconsin will exert some influence on out-of-state EGUs, and out-of-state policies will similarly exert some influence on Wisconsin EGUs. While this single-state study only simulates Wisconsin policies and omits out-of-state production, this same methodology could be extended to a full interconnect region in the future to capture state-to-state electricity transfer. To evaluate MyPower for this single-state study, 2008-BC electricity generation and emissions are compared with 2008 reported values from Clean Air Markets (CAM) database [46]. Table 1 and Figure 1 compare MyPower generation and pollutant emissions for the 37 Wisconsin EGUs in the CAM database (representing about 70% of the state's electricity generation).

MyPower-estimated total electricity generation from Wisconsin EGUs was about 17% higher than 2008 CAM reported generation, primarily a result of constraining the model from importing power from out-of-state EGUs. Nearly all (93%) of this overestimation is attributable to simulation of the 24 non-coal EGUs (i.e., peaking plants), consistent with errors expected from the "leakage" issues noted above. Although non-coal generation (especially natural gas) is overestimated by a factor of 2.4, overestimation in non-coal emissions is much smaller: 17% for NO_x and 5% for SO_2. This net difference in emissions results from the overestimation of non-coal generation, partially compensated with an underestimation in generation and emissions at the highest-emitting non-coal plant (Bay Front).

For the 13 coal-fired power plants in the state, MyPower compares well to CAM values. We find excellent agreement for generation and emissions on an aggregate level, with total differences around 1%, as well as a plant-by-plant basis, with mean fractional biases (MFB) of less than ±1%[3]. Because the coal plants are responsible for 95% and 99% of MyPower NO_x and SO_2 emissions, respectively, the overestimation in non-coal generation and emissions has little impact on the overall good agreement between estimated and historic total emissions, within 1% for both NO_x and SO_2.

Table 1 – MyPower Model Validation for the 2008-BC Scenario

		Generation (GWh)	**NO_x Emissions (tons)**	**SO_2 Emissions (tons)**
All Plants (37)	MyPower	53,597	48,542	134,388
	CAM	45,804	48,661	133,158
	% Difference	17.0%	-0.24%	0.92%
	Mean Bias	222.7	-3.2	33.2
	MFB (%)	-1.54%	-6.77%	-44.8%
All Coal Plants (13)	MyPower	41,175	46,048	132,996
	CAM	40,642	46,534	131,835
	% Difference	1.31%	-1.04%	0.88%
	Mean Bias	48.5	-37.3	89.2
	MFB (%)	0.17%	-0.76%	0.31%
All Non-Coal Plants (24)	MyPower	12,422	2,494	1,393
	CAM	5,162	2,128	1,323
	% Difference	140.6%	17.2%	5.27%
	Mean Bias	302.5	15.3	2.9
	MFB (%)	-2.33%	-10.0%	-75.7%

Comparison of annual MyPower electricity generation, NO_x emissions, and SO_2 emissions to CAM reported values for 37 Wisconsin power plants. Mean bias has the same units as the metric assessed (GWh or tons). Mean fractional bias (MFB) is a percentage. Non-coal plants include natural gas, distillate fuel oil, and biomass combustion plants. Two small steam turbine plants that are fired with coal in 2008 but converted to biomass in the 2024 scenarios are grouped with the non-coal plants.

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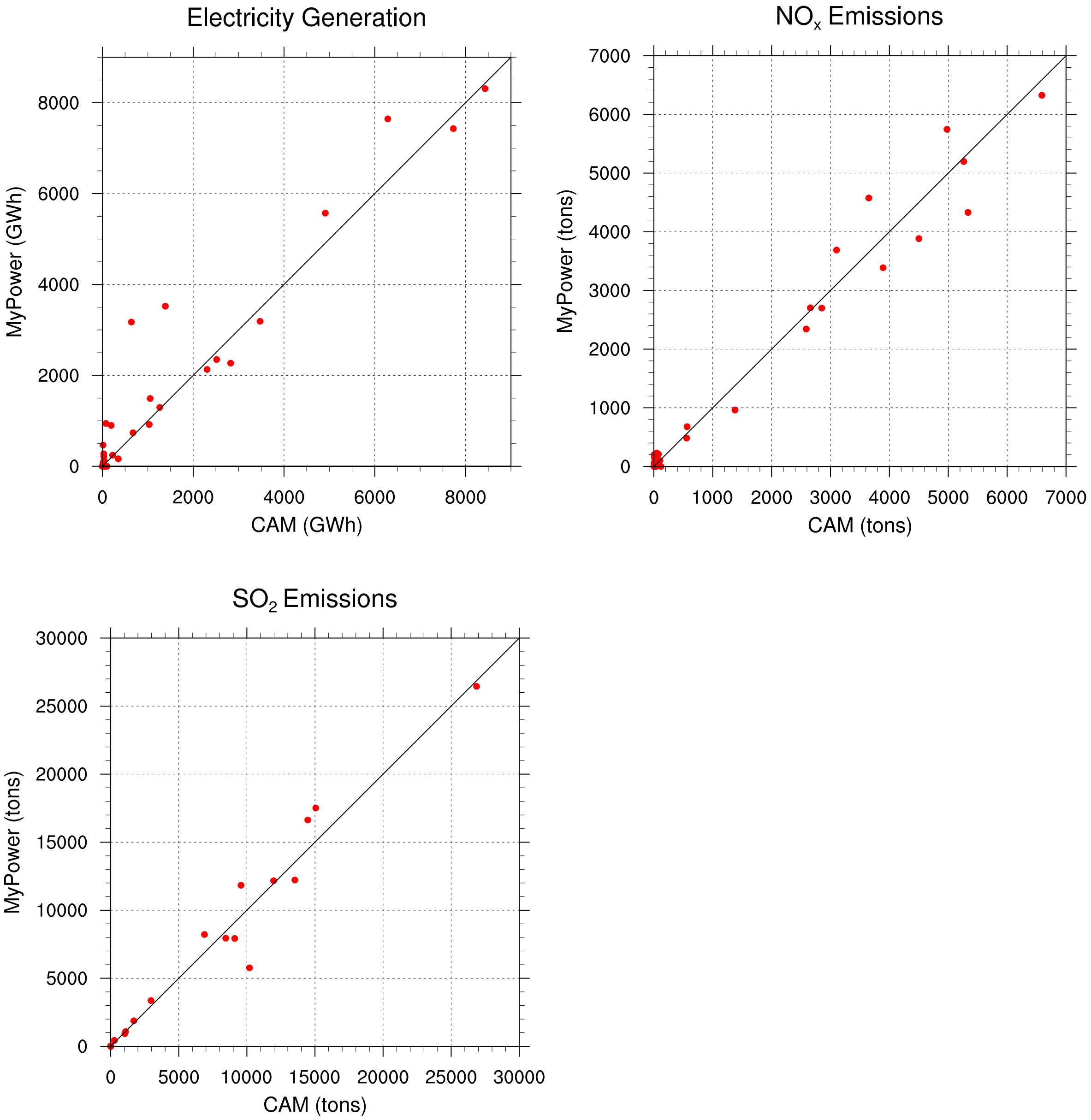
[3] Mean Fractional Bias (MFB) indicates whether the model is – on average – too high (positive MFB) or too low (negative) MFB. The very low 1% MFB here indicates very good performance by the MyPower model.

Figure 1 – Modeled and Reported Generation and Emissions for the 2008-BC Scenario



Scatter plots show annual MyPower model estimates versus CAM reported values for electricity generation, NO_x emissions, and SO_2 emissions for 37 Wisconsin power plants.

Results for the four MyPower scenarios are shown in Table 2. Electricity generation increases by about 35% from the 2008-BC to the 2024-BAU, based on assumed 1.9% annual growth rate. However, the demand reduction programs limit the generation increase in the 2024-RPS and the 2024-OFF to 12% (only a 0.7% annual growth rate). Based on Task Force scenarios, renewable generation increases from 3% in the 2008-BC to 12% in the 2024 BAU resulting from 2,600 MW of new renewable generation capacity (wind, biomass combustion, and biogas) [34]. Wisconsin CO_2 emissions increase by about 19% from the 2008-BC to the 2024-BAU, but decrease by more than 9% from the 2008-BC to the 2024-RPS (a 24% reduction from the 2024-BAU). Because the 2024-OFF achieves the same CO_2 emission reductions (compared to the 2024-BAU) as the 2024-RPS but employs external carbon offsets to meet 35% of these reductions, in-state CO_2 emissions in the 2024-OFF are 11% higher than in the 2024-RPS.

Table 2 – MyPower Electricity and Emission Results

	2008-BC	2024-BAU	2024-RPS	2024-OFF
Generation (GWh)	77,037	104,780	86,193	86,193
Renewable (%)	3%	12%	24%	16%
CO_2 Emissions	59,107,000	70,545,000	53,556,000	59,463,000
NO_x Emissions	51,483	34,279	23,372	27,302
SO_2 Emissions	139,223	85,683	57,527	68,379

Annual values for electricity generation, percent of electricity generated from renewable sources, and emissions (given in tons/year) for the four MyPower electricity scenarios. Because the 2024 CO meets the same CO_2 emission reductions as the 2024 RPS using external carbon credits, in-state CO_2 emissions are higher in the 2024 CO.

Emissions of NO_x and SO_2 in the 2024-BAU decrease 33% and 38%, respectively, from the 2008-BC despite the significant increase in demand. These reductions are due to additional pollution controls on the highest-emitting EGUs and, to a lesser extent, increasing reliance on newer lower-emitting or non-emitting EGUs. The 2024-RPS sees a greater reduction in NO_x and SO_2 emissions (55% and 59% respectively compared to the 2008-BC; 32% and 33% compared to the 2024-BAU) due to the lower electricity demand and increased utilization of renewables. As is the case with CO_2 emissions, NO_x and SO_2 emissions for the 2024-OFF are lower than for the 2024-BAU (20% and 20% respectively) but are higher than for the 2024-RPS because of less renewable generation implemented in Wisconsin.

We have estimated the impacts of carbon reduction policies on plant-by-plant electricity generation using a spatially explicit electricity model (MyPower). We find that MyPower overestimates total generation by about 17%, due primarily to the state-specific study design, which does not take into account electricity import and export. Because the 24 non-coal EGUs are responsible for nearly all (93%) of the overestimation in total generation but only represent a small percentage of total NO_x and SO_2 emissions (5% and 1%), total NO_x and SO_2 emissions from MyPower are within 1% of reported values.

Understanding the Air Quality Impacts of Energy Choices

To understand how calculated emissions from electricity impact air quality, we need to know how the pollution interacts with atmospheric chemistry and weather. To do this, we input calculated changes in NO_x and SO_2 emissions generated from MyPower into the EPA CMAQ model. CMAQ is a three-dimensional chemical transport model, able to quantify the changes in sulfate, nitrate, ozone, and precursors associated with the carbon-reduction policies. The model calculates chemistry and movement of pollution on a three-dimensional grid over the Upper

Midwestern U.S., including inflow from outside the region. Atmospheric chemistry and meteorology are expressed as mathematical equations, and then solved numerically with advanced computer software.

The U.S. EPA CMAQ model, version 4.6 [36], is used to quantify the regional air quality impacts of each emission scenario (2008-BC, 2024-BAU, 2024-RPS, and 2024-OFF). The model has been widely used in assessing $PM_{2.5}$ and ozone sensitivities in the eastern United States, which includes our study domain [15-19, 37]. As a three-dimensional Eulerian chemical transport model, CMAQ hat requires domain grid specification, initial and boundary conditions, meteorology, and gridded emissions as data inputs. Our modeling domain covers the Great Lakes region (GL) with a horizontal resolution of 12 km x 12 km, nested within a 36 km x 36 km continental U.S. (CONUS) domain to provide boundary conditions. Both domains have a 15 layer vertical resolution with shallower (higher resolution) layers near the Earth's surface. Simulations were performed for July 2003, a period of typically high temperatures and stagnant air masses associated with elevated concentrations of sulfate aerosol and ozone.

Meteorology for July 2003 was generated using the Weather Research and Forecasting model, Advanced Research version 3.0 (WRF-ARW) [38] with four-dimensional data assimilation from the North American Regional Reanalysis (NARR) meteorological dataset [39]. WRF was designed for a broad range of meteorological applications and is also employed in regional air quality modeling studies [40-43].

Emissions from both the NEI and MyPower are allocated in space and time using the Spatial Matrix Operator Kernel Emissions (SMOKE) Model, version 2.4 [44]. Scenario-specific Wisconsin EGU emissions are calculated from MyPower, while non-EGU emissions (mobile, area, biogenic, etc.) and EGU emissions outside Wisconsin are obtained from the 2002 NEI [45] for each scenario.

A first step in using a model like CMAQ is to evaluate its skill relative to ground-based measurements. We compared CMAQ ground-level pollutant concentrations from the 2002-NEI scenario with ambient observations for nitrate, sulfate, and ozone for July 2003 over the Great Lakes model domain. Modeled nitrate and sulfate were evaluated with daily mean observations from the U.S. EPA Speciation Trends Network (STN) [47] every 3 or 6 days (depending on the site) and the Interagency Monitoring of Protected Visual Environments (IMPROVE) [48] every 3 days. Modeled ozone was evaluated with maximum daily 8-hour average observations (MDA8, metric used for the NAAQS) from the U.S. EPA Air Quality System (AQS) database [49]. Comparison results are shown in Table 3[4].

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[4] Mean Fractional Bias (MFB) is the average of differences between model and observed data, whereas Mean Fractional Error (MFE) is average of the absolute value of differences between model and observed data. MFE answers "how different are observed and modeled data?" If these differences are equally distributed as over-estimates and under-estimates, we might have a high MFE but low MBE. But, if the model tends to be too high, or tends to be too low, this will be reflected in the MBE.

Table 3 – CMAQ Performance for the 2002-NEI Scenario

	Sites	Mean Obs	Mean CMAQ	MFB	MFE	r^2
Sulfate (*daily mean*)	72	5.81 μg/m^3	4.65 μg/m^3	-20.3%	42.8%	0.838
Nitrate (*daily mean*)	72	0.72 μg/m^3	1.31 μg/m^3	2.22%	88.0%	0.816
Ozone (*MDA8*)	316	51.1 ppb	67.6 ppb	29.4%	30.0%	0.457

Comparison of modeled and observed concentrations for July 2003 over the Great Lakes model domain. Shown are the number of observation sites, the monthly mean observed concentration, the monthly mean modeled concentration (only for observation-model pairs), the mean fractional bias (MFB), mean fractional error (MFE), and r^2 *value.*

CMAQ exhibits good performance for sulfate (mean fractional error, MFE, of 43%), with slight under-prediction at higher concentrations, but over-predicts nitrate with large errors (MFE of 88%). These results are consistent with other studies [15, 50], where summertime CMAQ performance is good for sulfate and more problematic for nitrate. It is worth noting, however, that other studies under-predict 2002 summer nitrate, whereas here CMAQ significantly over-predicts nitrate relative to measured values. This difference in nitrate performance is likely the result of meteorological differences from July 2002 to July 2003, different CMAQ chemical mechanisms employed between studies, and potential differences between emission inventories employed. CMAQ performance for ozone is reasonable (MFE of 30%) with consistent over-prediction, especially at lower concentrations, which is similar to other summertime CMAQ evaluations [51, 52].

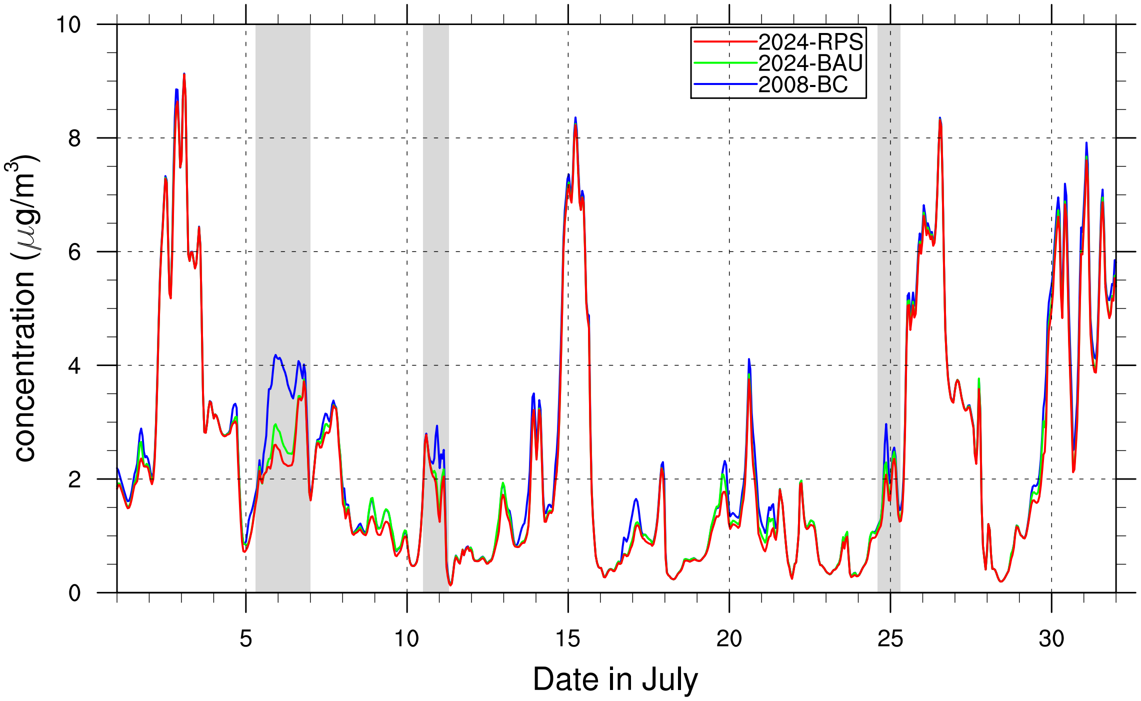
Although these results are within the range past model performance, recent improvements to CMAQ are expected improve these performance metrics. Based on lessons learned from this analysis, we have updated CMAQ to 4.7, SMOKE to v. 2.7.1, and we are now working with the updated NEI 2005 and emissions from LADCO for 2007. One outcome of our study has been this updated evaluation of our modeling tools over Wisconsin and the Midwest. As such, we have identified areas for improving model performance, and ongoing studies are benefiting greatly from the Focus on Energy work.

The discussion below focuses on sulfate PM, which shows the strongest response to proposed energy measures. Monthly mean ozone changes are very small, typically 1% or less, even near power plants. Nitrate reductions are larger, up to 3-7% on a monthly mean basis in some areas, and more significant in select episodes. However, summertime performance of nitrate is poor, and we do not trust the model for this limited simulation. We omit discussion here of summertime nitrate, but ongoing work extends this study to winter, where model performance of nitrate is much better, and uses the newer version of CMAQ, which may also improve summertime nitrate simulations.

To quantify the air quality impacts of the estimated emission reductions, we calculate the difference in ambient concentrations of sulfate, nitrate, and ozone between current (2008-BC), 2024-BAU, and 2024-RPS. Sulfate exhibits the largest monthly mean response to emission reductions during the July 2003 study period. Emissions reductions from the 2008-BC to the 2024-RPS show moderate monthly mean sulfate decreases (0.05-0.2 μg/m^3, 5-9%) throughout much of the state and extending over Lake Michigan into western Michigan (not shown). The largest calculated sulfate decreases (0.2-0.3 μg/m^3, 9-15%) occur in southeastern and south-central Wisconsin, consistent with spatial patterns in emission reductions. Sulfate decreases from the 2024-BAU to the 2024-RPS are less pronounced, given the scheduled SO_2 reductions from additional pollution controls included in the 2024-BAU.

Most relevant to air quality management is the impact of proposed emission reductions on peak events. Figure 2 shows hourly modeled sulfate concentrations for the three 2024 scenarios at Milwaukee, the largest urban area in Wisconsin. At this site, sulfate decreases are largest when sulfate concentrations are between 2 and 4 μg/m^3 (gray bars in figure) and $PM_{2.5}$ concentrations exceed 15 μg/m^3 (the annual $PM_{2.5}$ NAAQS limit) but not 35 μg/m^3 (the daily $PM_{2.5}$ NAAQS). These results suggest that peak sulfate episodes (typically associated with days violating the daily $PM_{2.5}$ NAAQS) in this limited example are largely unaffected by electricity emission reductions, but that the proposed changes would impact what we call here "secondary episodes" (days with concentrations above the annual NAAQS limit).

Figure 2 – Modeled Sulfate Decreases at Milwaukee, WI



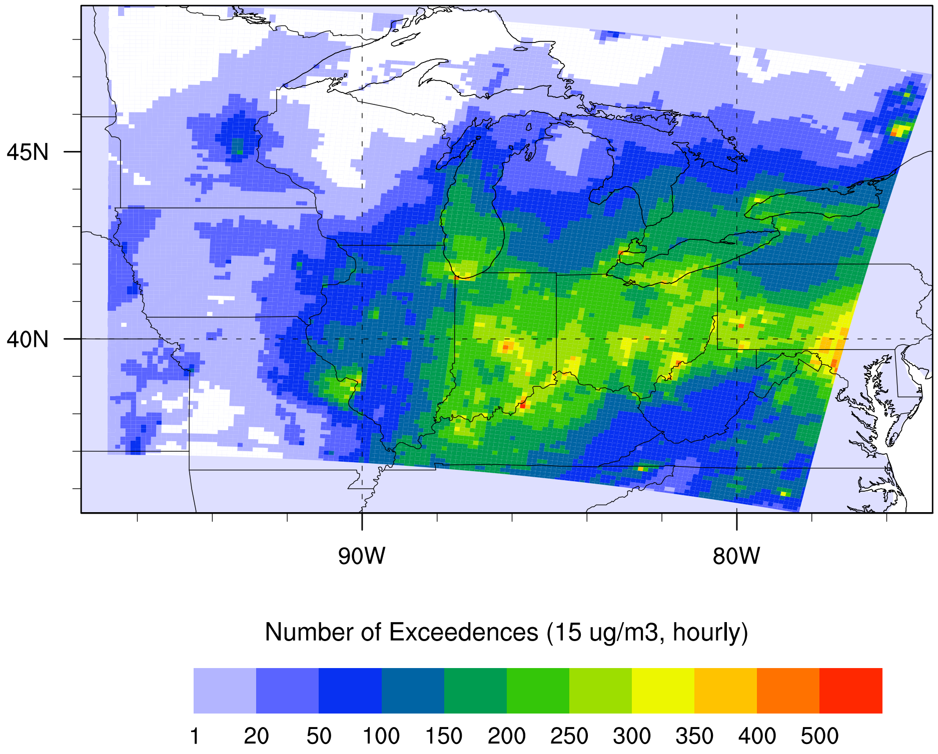
Hourly sulfate concentrations for the 2008-BC, 2024-BAU, and 2024-RPS during the July simulation period. Gray bars indicate the episodes of greatest sulfate decrease.

To extend this type of episodic analysis to our entire study region, we quantify the impacts of electricity emission reductions (2008-BC to 2024-RPS) on peak $PM_{2.5}$ and ozone episodes. For $PM_{2.5}$, we examined several peak thresholds for analysis, including daily mean values above 35 μg/m^3, hourly values above 35 μg/m^3, daily mean values above 15 μg/m^3, and hourly values above 15 μg/m^3. Here, we focus on hourly $PM_{2.5}$ values above 15 μg/m^3, the maximum allowable annual average concentration according to the EPA NAAQS, and the breakpoint for the EPA Air Quality Index to switch from "Good" (green) to "Moderate" (yellow). Our simulation covers a single month, so this value maximizes the number of data points available for analysis. Results for other $PM_{2.5}$ peak metrics show similar spatial patterns but with a more limited number of values in the July 2003 period.

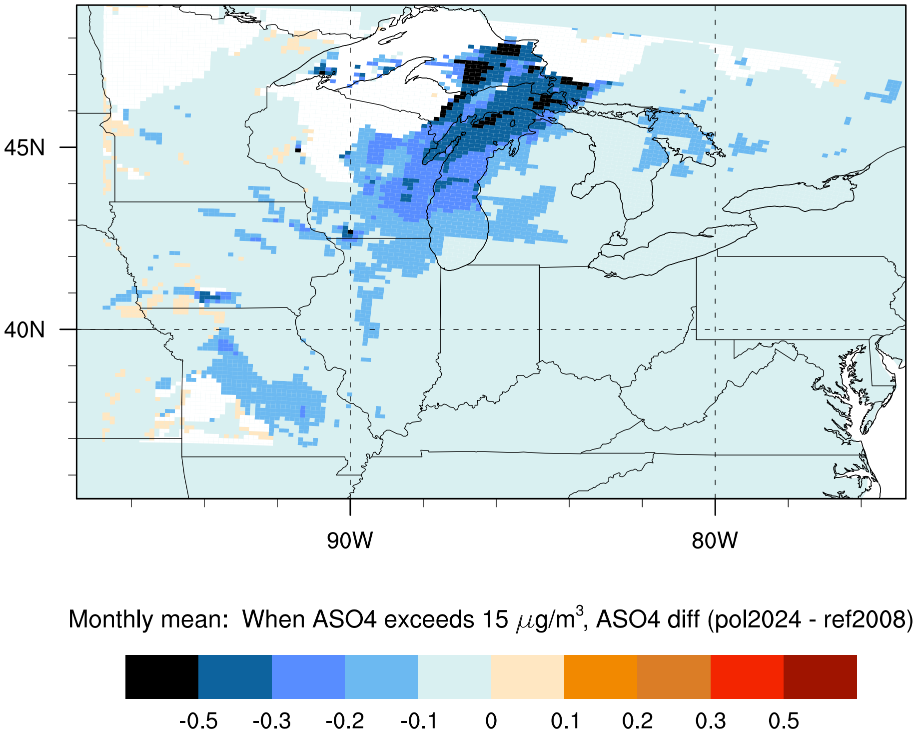
Model-calculated hourly $PM_{2.5}$ values above 15 μg/m^3 for the 2008-BC during July 2003 (Figure 3a) are greatest along and north of the Ohio River (200-350 occurrences) and in the Mid-Atlantic (250-400). Hourly occurrences in Wisconsin range from 150 in the far southeast to none in the north. During these >15 μg/m^3 $PM_{2.5}$ periods, the mean difference in sulfate between the 2008-BC and 2024-RPS scenarios (Figures 3b and 3c) exhibits a maximum over northern Lake Michigan, the Upper Peninsula of Michigan, and eastern Lake Superior, where sulfate constitutes 25-50% of total $PM_{2.5}$ (not shown). In these regions, during hours with $PM_{2.5}$ above 15 μg/m^3, CMAQ estimates sulfate decreases of 0.3-0.8 μg/m^3 or 5-15%.

Figure 3 – Impacts of Emission Reductions on Peak $PM_{2.5}$ Events

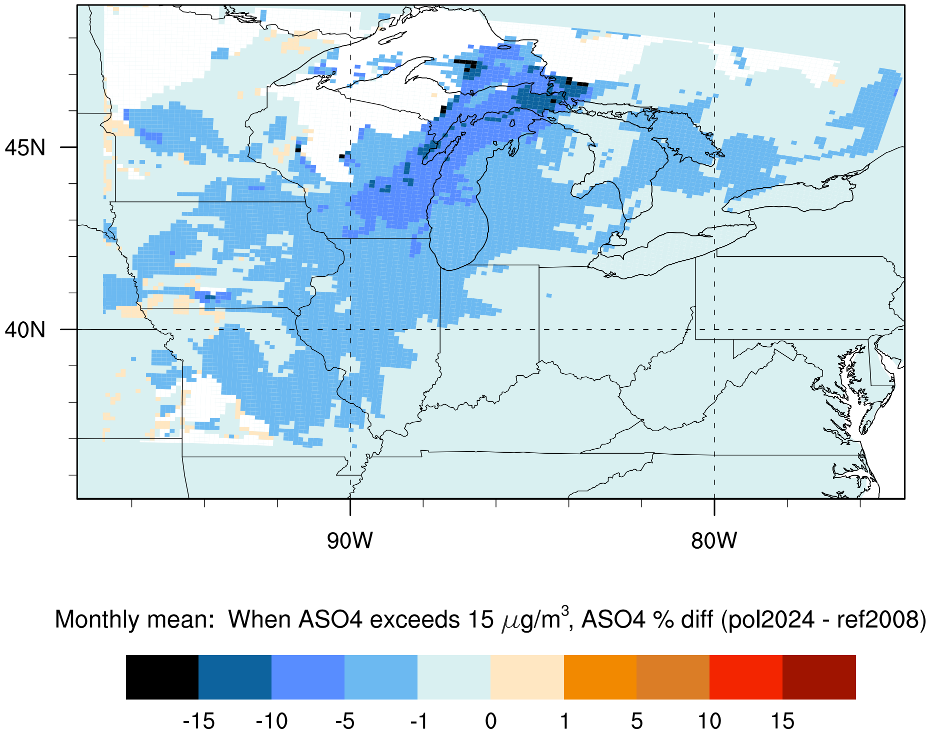
3a) The number of hours with $PM_{2.5}$ values above 15 μg/m^3 during July 2003 (744 total hours in month, white grid cells signify no hours above threshold)



3b) Mean sulfate absolute difference ($\mu g/m^3$) from 2008-BC to 2024-RPS during hours with $PM_{2.5}$ values above 15 $\mu g/m^3$



3c) Mean sulfate percentage difference (%) from 2008-BC to 2024-RPS during hours with $PM_{2.5}$ values above 15 $\mu g/m^3$



For ozone, we performed analysis for hourly values above 75 ppb (not shown). Model-calculated hourly ozone above 75 ppb for the 2008-BC during July 2003 is highest around Lake Erie (125-250 occurrences), Lake Michigan (125-200), and western Pennsylvania (125-200). CMAQ simulates far southeast Wisconsin as having 125 hours with ozone in excess of 75 ppb, and 10-50 hours for much of the rest of the state. During these elevated ozone periods, the mean ozone difference from the 2008-BC to the 2024-RPS is largest in plumes downwind of power plants with the largest NO_x reductions, with decreases of 0.4-1.4 ppb, typically <1%.

Using this methodology for Wisconsin, we find that power sector carbon reduction policies significantly reduce statewide emissions of NO_x and SO_2 compared to the business-as-usual scenario, with smaller reductions for policies utilizing external carbon offsets. The estimated patterns of emission reductions interact with the unique atmospheric environment of the Great Lakes region (e.g. weather, land cover, other emissions), producing spatially specific decreases in sulfate, nitrate, and ozone. Sulfate exhibits the largest monthly mean response to emission reductions. Both sulfate and nitrate exhibit significant responses during high $PM_{2.5}$ episodes (decreases of 5-15% for sulfate and 3-20% for nitrate centered over northeastern Wisconsin, the upper peninsula of Michigan, and the surrounding lakes), but model performance for nitrate is poor. Thus, the most significant findings from this work deal with the strong potential to reduce summertime sulfate, contributing to high-PM episodes, with increased efficiency and renewable energy generation.

In addition, the apparent above-lake amplification of emission reductions for sulfate, simulated both in the monthly mean and episodic impacts, suggests that Lake Michigan may promote sulfate formation in a matter analogous to ozone and secondary organic aerosol. Further research is needed to validate this model-based hypothesis, and to understand whether above-lake processes relate to coastal sulfate exposure.

Energy and Environmental Research for Wisconsin

These results highlight the value of an integrated modeling approach for coordinated management of air quality, carbon reduction, and energy policies. Effective decision-making on energy policies and technology investment requires an understanding of air quality co-benefits, along with other major environmental and economic priorities. Quantifying the impacts of lower-carbon electricity policies using this type of methodology could help inform the broader environmental and health impacts of energy and climate policies.

We have presented our work at a wide range of local and national venues, including the 2010 Focus on Energy meeting[5], the University of Wisconsin—Madison Law School, the 2010 Nelson Institute Earth Day Conference[6] and the Wednesday Night @ the Lab series, which has since aired on Wisconsin Public Television. Beyond sharing our work, and broader issues on environmental science, with a Wisconsin audience, this work has contributed to invited talks at other U.S. universities and it has been presented at the American Geophysical Union 2010 Fall Meeting. Project funds were used to support graduate and undergraduate students at the

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[5] http://www.focusonenergy.com/files/Document_Management_System/Environmental_Research/hollowayclimate_ppt.pdf

[6] http://www.nelson.wisc.edu/partnerships/programs/earth_day/docs/holloway.pdf

University of Wisconsin—Madison, all of whom are now employed (or scheduled for employment) at leading Wisconsin companies.

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Appendix A:

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Implications of incorporating air-quality co-benefits into climate change policymaking

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Abstract
We present an analysis of the barriers and opportunities for incorporating air quality co-benefits into climate policy assessments. It is well known that many strategies for reducing greenhouse gas emissions also decrease emissions of health-damaging air pollutants and precursor species, including particulate matter, nitrogen oxides, and sulfur dioxide. In a survey of previous studies we found a range of estimates for the air quality co-benefits of climate change mitigation of \$2-196/tCO_2 with a mean of \$49/tCO_2, and the highest co-benefits found in developing countries. These values, although of a similar order of magnitude to abatement cost estimates, are only rarely included in integrated assessments of climate policy. Full inclusion of these co-benefits would have pervasive implications for climate policy in areas including: optimal policy stringency, overall costs, distributional effects, robustness to discount rates, incentives for international cooperation, and the value of adaptation, forests, and climate engineering relative to mitigation. Under-valuation results in part from uncertainty in climatic damages, valuation inconsistency, and institutional barriers. Because policy debates are framed in terms of cost minimization, policy makers are unlikely to fully value air quality co-benefits unless they can be compared on an equivalent basis with the benefits of avoided climatic damages. While air quality co-benefits have been prominently portrayed as a hedge against uncertainty in the benefits of climate change abatement, this assessment finds that full inclusion of co-benefits *depends on*—rather than substitutes for—better valuation of climate damages.

Keywords: co-benefits, climate policy, air pollution, health

1. Introduction

Changing the energy system in order to stabilize the climate is likely to have a wide variety of effects that are not directly related to greenhouse gas emissions, including human health, macro-economic, geo-political, eco-system, agricultural yields, and employment patterns. Those effects that are favorable to human welfare are often termed 'co-benefits'. The use of the term *benefits* reflects the situation that decisions related to whether, how, and how much to address climate change are typically made with some consideration of the costs and benefits associated with various policy options. These decisions however do not usually consider the full range of effects of actions to address climate change. Among the most important of known co-benefit effects are those associated with air quality and the resulting impacts on human health. Changes in the technologies used to produce and consume energy, as well as the level of energy consumption, have two effects related to air quality. First, many of the changes that would reduce greenhouse gas emissions would reduce other emissions as well, such as nitrogen oxides (NO_x), sulfur dioxide (SO_2), particulate matter, and mercury, and the resulting pollution-related disease. Second, many of these changes would obviate the need for expensive pollution-control equipment—such as flue-gas desulfurization, selective catalytic reduction, and electrostatic precipitators—in order to comply with air quality regulations. How important are air quality (AQ) co-benefits? Why are they not considered in assessments of climate policy design? A primary finding is that the focus on cost minimization—rather than comparison of benefits and costs—diminishes the role of benefits in general. As a result, well-established AQ benefits are not a central part

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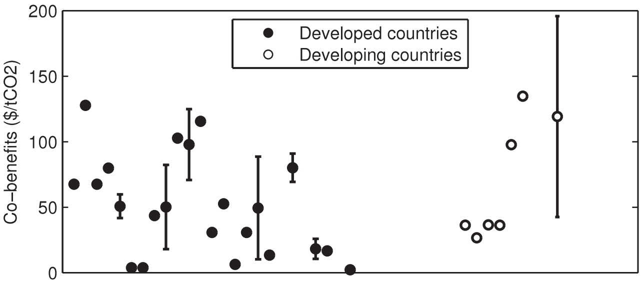

Figure 1. Estimates of the value of air quality co-benefits in developed (left) and developing country studies (right) in 2008\$/tonCO_2. Within each category, data are reported from left to right by date of study (1991–2010). Absence of values indicates a co-benefit study for which health impacts were assessed, but valuation in \$/tCO_2 was not.

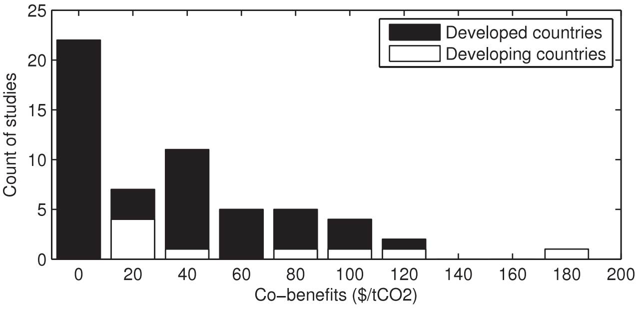

Figure 2. Frequency of values reported in air quality co-benefits studies.

of the climate policy discourse and probably rely on better characterization of climatic benefits in order to be fully valued.

We first review estimates of the value of air quality benefits of climate change policies and in section 3, the extent to which these co-benefits are valued in integrated assessment models. We then discuss the policy implications of including AQ co-benefit considerations in climate policy decision making and explore the reasons why economic policy models tend to ignore, even if they acknowledge, the value of co-benefits. We discuss data and modeling needs to resolve the existing impasse.

2. The value of AQ co-benefits is large

A large set of studies now makes clear that the magnitude of AQ co-benefits of climate change mitigation are non-trivial and have been observed across varied geographies, time periods, and sectors. We surveyed 37 peer-reviewed studies of AQ co-benefits (see the appendix). These studies provided 48 estimates of the economic value of air quality benefits of climate change mitigation, and span diverse geographies, time horizons, valuation techniques, and involve different mixes of economic sectors contributing to mitigation. Because the perspective of this study is on policy making amidst competing social priorities, we restricted our survey to those studies that (1) calculated an economic value of co-benefits, and (2) expressed values in terms of \$/ton of CO_2 avoided. This restriction means that we do not include the results from a number of the studies we surveyed, and a larger portion of the studies of developing countries.

In figure 1, studies of developed countries are shown on left and those of developing countries on right. Within each category, data are reported from left to right by date of study (1991–2010), consistent with the studies reported in the appendix tables. Absence of values indicates a co-benefit study for which health impacts were assessed, but valuation in \$/tCO_2 was not assessed. All values have been converted into constant 2008 dollars. Note that economic valuation was more frequent in developed country studies; 17 out of 24 developed country studies included \$/tCO_2 estimates compared to 2 out of 13 developing country studies.

Figure 2 shows the frequency of values cited across all studies. The values for developed countries are in black and those for developing countries in white. For the 22 estimates from the 24 developed country studies the range was \$2-128/tCO_2, the median was \$31/tCO_2 and the mean \$44/tCO_2. For the 7 estimates from the 13 developing country studies the range was \$27-196/tCO_2, the median was \$43/tCO_2 and mean was \$81/tCO_2. Values are generally higher in developing countries, although the difference in means is not significant ($0.10 < p < 0.05$) in part due to variation in sector assessed and the dearth of developing country studies that assign economic value to co-benefits.

Heterogeneity in the distribution of study results is partially attributable to constraints on the scalability of AQ co-benefits at more stringent emissions reduction levels. At higher levels of greenhouse gas (GhG) abatement, abatement costs rise but AQ co-benefits remain constant (Burtraw *et al* 2003). Moreover, the apparently higher values in developing country studies result from these countries beginning with higher pollution levels, at which incremental health benefits are large. As emissions reductions become more aggressive, AQ co-benefits play a smaller role. Thus, valuation of AQ co-benefits is most important in the early stages of a long-term climate change mitigation strategy, and most important for developing countries lacking significant air quality management programs.

3. AQ co-benefits are not included in climate policy analyses

Even though the AQ co-benefits of climate change actions are well established, policy analyses typically do *not* account for them. We surveyed 13 major climate policy assessments based on integrated assessment models, selecting based on prominence and their intention to specifically inform policy decisions related to climate change. We drew from those used by the Intergovernmental Panel on Climate Change (IPCC), as well as government sponsored reports to model the impacts of specific policies in the UK and US. With one exception, the models reviewed are integrated assessments in that they combine assessments of both the physical and economic impacts of climate policies. Most of the models listed in table 1 (A, B, D–G, I, K–M) are partial or general equilibrium models, known as *top-down* models, which assess the direct and indirect economic effects of policies. Two (C, H) are systems engineering models that include technological detail and take a *bottom-up* approach. Model J is a benefit-cost analysis. In most cases the objective function is based on minimizing the abatement cost of meeting a climate emissions

Table 1. Treatment of AQ co-benefits in integrated assessment models of climate change policy.

	Venue	Model name	Time	GhG emissions	Value climate impacts	Estimate AQ co-b.	Value AQ co-b.	Include in final values
A	IPCC	IMAGE[a]	2100	Yes	No	No	—	—
B	IPCC	MERGE[a]	2150	Yes	No	No	—	—
C	IPCC	MESSAGE[a]	2100	Yes	No	No	—	—
D	IPCC	MiniCAM[a]	2100	Yes	No	No	—	—
E	IPCC	SGM[a]	2050	Yes	No	No	—	—
F	IPCC	WIAGEM[a]	2100	Yes	No	No	—	—
G	Nordhaus (2008)	DICE-2007[b]	2200	Yes	Yes	No	—	—
H	UK	C.C. Act of 2008 Assessment (MARKAL)[c]	2050	Yes	Yes	Yes	Yes	Yes
I	UK	Stern 2005/PAGE2002[d]	2200	Yes	Yes	Yes	Yes	No
J	US	C.B.O. (2009)[e]	2019	No	No	No	—	—
K	US	EIA NEMS (2008)[f]	2030	Yes	No	No	—	—
L	US	EPA ADAGE (2008)[g]	2050	Yes	No	No	—	—
M	US	EPA IGEM (2008)[g]	2050	Yes	No	No	—	—

[a]IPCC (2007). [b]Nordhaus (2008). [c]DECC (2008). [d]Stern (2006). [e]CBO (2009). [f]EIA (2008). [g]EPA (2008).

goal; climate damage costs are excluded. Only one (I) maximizes welfare by accounting for the benefits of avoided damages. The following section discusses why this final distinction is especially relevant to the treatment of AQ co-benefits.

Although 12 of the 13 models surveyed estimate emissions of greenhouse gases, only three (G, H, I) estimate the value of the resulting climate change damages. The others simply minimize the costs of achieving a specified set of annual emissions targets. Of the three that do estimate both costs and benefits of climate policy, only two (H, I) estimate air quality benefits—and only one of those (H) includes these values in the final cost estimates. The Stern review (I) does discuss AQ co-benefits and even quantifies them in dollar terms as 'up to 1% of GDP' (Stern 2006). But crucially, that study excludes this value in their highly publicized final results of the impacts and costs to address climate change. Only the UK Climate Change Act 2008 Impact Assessment (H in table 1) includes a value for improved air quality (£32b) in their final estimate (DECC 2008).

Beyond these high profile studies, recent work provides examples of more comprehensive inclusion of AQ co-benefits. Ostblom and Samakovlis (2007) include co-benefits in a CGE model for Sweden and find that the costs of climate policy are overstated if they are excluded. Bollen *et al* (2009) adapt a version of model B above to perform a cost-benefit analysis that includes both climatic and AQ impacts; they find the AQ co-benefits twice as large as climatic benefits. Early results from models such as GAINS combine estimates as well (Amann *et al* 2009).

An essential problem hindering inclusion of AQ co-benefits in policy decisions is that debates are framed in terms of minimizing the costs of climate policy. Because the benefits of avoided climate change are not explicitly considered, AQ benefits must somehow be compared to abatement costs[4]. Abatement levels are typically chosen exogenously with very little explicit justification for the specific targets adopted. For example, some targets attempt compliance with the ambiguous objective of avoiding *dangerous interference with the climate system,* as agreed on in the 1992 UN Framework Convention on Climate Change (Kriegler 2007). If full benefit-cost analyses were performed, the valuation of AQ co-benefits would be much simpler, as the addition of AQ co-benefits would imply a more stringent level of pollution abatement. The left panel of figure 3(a) shows that inclusion of air quality impacts would shift the marginal damages cost curve (MDC) upward so that its intersection with the marginal abatement cost curve (MAC) move to the right and as a result, the optimal level of pollution abatement would increase from q^* to q'. In practice, however, optimizing the level of emissions is not the objective of policy makers and is not the approach taken by analysis to inform them.

With exogenously specified targets, the marginal damages of climate change do not influence choices among policy options. Rather, the goal of policy design is to minimize the cost of meeting previously selected abatement levels. Inclusion of AQ co-benefits is less straightforward in this situation because policy debates are focused on the costs of pollution abatement; benefits are not a central part of the policy discourse. From this perspective, AQ co-benefits have to somehow affect the slope or position of the marginal abatement cost curve, rather than the damage curve. For example, the right panel of figure 3(b) shows that addition of AQ co-benefits could be interpreted as shifting the MAC curve downward. The marginal damage curve has been removed from that panel because it does not affect decisions. This shift requires the awkward re-interpretation of the AC as the sum of climate change abatement costs and AQ co-benefits ($\mathrm{MAC}_{\mathrm{CC}}+\mathrm{MDC}_{\mathrm{AQ}}$). The shift reduces the cost of climate policy such that the marginal cost, given the exogenously selected abatement level q^*, falls from p^* to p' as a result. The cost of the policy has gotten cheaper for the same level of emissions reductions. Most co-benefits studies and their normative policy claims result from conceiving of the abatement cost curve as this hybrid of climate costs and AQ benefits, even if estimation of p' is rarely explicit. For example, claims of 'no regrets'

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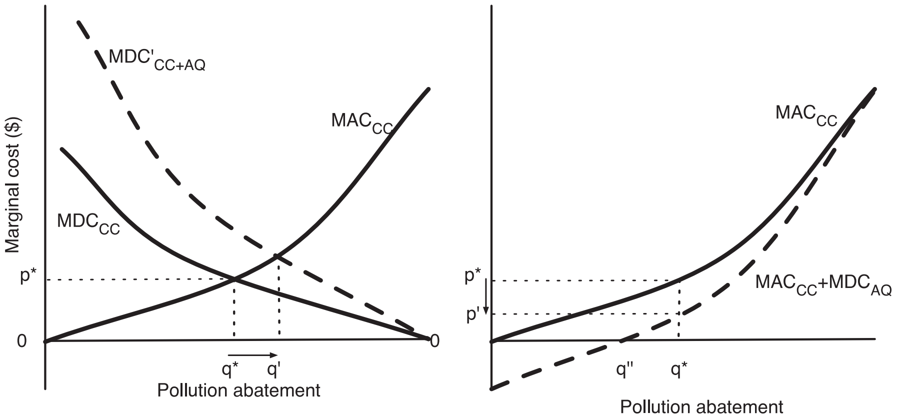
[4] While it is optimal to use one policy instrument for each source of market failure, in reality the climate policies in discussion today include dozens of policy instruments within each piece of legislation. In part this is due to the presence of multiple market failures (Jaffe *et al* 2005).



(a.)

(b.)

Figure 3. Effect of inclusion of air quality co-benefits on the marginal cost of climate policy. Left panel (a) shows air quality co-benefits interpreted as avoided damages. Right panel (b) shows air quality co-benefits interpreted as reducing abatement costs.

climate policy refer to the existence of abatement opportunities to the left of q'' where policy costs are below zero due to positive co-benefits. Rather, we are given p^* and told it is an overestimate—even in studies as thorough and as prominent as the Stern review. Full valuation of AQ co-benefits requires a more explicit discussion of how these cost impacts are calculated.

4. Implications of including AQ co-benefits

More thorough inclusion of AQ co-benefits would have several important effects on climate policy debates—both on optimal design and on positions held by stakeholders. The first implication is that inclusion of AQ co-benefits will reduce the societal cost of climate policy, as in figure 3(b). Alternatively, co-benefits may justify more stringent climate change policy by increasing the avoided societal damages, as in figure 3(a). Second, co-benefits improve the robustness of stringent climate policy. Acknowledging uncertainty in both the damage function and the abatement cost function, inclusion of AQ co-benefits provides a hedge against lower than expected climate damages or higher than expected mitigation costs. AQ co-benefits also occur earlier than climatic ones, making the social benefits calculation less sensitive to the choice of discount rate, thereby diminishing the significance of using low (Stern and Taylor 2007) or high discount rates (Nordhaus 2007). By increasing the robustness of climate policy to uncertain damages, abatement costs, and discount rates, co-benefits support more aggressive near term climate action even in the face of large uncertainty (Manne 1995).

An extension of this set of arguments on lower costs, higher stringency, and robustness is that inclusion of co-benefits provides stronger incentives for cooperation from developing countries than do climatic benefits alone. Due to lower incomes, an earlier stage of development, and negligible historical contribution to the stock of atmospheric greenhouse gases, rapidly growing developing countries are particularly sensitive to abatement costs and have shown little enthusiasm for reducing emissions. However, reducing their emissions from the trajectory of the last decade is essential to addressing the global problem. Game theoretic models show that the nearer term and more localized AQ co-benefits of climate change mitigation might be sufficiently important to developing countries that they would participate in international agreements (Pittel and Rubbelke 2008). Indeed, in figure 2 the value of AQ co-benefits in developing countries appears higher than in developed countries, although not significantly so given the few valuation studies in developing countries.

A second main implication is that including AQ co-benefits has a distributional effect because it changes the beneficiary of climate change actions. In particular, as the geographic benefits of international offset projects in the energy sector become more local, the value of offset projects for developing countries increases because the value of AQ co-benefits are added to the value of financial transfers from developed countries. As a result, entities in developed countries should expect to pay lower prices for offset projects in developing countries, while the value of domestic mitigation in developed countries will also increase. Thus, the cost of carbon mitigation decreases for both domestic and international abatement measures. A comparison of the value of co-benefits in developed countries in section 2 above (median $=$ \$31/tCO$_2$) to the prices paid for offsets at present (∼\$20/tCO$_2$) suggests that developed countries may prefer local mitigation, which creates AQ co-benefits, over purchasing international offsets; many international offset projects will be more expensive than domestic projects, even if international offsets would be cheaper with AQ co-benefits valued than without. The valuation of local AQ co-benefits is likely to have a diminishing effect on the flow of offset funds from developed to developing countries. This outcome suggests that the goal of financial transfer from developed to developing countries would be more effectively accomplished through direct support for activities, such as adaptation and poverty alleviation, rather than relying mainly on international offset projects as the transfer mechanism.

A related issue is that the geographic dispersion of the benefits of mitigation will become more closely tied to location of emissions. A fundamental justification behind GHG emissions trading is that the atmosphere is indifferent to the location of emissions since the six greenhouse gases regulated under the Kyoto protocol are long lived and are well mixed throughout each hemisphere (for methane) or the globe (for others, including CO_2). The broadening of scope from climate benefits to air quality benefits raises the importance of the location of emissions. Given the wide dispersion in the costs to reduce GHG emissions, it is possible that trading could concentrate emissions in locations with high abatement costs (Farrell and Lave 2004). While the development of such *hotspots* does not affect the geographic incidence of climatic damages, it would introduce environmental justice concerns if air pollution health effects become concentrated as a result.

Third, actions that are equivalent in radiative forcing are not equivalent in value. Inclusion of AQ co-benefits increases appeal of transforming energy production and use relative to other means of addressing climate change, which have less pronounced effects on air quality. For example, the appeal of forest preservation will diminish relative to emissions mitigation when AQ co-benefits are included—though of course valuation of other co-benefits such as biodiversity would increase the relative appeal of forests. Similarly, AQ co-benefits reduce the attractiveness of adaptation and climate engineering relative to mitigation. To be sure, adaptation is still necessary, but its role as an appealing alternative to costly mitigation is diminished. Concerns about climate engineering schemes that propose reducing radiative forcing without necessarily changing emissions have been raised due to uncertainties about efficacy and side effects (Bengtsson 2006). Indeed, some solar radiation modification schemes have the potential to reduce air quality (Crutzen 2006, Victor 2008), and even those with no adverse affect must take into account the opportunity cost of missed air quality improvements. The observed under-prioritization of adaptation and climate engineering relative to mitigation (Pielke *et al* 2007) may be partially attributable to concern over the loss of AQ co-benefits, even if not explicitly expressed.

Finally, it is not obvious that all climate change mitigation actions that provide AQ co-benefits will be pursued. Policy makers may simply choose to address AQ directly since it is almost certainly cheaper to reduce local air pollution directly rather than via climate policy (Johnson 2001). This possibility seems especially pertinent in developing countries where, for the reasons discussed above, climate change mitigation has to date been considered a developed country responsibility. It may also be a concern at higher levels of GhG mitigation where abatement costs become expensive and AQ co-benefits start to look relatively small. It may become reasonable for countries, especially developing ones, to consider avoided climate change damages as a co-benefit of efforts to reduce air quality. If high-CO_2-emitting developing countries were to take such a perspective, it would complicate implementation of an international climate agreement. For example, emissions trading between countries would be difficult if one country were to set a national limit on GHG emissions while the other had a national limit on SO_2, NO_x, or other pollutants. Although it may ultimately prove essential to overcoming international collective action problems, it would require a high degree of flexibility and a tolerance for heterogeneity in national implementation plans that goes well beyond what has been agreed upon so far in the international climate regime.

5. Why are AQ co-benefits acknowledged but ignored?

Given these implications, ignoring co-benefits skews policy decisions and leads to sub-optimal social outcomes. Many studies discuss the benefits of a more comprehensive assessment and policy (IPCC 2007, Haines *et al* 2007, Bond 2007). If AQ co-benefits are so substantial and their implications so important, why do not they play a larger role in affecting climate policy design? Several characteristics of AQ co-benefits contribute to their under-valuation.

5.1. Uncertainty in climatic damages and abatement costs

Uncertainty about both the costs and benefits of climate change mitigation reduces the role of air quality benefits in policy debates because it complicates comparisons. This is in contrast to prominent arguments that assert that AQ co-benefits make *no regrets* climate policy possible because the greater certainty of AQ co-benefits reduces the importance of uncertainty over climatic damages. However, the large uncertainty over the benefits of avoided climate change has shaped the policy discourse so that policy design is framed as a problem of cost minimization; benefits are not counted explicitly because estimates are not sufficiently reliable. The resulting marginalization of climatic benefits has had the effect of excluding quantitative representation of benefits in general, including AQ benefits. AQ co-benefits have so far not diminished the importance of climatic uncertainty; rather, deep and persistent climatic uncertainty has led to a policy discourse in which it is extremely difficult for AQ benefits to play a central role.

Cooperation on climate change is difficult in part because the abatement costs in climate policy are so uncertain (Swart *et al* 2009). Claims are made both that climate policy will cost several per cent of gross world product and that climate policy will actually stimulate economic growth (Tol 2009). Estimates reported by the IPCC alone show a range of carbon prices from \$20-100/tCO_2 for 25% emissions reduction from business as usual by 2030 (Nemet 2010). That almost every climate policy proposal involves a quantity-based target rather than price-based target sustains cost uncertainty. In practice, assumptions about base case emissions growth, the supply of loss-cost energy efficiency investments, the cost of renewables, the diffusion of nuclear, and the availability of carbon capture and sequestration technology, as well as other items, leads to large dispersion in abatement costs. In contrast, the technologies involved in air quality improvement are less dynamic, have a longer history, involve a much more limited set of options, and do not require changes to existing infrastructures.

While the overwhelming portion of the discussion on climate policy is focused on abatement costs, the more important source of uncertainty for AQ co-benefits arises from in climate damages. More specifically, estimates of the climate-related damages *avoided* as a result of climate policy are the central concern for policy makers. Estimation of avoided damages involves ‘deep uncertainty’ because reliable probability distributions of possible outcomes are not available (Lempert 2002, Keller *et al* 2008, Gosling *et al* 2009). One recent survey of published estimates found a range of climate damages from \$0-33 000/tCO_2, depending on assumptions related to risk aversion, equity, and time preferences (Anthoff *et al* 2009). Of particular concerns is the potential for positive feedbacks, irreversibility and rapid change to the climate system (Torn and Harte 2006). In contrast, estimation of AQ damages is less problematic, in part because the effects of air pollution on human health are nearer term, less geographically dispersed, and are well studied.

Even though damages are the ultimate motivation for climate policy, as shown above, they are not typically included in assessments of climate policy. One interpretation is that we simply distrust the reliability of climate impact studies. An alternative hypothesis is that since the uncertainties are so large and values hinge on choice about small changes in discount rates, that discussion quickly becomes philosophical, and not amenable to policy discourse. Another reason that damage values are infrequently discussed is that willingness to pay to avoid them appears quite low; a contingent valuation study of willingness of US residents to pay for the Kyoto Protocol estimated that households valued the benefits at just under \$191 per household per year (Berrens *et al* 2004), which implies political support for a carbon price in the mid-single digits of \$/tCO_2. More broadly, contingent valuation studies suffer from ignorance about what type of climate people actually want (Dietz and Maddison 2009). Finally, the characteristics of the risks being compared are different (Slovic 1987); the lethal aspects of the health impacts of air pollution may provide a catalyst for regulatory action that, at least at present, is missing in climate change.

5.2. Measurement and valuation

Another reason that AQ co-benefits are typically excluded is that valuation results are sensitive to choices about methodology and parameter values (Bell *et al* 2008). Even if the benefits are widely found to be substantial, standard metrics for economic valuation of health impacts do not exist, which is a particular problem in valuing loss of life and assessing heterogeneous sub-populations. Development of 'Health Impact Assessment' provides one avenue to remedy this problem (Patz *et al* 2008). Valuation of health and life is made worse by disagreement over the appropriate discount rate to use (Stern and Taylor 2007, Nordhaus 2007, Anthoff *et al* 2009). The smaller temporal and geographical scales of AQ impacts relative to climatic impacts make comparison difficult as well. The more diverse set of pollutants that need to be taken into account to optimize the pursuit of AQ and climate benefits, combined with the nearer term impact of AQ impacts, heightens the sensitivity of valuation results to choices of global warming potentials to compare gases (West *et al* 2007, Smith and Haigler 2008). Finally, some have suggested that the transactions and information costs associated with AQ co-benefits are so high that they would offset incremental benefits (Elbakidze and McCarl 2007); however, the values found in section 2 imply that those costs would have to be extremely high. The paucity of studies that value co-benefits in developing countries—for example in figure 1—suggests that the challenges of valuation are even more problematic in those contexts.

5.3. Institutions and epistemic communities

Institutional barriers, in both the scientific and political domains, also discourage inclusion of co-benefits. Scientifically, the networks of institutions and individuals contributing knowledge on air quality have little overlap with those on climate change (Swart 2004). The lack of shared assumptions, methods, and data makes integration of scientific results difficult (Norgaard 2004). The international policy regime reflects a similar separation; the UN Framework Convention on Climate Change and the Convention on Long-Range Transboundary Air Pollution remain separate despite calls for better integration (Holloway *et al* 2003). The adverse consequences of this division of international governance are likely to heighten if countries adopt divergent priorities on climate change and air quality. For example, large developing countries might value avoided climatic damage as a co-benefit of their pursuit of air quality improvement while developed countries might focus on climate impacts directly, with AQ as an ancillary benefit. In effect, climate change may become an 'impure' public good, with private gains from mitigation alleviating free-rider issues (Finus and Ruebbelke 2008). While heterogeneous pursuit of common outcomes might provide a promising context with which to resolve collective action problems, the separation of governance regimes is likely to impede progress. Finally, the implications described above may realign interest group coalitions that are affecting the political process in favor of action on mitigation. The relative decline in the attractiveness of afforestation, adaptation, and climate engineering once AQ co-benefits are taken into account, may threaten the cohesion of coalitions of support of climate policy at the national and international levels. Adding complexity to an already complex regime may reduce salience and consequent political feasibility as well (Young 1989, Rypdal *et al* 2005). This challenge need not be paralyzing; a US Senate committee passed a 'four pollutant bill' for CO_2, SO_2, NO_x, and Hg in 2002 (S.556) and Senators were discussing introducing a similar bill in late-2009.

6. Conclusion

The full inclusion of AQ co-benefits in the design and evaluation of climate policy would almost certainly enhance social outcomes because these co-benefits are large and because policy analysis has not valued them. Moreover, that AQ co-benefits are more local, nearer term, and health related has the potential to enhance incentives for cooperation by engaging actors that are averse to the costs of climate policy or unmotivated by avoided climatic damages. Still, a variety of barriers exist to their inclusion. The framing of the climate policy discourse is likely to continue as one of cost minimization until the benefits of avoided climate change can be more reliably estimated. As a result, a risk remains that AQ co-benefits will be treated as serendipitous and tangential, rather than as driving forces for strong climate policy. Full consideration of AQ co-benefits in policy debates will require improved evaluation techniques for *both* the climatic benefits and the air quality benefits of climate policy. Improving valuation of AQ co-benefits alone is unlikely to promote more stringent climate policy, even if it helps justify more stringent air quality regulation. In a more general sense, the effort to fully consider the value of co-benefits with vastly different risk characterizations, as well as time and spatial scales, foreshadows challenges in considering other co-benefits of actions to reduce climatic damages. Additional benefits may include effects on crop yields, acid deposition, macro-economic shocks, and geo-political conflict.

Acknowledgments

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Appendix

Table A.1. Studies estimating the co-benefits of climate change mitigation in developed countries.

				Value of co-benefits (2008$/$tCO_2$)		
	Study	Geography	Sectors included	Midrange	High	Low
1	Ayres and Walter (1991)	US	All	68	n.e.	n.e.
2	Ayres and Walter (1991)	Germany	All	128	n.e.	n.e.
3	Pearce (1992)	Norway	All	68	n.e.	n.e.
4	Pearce (1992)	UK	All	80	n.e.	n.e.
5	Alfsen *et al* (1992)	Norway	All	51	60	42
6	Holmes *et al* (1993)	US	Electric	4	n.e.	n.e.
7	Dowlatabadi *et al* (1993)	US	Electric	4	n.e.	n.e.
8	Goulder (1993)	US	All	44	n.e.	n.e.
9	Barker (1993)	UK	All	50	82	18
10	Barker (1993)	US	All	103	n.e.	n.e.
11	Barker (1993)	Norway	All	98	125	71
12	Viscusi *et al* (1994)	US	Electric	116	n.e.	n.e.
13	Rowe (1995)	US	Electric	31	n.e.	n.e.
14	Boyd *et al* (1995)	US	All	53	n.e.	n.e.
15	Palmer and Burtraw (1997)	US	Electric	6	n.e.	n.e.
16	EPA (1997)	US	Electric	31	n.e.	n.e.
17	Mccubbin (1999)	US	Electric	49	89	10
18	Caton and Constable (2000)	Canada	All	13	n.e.	n.e.
19	Syri *et al* (2001)	EU-15	All	n.e.	n.e.	n.e.
20	Han (2001)	Korea	All	80	91	69
21	Syri *et al* (2002)	Finland	All	n.e.	n.e.	n.e.
22	Bye *et al* (2002)	Nordic countries	All	18	26	11
23	Burtraw *et al* (2003)	US	Electric	17	18	15
24	Proost and Regemorter (2003)	Belgium	All	n.e.	n.e.	n.e.
25	Joh *et al* (2003)	Korea	All	2	n.e.	n.e.
26	van Vuuren *et al* (2006)	Europe	All	n.e.	n.e.	n.e.
27	Bollen *et al* (2009)	Netherlands	All	n.e.	n.e.	n.e.
28	Tollefsen *et al* (2009)	Europe	All	n.e.	n.e.	n.e.

Notes n.e. = not estimated in $/$CO_2$ terms. Especially useful previous reviews include: Ekins (1996), Burtraw *et al* (2003), IPCC (2007).

Table A.2. Studies estimating the co-benefits of climate change mitigation in developing countries.

				Value of co-benefits (2008$/$tCO_2$)		
	Study	Geography	Sectors included	Midrange	High	Low
29	Wang and Smith (1999)	China	Electric	n.e.	n.e.	n.e.
30	Cifuentes *et al* (2001)	Brazil	All	n.e.	n.e.	n.e.
31	Cifuentes *et al* (2001)	Mexico	All	n.e.	n.e.	n.e.
32	Bussolo and O'Connor (2001)	India	All	n.e.	n.e.	n.e.
33	O'Connor *et al* (2003)	China	All	n.e.	n.e.	n.e.
34	Dessus and O'Connor (2003)	Chile	All	n.e.	n.e.	n.e.
35	Aunan *et al* (2004)	China	Electric	36	n.e.	n.e.
36	Aunan *et al* (2004)	China	Electric	27	n.e.	n.e.
37	Aunan *et al* (2004)	China	Electric	36	n.e.	n.e.
38	Aunan *et al* (2004)	China	Electric	36	n.e.	n.e.
39	Aunan *et al* (2004)	China	Electric	98	n.e.	n.e.
40	Aunan *et al* (2004)	China	Electric	135	n.e.	n.e.
41	Kan *et al* (2004)	China	All	n.e.	n.e.	n.e.
42	Kan *et al* (2004)	China	All	n.e.	n.e.	n.e.
43	Morgenstern *et al* (2004)	China	Electric	119	196	43
44	West *et al* (2004)	Mexico	All	n.e.	n.e.	n.e.
45	McKinley *et al* (2005)	Mexico	All	n.e.	n.e.	n.e.
46	Li (2006)	Thailand	All	n.e.	n.e.	n.e.
47	Vennemo *et al* (2006)	China	Elec. & Industrial	n.e.	n.e.	n.e.
48	Zhang *et al* (2010)	China	All	n.e.	n.e.	n.e.

Notes n.e. = not estimated in $/$CO_2$ terms. Especially useful previous reviews include: Ekins (1996), Burtraw *et al* (2003), IPCC (2007).

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